Doktorarbeit

auf dem Gebiet der Recyclingwirtschaft

The Economics of Electronics Recycling:

New Approaches to Extended Producer Responsibility

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vorgelegt von

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A	areal spread of analyzed economy
$a_{i,(k)}$	segment-specific depollution cost parameter (of fraction k)
acc_t	average collection costs per ton
$acu_{i,t}$	average capacity utilization in segment i at time t
$adc_{i,(k)}$	average depollution costs per ton (of fraction k) in segment i
$adt_{i,(k)}$	average dismantling time per ton (of fraction k) in segment i
$amdc_i$	average material disposal costs per ton in segment i
$ARELP_i$	average relative end-of-life performance of products in segment i
$arsr_{i,t}$	average recycling service remunerations per ton in segment i at time t
$asc_{i,h}$	average sorting costs per ton in segment i for process requirements h
$asmr_i$	average secondary material revenues per ton in segment i
$ass_{i,t}$	average storage span in segment i at time t
atc_i	average transport costs per ton in segment i
atd_i	average transport distance between collector and recycler in segment i
$atdt_i$	average transport distance per tour in segment i
$aus_{i,t}$	average use span in segment i at time t
$b_{i,(k)}$	segment-specific depollution cost parameter (of fraction k)
$bsd_{i,t}$	unit-based apportioned absolute costs of EPR in segment i at time t
$c_{i,(k)}$	segment-specific depollution cost parameter (of fraction k)
$cc_{j,t}$	collection cost per ton of collection option j at time t
cpd_j	collection point density /(event frequency) of collection option j
cr	collection rate
d_i	segment-specific fine function paramter
$daic_{i,t}$	depreciation and amortisation per ton of archetype plant in segment i at time t
$dc_{i,t}$	dedicated collection in segment i at time t
$dcr_{i,t}$	dedicated collection ratio in segment i at time t
$deI_{i,t}$	disposed EEE from EEE in use in segment i at time t
$deII_{i,t}$	disposed EEE from stored EEE in segment i at time t

$dql_{i,(k)}$	depollution quality level (of fraction k) in segment i
DRP	decision-relevant profit
e_i	segment-specific fine function parameter
$EACC_{i,t}$	economy-wide collection costs in segment i at time t
$EADC_{i,t}$	economy-wide depollution costs in segment i at time t
$EASC_{i,t}$	economy-wide sorting costs in segment i at time t
$EATC_{i,t}$	economy-wide transport costs in segment i at time t
$EARSR_{i,t}$	economy-wide recycling service remunerations in segment i at time t
$edfi$	expected monetarized economic disadvantages from illegal export
$edfn$	expected monetarized economic disadvantages from non-compliance
$EMRCD_{i,t}$	economy-wide demand of MRCs in segment i at time t
$ee_{i,t}$	exported second hand EEE in segment i at time t
$eeprc_{i,t}$	EPR costs per ton in segment i at time t
$EEPRC_{i,t}$	economy-wide EPR costs in segment i at time t
$ehmr_{i,k}$	expected maximum mass of hazardous materials k recoverable in segment i
$ei_{i,t}$	illegal export in segment i at time t
$eir_{i,t}$	illegal export ratio in segment i at time t
$eirr$	illegal export recycling rate
$EMRCP_{i,t}$	economy-wide MRC costs in segment i at time t
$ERHM_{k,i,t}$	economy-wide recovery of fraction k in segment i at time t
ES	size of analyzed economy
$EMRCS_{i,t}$	economy-wide supply of MRCs in segment i at time t
$eu_{i,t}$	EEE in use in segment i at time t
f_i	segment-specific fine function parameter
fa	facility availability of archetype plant
$fccp_j$	fixed collection costs of collection option j
fcu_i	facility capacity utilization of archetype plant in segment i
fcy_i	facility capacity of archetype plant in segment i
$fic_{i,t}$	financing costs per ton of archetype plant in segment i at time t

f_k	classified fraction k
$fms_{i,k}$	output mass share of fraction k in segment i
fp_k	disposal price/secondary material price per ton of fraction k
$hmr_{i,k}$	mass of hazardous materials k actually recovered in segment i
hs	working hours per shift in an archetype plant
$hw_{i,t}$	disposal via household waste in segment i at time t
$hwr_{i,t}$	household bin ratio in segment i at time t
$id_{i,t}$	illegal dumping in segment i at time t
$idr_{i,t}$	illegal dumping ratio in segment i at time t
$ie_{i,t}$	imported second hand EEE in segment i at time t
$iomrc_{i,t}$	individual MRC producer obligation in segment i at time t
$la_{i,t}$	landfill and incineration in segment i at time t
$lar_{i,t}$	landfill and incineration ratio in segment i at time t
lf_i	load factor in segment i
lr	labor cost rate
mdc_i	costs for material disposal in segment i
$mf_{i,k}$	produced mass of fraction k in segment i
$mr_{i,t}$	material recycling in segment i at time t
$MRCa_{i,t}$	ask price for MRCs of central clearing house in segment i at time t
$MRCb_{i,t}$	bid price for MRCs of central clearing house in segment i at time t
$mrcp_{i,t}$	material recovery certificate price in segment i at time t
$mrcd_{i,t}$	MRC demand per ton sales in segment i at time t
$mrcs_i$	MRC supply per ton input in segment i
$mrr_{i,t}$	material recycling ratio in segment i at time t
$ne_{i,t}$	new EEE in segment i at time t
$oe_{i,t}$	obsolescent EEE in segment i at time t
$ofc_{i,t}$	other fixed costs per ton of archetype plant in segment i at time t
$PCML$	performance in closing material loops
$pncn$	probability to be checked and discovered as non-compliant

$pnci$	probability to be checked and discovered as an illegal exporter
$rec_{(i)}$	material recycling rate (in segment i)
rfd_i	recycling facility density for segment i
RF_i	amount of recycling facilities for segment i
$rr_{i,t}$	remanufacturing, repair, and reuse in segment i at time t
rrc	repair or remanufacturing costs per ton
$rrr_{i,t}$	remanufacturing, repair, and reuse ratio in segment i at time t
$rsr_{i,t}$	recycling service remunerations per ton in segment i at time t
rsv	resale value per ton
$s_{i,(t)}$	static/dynamic MRC market steering lever in segment i (at time t)
$sa_{i,t,g}$	share of appliances of type g in segment i at time t
$scc_{i,h}$	sorting coefficient for segment i and process requirements h
sco_j	class-specific share of collection option j
sec	shredding equipment capacity in an archetype plant
sf_k	scoring factor of fraction k
shc	shipment costs
$sher_{i,t}$	second hand export ratio in segment i at time t
$shir_{i,t}$	second hand import ratio in segment i at time t
sii	percentage shredder input in an archetype plant
sme	secondary materials from illegally exported wastes
smr_i	secondary material revenues in segment i
smm	secondary materials from material recycling
$so_{i,t}$	stored obsolete or broken EEE in segment i at time t
spr_h	set h of sorting process requirements
$sr_{i,t}$	storage ratio in segment i at time t
ssd	shredding shifts per day in an archetype plant
tcf	transport cost factor
uf	utilization factor
v	detour factor
$vccp_j$	variable collection costs of collection option j

vco_i	variable costs from operations of archetype plant in segment i
$WACC$	industry-specific weighted average cost of capital
wd	working days per year in an archetype plant
wsr	WEEE sales revenues
x	WEEE range at a recycler’s disposal

Abbreviations

ARF	Advanced recycling fee
B2B	Business-to-Business
B2C	Business-to-Consumer
C&C	Command & Control
CFA	Cooling and Freezing Appliances
CFC	Chlorofluorocarbons
cm^2	sqare centimeters
CO^2	Carbon dioxide
CPR	Collective Producer Responsibility
CRT	Cathode ray tube
DfD	Design for Disassembly
DfE	Design for Environment
DfR	Design for Recycling
€	Euro
EAR	Elektro Altgeräte Register
EEE	Electrical and electronic equipment
e.g.	for example
ElektroG	Elektrogesetz
EPR	Extended Producer Responsibility
et al.	and others
EU	European Union
EUA	EU Allowances
e-waste	Electronic waste
HC	Hydrocarbons
HCFC	Hydrochlorofluorocarbons

HFC	Hydrofluorocarbons
IE	Industrial Equipment
IPR	Individual Producer Responsibility
IT	Information technology
kg	kilogram
LCA	Life Cycle Assessment
LCD	Liquid crystal display
LE	Lighting equipment
LHA	Large household appliances
M	Million
m^2	square meter
MRC	Material recovery certificate
NEPSI	National Electronics Product Stewardship Initiative
NGO	Non-governmental organization
NH_3	Ammonia
OEM	Original equipment manufacturer
p.	page
PC	Personal computer
PCB	Polychlorinated biphenyl
PRO	Producer responsibility organization
PUR	Polyurethane
ROHS	Restrictions on hazardous substances
R XXX	Gas classification
SENS	Stiftung Entsorgung Schweiz
SHA	Small household appliances
STEP	Solving the e-waste problem

SWICO	Schweizerischer Wirtschaftsverband der Informations-, Kommunikations- und Organisationstechnik
UK	United Kingdom
US	United States
VOC	Volatile organic compound
VREG	Verordnung über Rückgabe, Rücknahme und die Entsorgung elektrischer und elektronischer Geräte
WACC	Weighted average cost of capital
WEEE	Waste from electrical and electronic equipment

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After the start of my industry survey in Europe, I happened to present my research work at SENS in Zurich. Here, I encountered the eventual “practice father” of my doctoral thesis, Dr. Robert Hediger, who presumably deserves the biggest thanks in this section. Robert did not only allow me to embed my research activities in SENS’ project work which served as a door opener to industry and its stakeholders. Moreover, he gave me the possibilities to continuously interact with SENS experts while developing a practice-oriented regulation instrument. In this regard, I especially want to thank Dr. Erhard Hug and Dr. Ueli Kasser for their time and efforts in the last year. Last but not least, financial support from SENS allowed me to conduct a meaningful analysis of the industry over a longer period of time as well as to finalize my PhD thesis. Again, I want to state my honest gratitude to the people that made this possible: Dr. Robert Hediger and Dr. Andreas Roethlisberger.

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1 Introduction

1.1 The Challenges of Electronics Recycling

The treatment of waste from electrical and electronic equipment (WEEE) has been subject to growing concern in the last decade and has recently lead to vigorous action among legislation setting bodies across the globe. Waste from electrical and electronic equipment (WEEE) is among the fastest growing categories of municipal solid waste (Cairns 2005) and constitutes approximately 4% of the overall municipal solid waste stream (ETC/WMF 2002). Facing approximately 10 million tons of WEEE in the European Union (Knoth et al. 2001), sound waste management has developed into a significant challenge for all institutions involved in the end-of-life management of electronic devices. The overwhelming WEEE mass contributes to a growing volume of toxic inputs into the local waste streams. Dependant on the treatment process, WEEE can have a considerably detrimental impact on ecosystems and their environment. Yet, e-waste or WEEE is not only "a problem to be solved". Indisputable, the materials contained in WEEE are of considerable value and if e-waste as a supply stream was in its size comparable to the volumes processed in traditional mining operations, WEEE would represent one of the most attractive veins of ore to be located on the globe. Given its economic attractiveness as a source of valuable raw materials on the one hand and its potentially harmful impact on ecosystems on the other, a key concern for sound and well thought-out WEEE management are the flows of these materials in a cycle economy and treatment strategies applied to shape and influence these flows. A respective analysis must focus on end-of-life processes as well as the upstream part of the value chain which includes logistics, sorting, and collection strategies.

During the last decades, traditional end-of-life options like landfill and incineration were prevailing in the treatment of WEEE. However, recycling, remanufacturing, and reuse have recently been strongly encouraged. Arguments for recycling are manifold. The most obvious and simple reason is the avoidance of waste which is strongly connected to the scarcity of landfill space in most countries (Ackerman 1997). Reductions in litter and improper disposal are also brought forward to support recycling. Industrial ecologists put emphasis on the vital role of recycling and reuse for the closing of material loops that is deemed as a crucial contribution to a sustainable economy. Their rationale is supported by the thought that "there are only two possible long-run fates for materials – dissipative loss and recycling or reuse" (Richards et al. 1994, p.8). This is aligned with Porter's notion who suggests that recycling is a contribution to an “intergenerational smoothing of living standards” (Porter 2002, p.126). Environmentalists’ main points are reductions in energy use and related emissions through decreased virgin material exploitation. Economists draw attention to its potential to create jobs and its profitability compared to landfilling. Brandes adds a strategic rationale relevant for different countries across the globe (Brandes 1997): “For a country like the Federal Republic of Germany which has only a few mineral resources of its own, (…), cycle management is an essential prerequisite for economic success.”

Given all these arguments, it does not come as a surprise that recycling has gained a predominant role in the end-of-life processes currently observed in industrialized countries. However, recycling is a vaguely defined term that serves as a melting pot for numerous end-of-life utilization strategies applied under the recycling signet. Several key features of recycling processes are to be taken into account when assessing the quality of recycling strategies, e.g separation and proper handling of hazardous materials, workers’ safety and health standards, treatment depth, recycling rates, and the level of material reapplication. Observations of the end-of-life management of electrical and electronic equipment (EEE) provide evidence that some progress has been achieved in recycling practice, especially among industrialized countries in Europe, North America and Asia. Nonetheless, a clear assignment of financial and organizational responsibilities is necessary to establish upstream value chain infrastructure and high-quality recycling processes.

Extended producer responsibility (EPR) has widely been recognized as a suitable policy means to tackle this issue (Lee et al. 2004, Lindhqvist et al. 2003, Mayers et al. 2005, Roine et al. 2006). In addition to the clarification of financial and organizational responsibilities, EPR also holds the potential to set incentives for producers to incorporate end-of-life aspects already in the design stage of new electronic products, also referred to as the product design feedback loop. However, expecting industry to fully implement sustainable end-of-life and recycling practices voluntarily in a free market without clearly defined requirements is quixotic. In spite of growing awareness and the incorporation of environmental and sustainability concerns in corporate principles (Neal et al. 2001, Stevels 2002), it is necessary to provide adequate incentive structures for all involved market participants via well-designed regulatory policies (Richards et al. 1994). This notion is supported by a study of two Norwegian scientists that reveals the crucial role of environmental regulations (Lee et al. 2004). As shown in Figure 1, their survey participants answered that the main driving forces for companies to start making green technological changes arise from environmental regulations in the European Union.

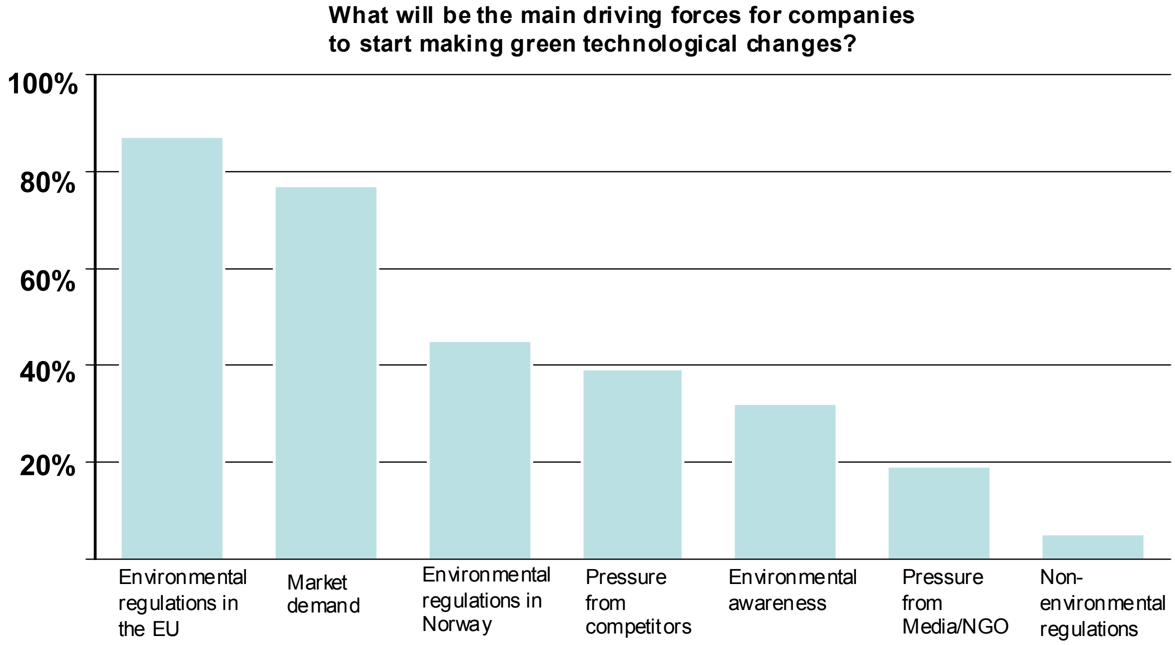

Figure 1: Main driving forces for green technological changes (Lee et al. 2004)

During EPR's infancy in the last decade, pioneering initiatives in some countries in Europe and Asia have demonstrated EPR's feasibility. First experiences were gained in Switzerland where take-back systems in order to recycle WEEE were established beginning in 1992. A few years later, EPR was also established by industry players in other countries under the pressure of environmental authorities. In Europe, the Netherlands, Belgium, Norway, and Sweden prepared and established producer responsibility organizations for WEEE recycling between 1997 and 1999. In Asia, Japanese producers set up two industry groups to assume responsibility for the end-of-life management of electronics under the "Home Appliances Recycling Law" in 2001. Taiwan took action in the same year and Korea introduced regulations shortly afterwards. In 2003, Europe's WEEE directive marked a milestone in the evolution of EPR for electronics. 25 countries were supposed to implement national systems where electronics manufacturers and importers are responsible for the financing of take-back and recycling of at least 4 kg WEEE/capita. Depollution standards were set for hazardous WEEE and the achievement of certain recycling and recovery quotas was mandated. (COM 2003, Hieronymi 2001, Hieronymi 2004, Takeda 2003)

However, the EU directive has been experiencing scathing criticism because a consistent mapping of theoretical goals and practical means has not been achieved (Bohr 2006, Andersen et al. 2006, van Rossem et al. 2006, Stevels 2003). Vaguely defined stipulations have created a versatile WEEE recycling system landscape across the EU. The directive's stipulations are interpreted differently in various member states and their enforcement stringency and speed of implementation differs considerably (Perchards 2005, Savage et al. 2006). This jeopardizes a level playing field in the EU, compromises a common market, and leads to considerable uncertainty among producers about their obligations. In addition, recycling experts have criticized chosen "command-and-control" approaches under the WEEE directive in the European Union for several environmental and economic shortcomings (Stevels 2005, Griese et al. 2004, Hornberger et al. 2004). This affects, for example, issues connected with the level of reapplication of materials, the incentives for technological innovation in recycling, or an appropriate reflection of environmental burdens associated with different materials (Jehle 2005, Huisman 2003, Stevels 2003). Additionally, some technical shortcomings are to be added to the agenda for the directive's revision in 2008, e.g. an unskillful grouping of products that complicates reporting and monitoring in practice.

Regarding its actual implementation, it is fair to say that the directive itself is not responsible for absurdities observed in European implementation and enforcement practice. While the directive only provides a framework that is supposed to be clarified and supported with additional rules in the national context, strong lobbying and conflicts of interest have turned some national EPR approaches into peculiar systems with arbitrary rules and mechanisms. Such complicated mechanisms were partly created by industry bodies itself and the blame can not be put on environmental authorities. However, it is in the responsibility of a proficient and capable regulatory authority to carefully assess implementation guidelines and rules with respect to their feasibility in practice. In the WEEE case, several parameters with respect to monitoring, collection, logistics, treatment, measurability and traceability of outputs, as well as organizational requirements and other boundary issues, are to be carefully chosen. Nonetheless, the major responsibility lies in the framework and accordingly on the EU level. Concise and clear instructions help shaping national implementation approaches. Economics and resulting incentive structures must be taken into account in order to achieve environmental goals. Again, the current directive is characterized by a poor reflection of considerations about economies of scale and diseconomies of scope – both have a substantial impact on the economics of WEEE recycling.

With a first major review of the EU WEEE directive approaching, it is apposite to assess the potential of alternative EPR approaches. A changing business environment and experiences gained with current regulatory approaches prompt a redesign of WEEE regulation. Free market tools shall prove to serve as promising alternatives to command-and-control regulation. Several scientists advocate for a broader use of such tools to tackle environmental issues, and a strong emphasis is placed on the necessity to internalize externalities in a cost-efficient way. Especially certificate markets have been subject to vigorous academic interest but their suitability for WEEE recycling has not been demonstrated yet. These tasks have been addressed in a joint research effort of the Technical University Berlin, the Massachusetts Institute of Technology, and SENS (Stiftung Entsorgung Schweiz), a recycling system operator in Switzerland. Analytical research has been supported by a comprehensive international industry survey with over 40 on-site visits, expert interviews, and discussions in 9 countries. Based on both empirical and theoretical work, approaches using material recovery certificates (MRCs) are recommended to implement the concept of EPR. The supporting rationale and the concrete certificate market design will be thoroughly illustrated in the next chapters, embedded in a comprehensive portrait of the WEEE recycling industry. However, before presenting actual contents and results of this work, an overview of research goals identified, research strategy and methods, and the resulting structure of this thesis shall enable the reader to orient himself and choose his topics of interest.

1.2 Research Goals

Extended Producer Responsibility is ambivalently perceived by theorists and practitioners. An almost unchallenged recognition in theory encounters vigorous criticism in implementation practice. Where does this gaping disparity stem from? Are there reasons to believe that chosen policy instruments fail to reflect EPR's original goals? And if so, how can the EPR concept be taken from theory to practice without compromising its actual goals?

A clarification and definition of EPR's goals in the electronics recycling context serves as a basis to address these questions. Furthermore, this basis allows *deriving an assessment framework for concrete policy instruments.* Given such a framework, the main research goal is identified as follows:

> *How should EPR policy instruments for electronics recycling be designed?*

This implies initial research on the following aspects:

> *Which policy instruments are potentially available to regulate electronics recycling and which instruments are used in practice?*
>
> *What are the strenghts and weaknesses of existing and potential approaches?*

What are the manifold design options for the different policy instruments and which incentive structures are created and faced by affected players and stakeholders under a policy?

What is the impact of a policy instrument on the recycling system and the behavior of players and stakeholders?

Achieving meaningful results at this stage already requires *basing the analysis on a sound understanding of the industry,* including intentions and interests of major players and stakeholders, functioning of the recycling value chain, material flows in the recycling systems, and its underlying economics. After identifying the most promising policy instruments, it is necessary to

- *devise and elaborate practical designs of applicable policy instruments,*
- *assess their effects on electronics recycling systems,*
- *contrast them with existing regulation approaches.*

Based on these results, qualitative and quantitative analysis should enable the researcher to

- *model electronics recycling systems and the impact of different policy designs,*
- *simulate the outcomes of different policy instruments in a defined environment,*
- *quantitatively demonstrate the superiority of recommended policy designs,*
- *give a concrete policy design recommendation.*

Several results from the pursuit of these research goals are presented in this thesis according to the structure portrayed in chapter 1.4. The research strategy and applied methods is described in the following section.

1.3 Research Strategy and Methods

The research work underlying this thesis has been executed in three stages. The first stage comprised studies on regulation policy instruments and an analysis of current EPR approaches for electronics recycling. An *extensive literature review* served to develop an overview of established policy instruments and *telephone interviews with selected industry experts* were used to corroborate or repudiate identified strengths and weaknesses of existing approaches in the electronics recycling case.

The first stage revealed that current policies caused major system inefficiencies and did poorly account for the prevailing incentive structures in industry. As a result, a strong need for alternative and more goal-focused instruments was identified. Preliminary assessments indicated a great potential of certificate market approaches. This led to the working hypothesis that a specifically designed certificate market instrument would create better environmental and economic results in practice. Given EPR's differing perception in theory and practice, it became clear that sound policy recommendations and a quantification of their impacts required a profound understanding of the industry.

In the second stage, a *comprehensive industry survey*[1] was conducted to complement the theoretical work and to ensure a sound practical anchorage of the research results and derived recommendations. Over 30 o*n-site visits and expert discussions* with several stakeholders, among them producer responsibility organizations, take-back system operators, recyclers, producers, regulatory authorities, and secondary recyclers in 8 countries were used to gather data and information and to include industry expertise for the assessment and design of policy instruments. The survey served two main purposes: first to obtain feedback on certain policy designs and ideas, and second to enable the researcher to develop a comprehensive system model which would allow simulating systems and the impacts of different policies. In order to access crucial proprietary information not explicitly revealed in this thesis but used for a sound overall judgement, *closer working relationships were developed with selected players and stakeholders* from the survey. This involved several meetings, visits, and continuous interaction over a longer period of time. Such closer working relationships were established with producer responsibility organizations, recyclers, and technical control experts, and allowed to continously support the evolution of an alternative regulation approach.

A combination of *quantitative and qualitative methods* was applied in the survey and a *mix of primary and secondary data* was collected during its execution. The use of various methods in one study, also referred to as *triangulation*, was necessary in order to neutralize distorting effects from applied data collection methods (Saunders et al. 2003) and from interests of information providers alike. For example, intransparent secondary material markets required to cross-check recycler information on fraction pricing with data provided by secondary treatment companies.

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[1] A complete list of visits and participating companies and institutions is provided in annex I.

The actual industry survey mainly consisted of *semi-structured expert interviews* and *on-site facility tours. A judgemental sampling* was made using the expertise of collaborating institutions in order to determine suitable companies, institutions, and interview partners for an inclusion in the survey. The basis for the interviews in each stakeholder group were lists of themes and questions of which not all had to be addressed by the interviewee. This allowed to spend more time on topics where the interviewee had specific expertise and to avoid themes where only superficial information was available. Figure 2 shows the key areas covered by the survey, the countries involved, and project goals pursued.

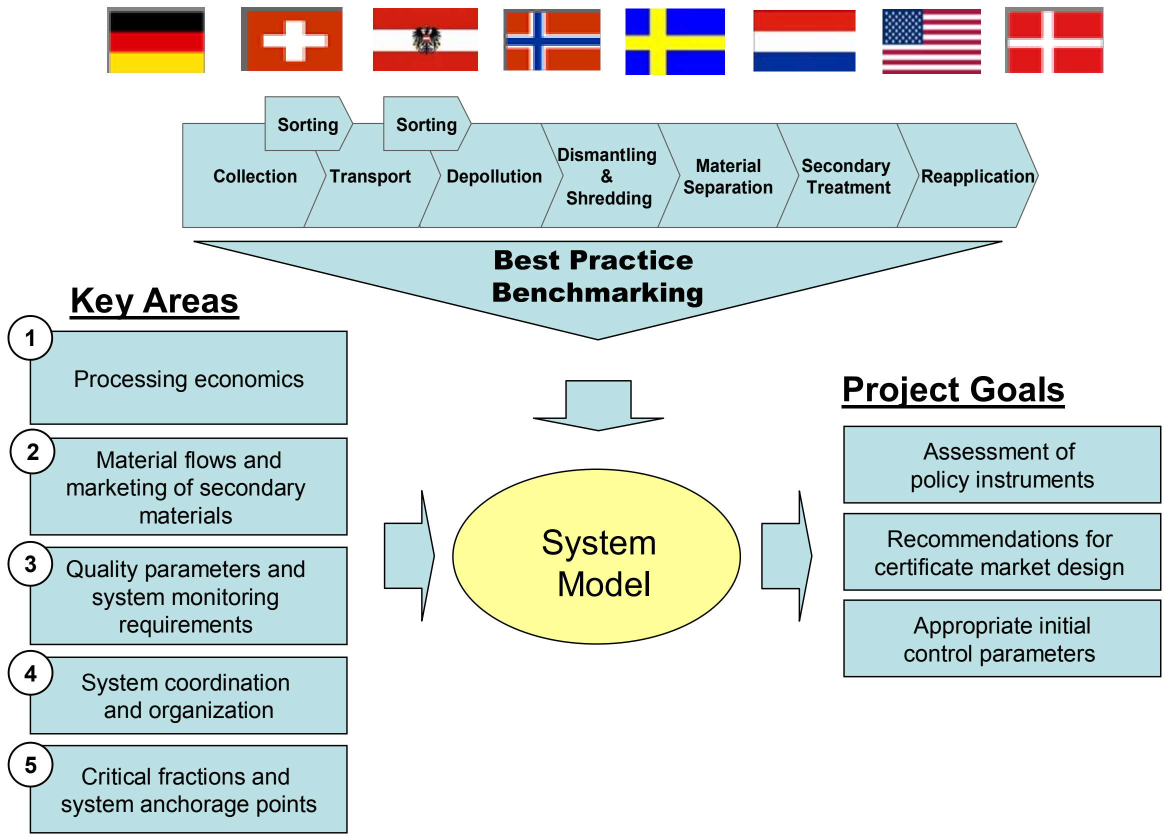

Figure 2: Key areas and project goals of the electronics recycling industry survey

The third work stage was initiated in parallel to the industry survey and comprised the elaboration of specific policy instruments and the development of a material flow model and an economic model. The combination of both models is further referred to as the system model. The models were continuously improved and served to simulate the impacts of different instruments and design variants. System stocks and flows were visualized and simulated using the system dynamics software Vensim and the behavior of major players was modeled assuming profit maximization as a key driver for decision-making. The system model allows for a quantified analysis of the impacts of policy designs and a prediction of the magnitude of system stocks and flows in case study simulations. In order to demonstrate the functioning of the instruments recommended, case studies were conducted of which some results are presented in this thesis.

1.4 Thesis Structure

The presentation of research work in this thesis attempts to take the background and expectations of both academia and practitioners into account. Figure 3 shows the structure of the thesis and the contents and results to be expected from each chapter.

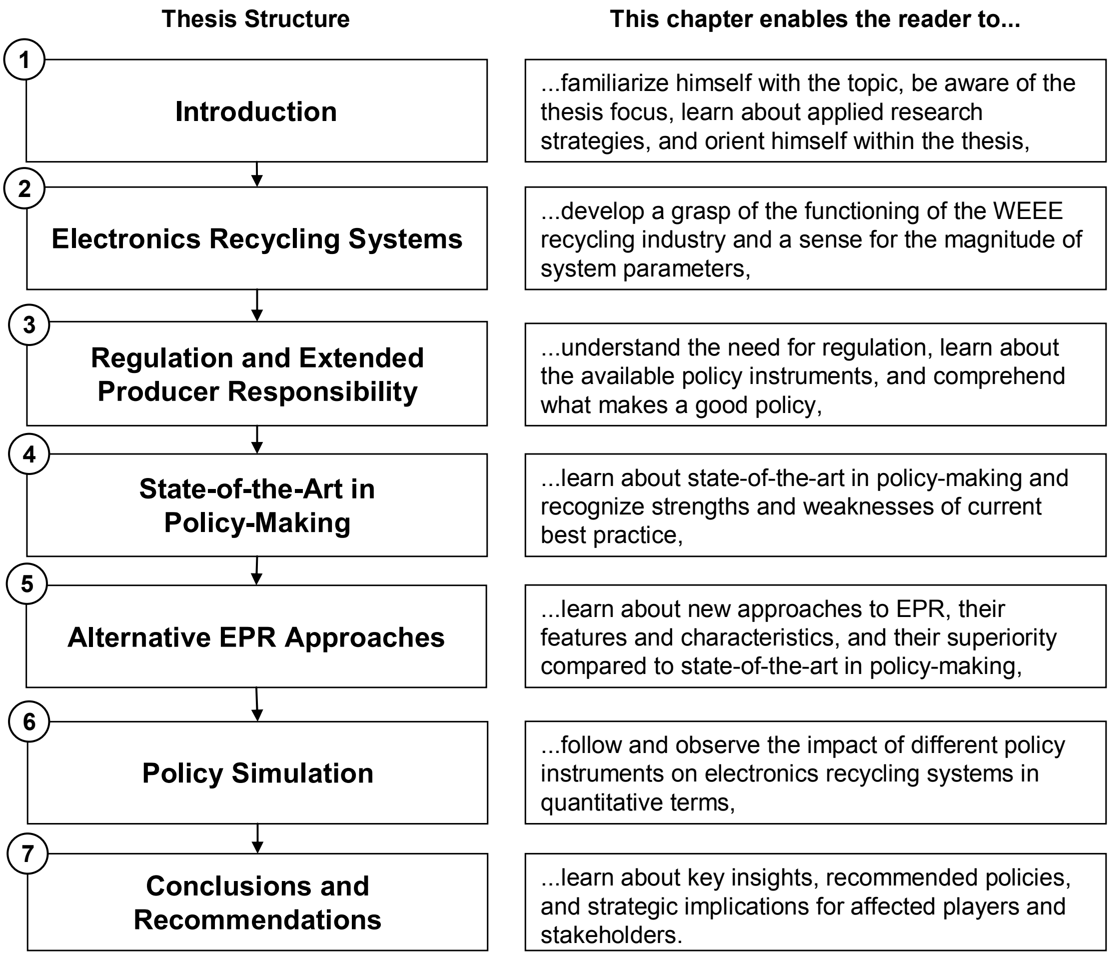

Figure 3: Thesis structure

After an introduction, the reader finds a comprehensive portrait of electronics recycling systems in chapter 2. This chapter serves to aquaint the reader with the practical functioning of the WEEE industry and to introduce formalized model structures subsequently used for system simulations. The practical knowledge is crucial for a sound understanding of the conclusions drawn from the policy analysis in chapter 4. Before this analysis, chapter 3 deals with the characteristics and goals of WEEE regulation and provides an overview of the features of extended producer responsibility in order to guide further analysis and evaluation.

Chapter 4 identifies a need for new approaches and policy instruments. Alternative regulatory approaches and the development of an EPR system with material recovery certificates (MRCs) in the context of electronics are described in chapter 5. Strenghts and weaknesses of the recommended approach are contrasted with the characteristics of current best regulation practice on a qualitative level. A quantitative analysis follows afterwards in chapter 6. The analysis is based on a comprehensive system model and data excerpts from a case study simulation are presented. The results quantify the impacts of differing policies and allow for a prognosis of costs and material flows in the underlying economy. Chapter 7 recapitulates the key insights and reveals strategic implications for affected players and stakeholders under the recommended MRC approach.

2 Electronics Recycling Systems

A substantiated and concise assessment of regulatory policies requires a profound understanding of the regulated industry's functioning. This chapter will provide a comprehensive picture of the electronics recycling industry, including players and stakeholders with their interests, tasks, and roles, EEE and WEEE characteristics, and the recycling value chain. Where appropriate, the reader will already be familiarized with the formalized modeling structures used for policy simulations in chapter 6 and some data excerpts will be embedded to allow the reader to develop a sense for the magnitude of key variables and parameters.

2.1 Players and Stakeholders

The cycle economy in the electronics industry is run and shaped by several actors. "Players" take part in the material flow system (squares); "stakeholders" only influence the players' behavior (circles). Figure 4 presents the main players and stakeholders and the system's material flows (curved arrows) in a cycle model.

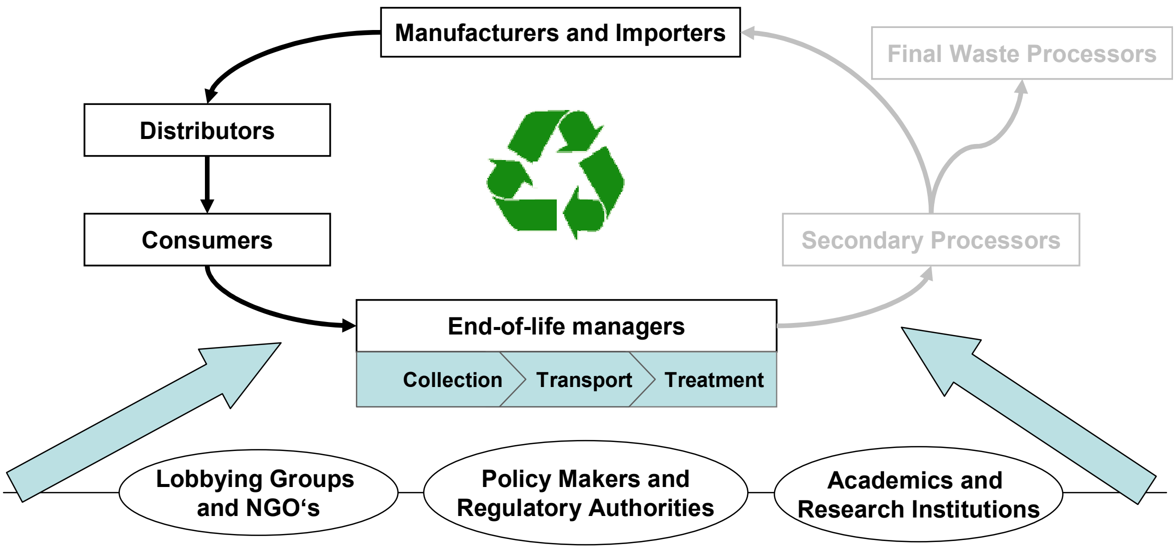

Figure 4: Model of a cycle economy with players and stakeholders

Both players' and stakeholders' role, responsibility, and interests are summarized in brief:

- **Manufacturers and importers** are protagonists and traditionally responsible for design, component selection, manufacturing, and assembly of a product. They are interested in minimizing end-of-life management costs and ensuring legal compliance. Theoretically, manufacturers can influence their end-of-life management costs via well-designed products if end-of-life design efforts are connected to resulting monetary advantages in the recycling process. The recyclability of a product is strongly affected by its design, for example through disassembly times and unlocking properties (Ohlendorf et al. 2003, Herrmann et al. 2006, Mueller et al. 2003). This is reflected in several tools (Rifer et al. 2003, Herrmann et al. 2002) developed to assist design-for-recyclability (DfR) and design-for-disassembly (DfD) which are the catchwords for such approaches. Both DfR and DfD can be interpreted as part of the broader conceptual idea pursued with design-for-environment (DfE) (Griese et al. 1997, Fullerton et. al. 1998). However, even in the case of thoroughly enforced individual producer responsibility, the economic incentive is very weak due to several reasons. In the case of collective responsibility without other flanking measures, individual end-of-life management costs and a manufacturer's product design are currently completely independent.

In general, end-of-life management would be simplified if manufacturers provided information on product structure, disassembly paths and material content that enabled recyclers to apply better processing strategies. Furthermore, more reuse or remanufacturing of components and the integration of recycled materials in new products would considerably impact the performance in closing material loops (Cairns 2005, Housman 1994). However, manufacturers naturally prioritize functionality and customer value which are strategically more important in order to attain and preserve economic success (Stevels 2005).

- **Distributors** are typically large retailers, specialist shops, direct importers, or internet shops. They play a crucial role when it comes to product market introduction, product listing, placement, and eventually marketing & sales. In the light of green purchasing strategies partly found and mandated among governmental and public organizations, it is astonishing that some sort of "extended distributor responsibility" hasn't gained more attention. Besides their actual role as vendor or provider of distribution channels for electronic products, distributors can participate in the collection of WEEE through take-back opportunities via their distribution channels. In the case of an advanced recycling fee, billing, collecting, and money transfer to service providers is naturally in their responsibility, too.

- **Consumers** and their needs fuel the EEE economy. As already reflected in the paragraph on manufacturers, the consumers' priorities lie on product functionality, style, and price (Stevels 2005). Environmental aspects have not yet proven to be a promising marketing aspect (Berner 2006). Consumers indirectly pay the bill for end-of-life management and are accordingly interested in low end-of-life management costs. However, efforts for product disposal are to be minimized as well and consumers therefore ask for convenient collection infrastructures.

- Typically, producer responsibility organizations, municipalities, waste management providers, transport companies, recyclers, dismantlers, and scrap brokers are found among the **end-of-life managers**. Figure 5 shows exemplified archetypes of the end-of-life managers in an activity board for the WEEE industry. The board integrates value chain steps horizontally and business sections (see chapter 2.2) vertically.

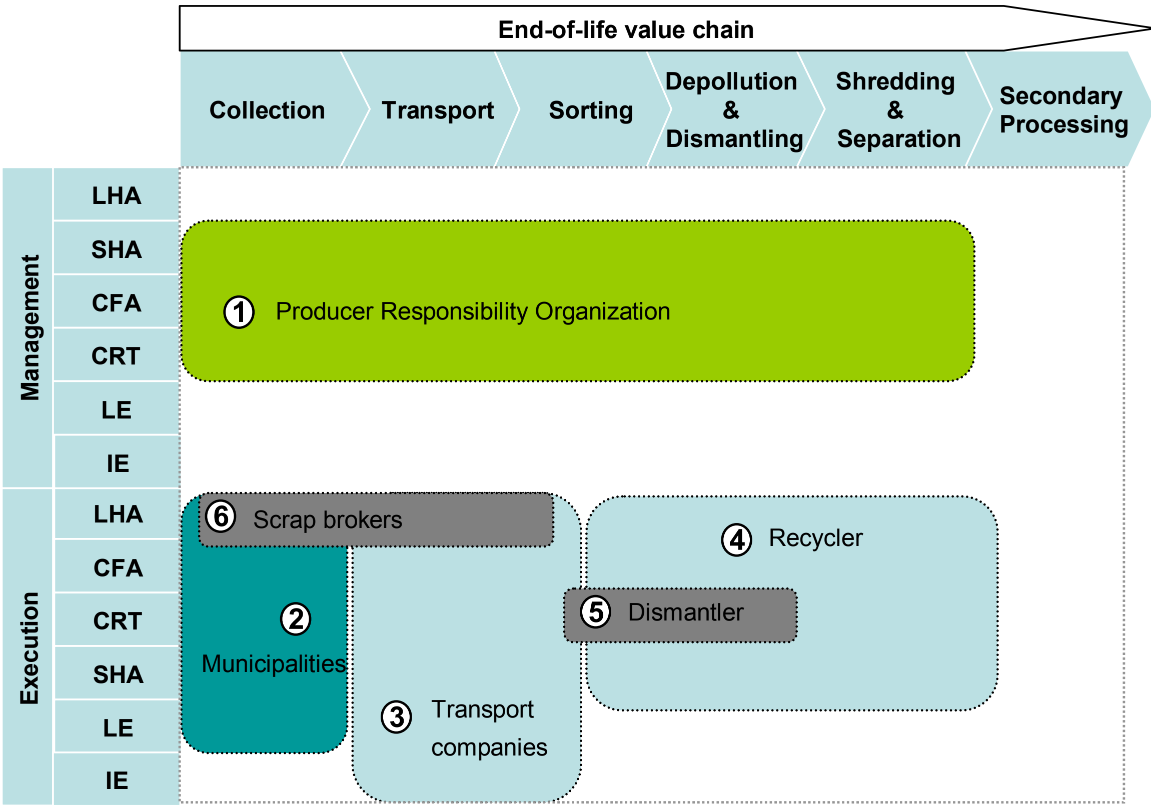

Figure 5: Activity board for end-of-life managers

PROs and waste management providers understand themselves as service providers to the manufacturers and importers and attempt to facilitate and ensure their legal compliance by performing several tasks and services in the electronics end-of-life sphere. Typically, this includes waste management and determination of subcontractors for the execution of physical tasks in the end-of-life sphere, quality assurance, reporting to authorities, and public relations. Municipalities were historically confronted with WEEE streams and responsible for their management. Under extended producer responsibility, most responsibilities are transferred to other players and municipalities are solely affected by the need for collection infrastructures which also affects the service level provided to their citizens. Transport companies, recyclers, and dismantlers perform end-of-life services and attempt to maximize their profit. The same holds for scrap brokers that interfere with the other players' businesses on different levels, buying and selling WEEE materials in a somewhat informal way.

Three large stakeholder groups can be identified which influence or alter the system through interaction at different stages. They can be classified as follows:

- **Policy makers and regulatory authorities** have acknowledged the importance of WEEE end-of-life management. They promote prevention, reuse, recycling and other forms of WEEE recovery (COM 2003, EAJ 2000) under the flagship sustainable development. Policy makers define goals and provide legislation for the functioning of WEEE recycling systems. Regulatory authorities are to devise regulatory frameworks that achieve the defined goals via well-designed incentive structures.
- **Lobbying groups** promote particular or industry interests. **NGO's** usually intend to integrate societal concerns in the evolution process of legislatory frameworks. All players have build lobbying associations that work toward their interest in the shaping of public or governmental opinion.
- **Academics and research institutions** contribute to continuous development and improvement of several system characteristics. This affects, for example, research on policy designs that shape the system (strategic), processing technologies, separation techniques, and upgrading of secondary materials (technological), or studies on logistical or management optimization (organizational). Their guidance is welcomed for decision-making on a political level.

2.2 Electrical and Electronic Equipment (EEE)

2.2.1 Definition, Types and Classification

"Electrical and electronic equipment means equipment which is dependent on electric currents or electromagnetic fields in order to work properly and equipment for the generation, transfer, and measurement of such currents and fields" (COM 2003). Table 1 displays a classification of products falling under this definition, derived and modified from Annex 1A of the WEEE directive (COM 2003).

Large (non-industrial) appliances [i=1]	Small (non-industrial) appliances [i=2]	Cooling and freezing appliances [i=3]
• Washing machines • Clothes dryers • Dish washing machines • Cooking equipment • Electric stoves and hot plates • Microwaves • Electric heating appliances • Electric fans • Automatic dispensers • Other large appliances	• Vacuum cleaners • PCs and Laptops • Radio sets • Video cameras • Toaster, Fryers • Telephones and cell phones • Coffee machines • Game consoles • Printers • Other small appliances	• Refrigerators • Freezers • Other cooling and freezing appliances
CRT devices [i=4]	**Lighting equipment [i=5]**	**Industrial EEE [i=6]**
• Television sets • Monitors • Other CRT devices	• Luminaires or fluorescent lamps • High intensity discharge lamps • Low pressure sodium lamps • Other lighting equipment	• Laboratory equipment • Heavy industry machinery • Radiotherapy equipment • Industrial control equipment • Industrial steering equipment • Other industrial equipment

Table 1: A classification of electrical and electronic equipment (EEE)

In order to allow for a consistent allocation of EEE to groups, the classification presented here distinguishes "industrial" and "non-industrial" EEE. Any "electrical or electronic equipment that is of no use to any private consumer" is defined as "industrial EEE". Accordingly, devices that can be useful to both private and business consumers are classified as "non-industrial EEE". Non-industrial EEE is further subclassified according to WEEE treatment streams in recycling systems. Currently, state-of-the art WEEE recycling facilities treat non-industrial WEEE in at least 5 separate groups due to technological and economic framework conditions. This leads to 6 product segments: large appliances (LHA), small appliances (SHA), cooling and freezing appliances (CFA), cathode ray tube devices (CRTs), lighting equipment (LE), and industrial EEE (IE).

The environmental and economic end-of-life performance of EEE is strongly affected by its design. A product's design and its materials determine the costs and revenues from its recycling as well as the material and energy recovery achieved by a certain recycling strategy (Kahmeyer et al. 1996, Plötz et al. 1994). Product properties such as material selection in terms of homogeneity/mix, toxicity, recyclability and compatibility, as well as product structure and joining techniques strongly influence the product's ability to be recycled and disassembled as well as the effort necessary to apply these processes (Santochi et al. 2001).

2.2.2 Modeling EEE Stocks and Flows

The basis for a profound description and long-term prediction of electronics recycling systems is a thorough tracking of EEE material flows over several life-cycle stages. For the purposes of this thesis, these flows are captured and described using a material flow model implemented with the simulation software Vensim[2]. Producer responsibility starts with the sale of new products to consumers. Therefore, modeling begins with an analysis of material flows in the consumer sphere. Figure 6 presents the first steps for the model construction.

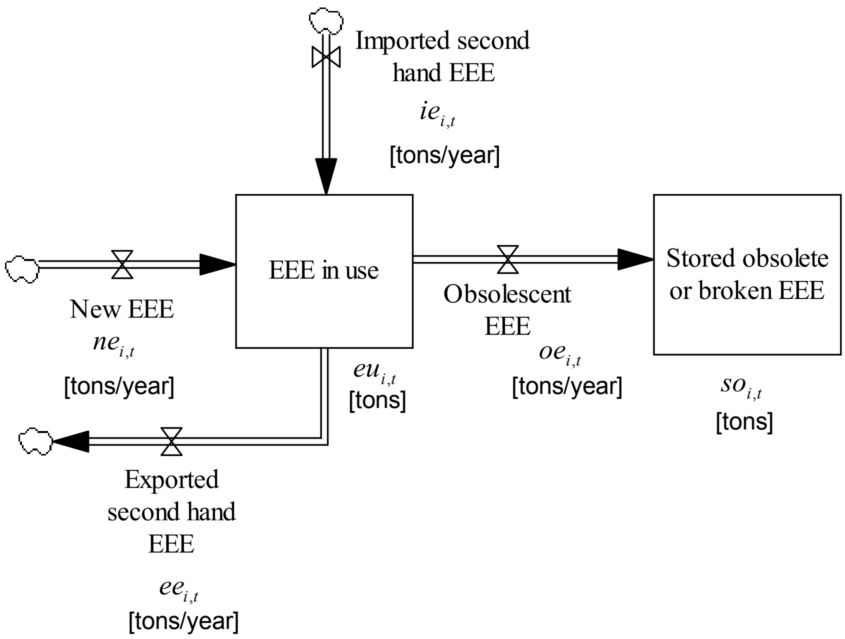

Figure 6: EEE stocks and flows in the consumer sphere

Sales amounts of new EEE in each segment i at time t are depicted with $ne_{i,t}$. The new EEE $ne_{i,t}$ is measured in [tons/year] and accordingly modeled as a flow. Flows are shown as arrows with valves in the visual model representation. Imported second hand EEE $ie_{i,t}$ [tons/year], exported second hand EEE $ee_{i,t}$ [tons/year], and obsolescent EEE $oe_{i,t}$ [tons/year] are also modeled as flows. EEE in use $eu_{i,t}$ as well as stored obsolete or broken

[2] Vensim is a system dynamics modeling software. A comprehensive introduction into modeling systems with Vensim is provided by (Sterman 2000).

EEE $so_{i,t}$ are stocks, represented as boxes and measured in [tons]. System borders are visualized with clouds which indicate that the respective origin or destination of a flow is not relevant for the modeling purpose. For example, the origin of second hand imports is not relevant for the eventual prediction of WEEE flows and the influence of policy instruments on such flows and their treatment.

New EEE depicts imported or locally manufactured new products sold in an analyzed economy. Imported second hand EEE implies used products imported from any source outside the analyzed economy whereas exported second hand EEE represents consumer-initiated used equipment exports. Second hand sales within the underlying economy are not separately displayed as flows but reflected in the average use span. EEE in use represents the currently used EEE equipage in the analyzed economy at time t. Stored obsolete or broken EEE is the amount of EEE kept in households or storage rooms because the consumer either believes that the EEE still has a certain value or because he doesn't want to face the hassle to properly dispose of it. Stored obsolete or broken EEE is either damaged and not functional anymore or perceived obsolete because of out-dated technology or changed consumer preferences.

Stocks and flows are interdependent and connected via functional relations. Both can be influenced or steered by auxiliary variables. The amount and structure of auxiliary variables is to be determined according to practical mathematical and consistency requirements and the modeling purpose. Numerous parameters could be operationalized and analyzed in terms of their impact on the EEE stocks and flows presented so far. However, the model is to be focused on the impacts of policy instruments on the end-of-life manager sphere. Accordingly, the modeling of EEE stocks and flows in the consumer sphere is based on simple, intuitively comprehensible variables and parameters. New EEE sales can prompt second hand EEE export to some extent. This effect was identified as a key driver for consumer-initiated second hand exports which is captured in the second hand export ratio $sher_{i,t}$. The storage ratio $sr_{i,t}$ determines the relation between the obsolescent EEE flow (ends up in stored obsolete or broken EEE) and directly disposed EEE, a flow introduced in the next subchapter. It is a parameter that represents the consumers' desire to keep EEE in households rather than disposing it directly. The average use span $aus_{i,t}$ serves to quantify the delay parameter connecting new EEE and obsolescent EEE. A longer use span augments the retention period of EEE in use. Figure 7 shows the auxiliaries and the connections used for defining functional relations and equations.

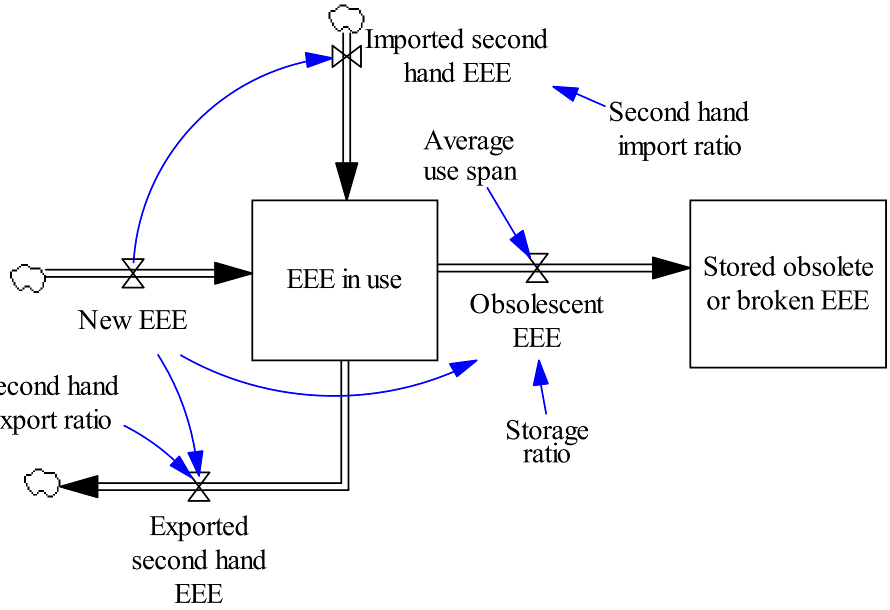

Figure 7: EEE stocks and flows and their auxiliary variables and parameters

2.2.3 EEE Characteristics in Practice

System data obtained from numerous literature sources is usually fragmented or inconsistent and therefore to be appreciated with prudence. Confidence in the figures presented here was developed using cross-checks, parallel sourcing of informations, and expert discussions. Some data can be referenced as such to literature sources, other stems from primary research or its combination with existing sources and the application of the abovementioned methods. Derived with the latter method, Table 2 shows data on the magnitude of EEE masses sold per year and million capita in a central European country (new EEE) (own analysis, partly based on EAK Austria 2006, SENS 2006, SWICO 2006).

Segment	New EEE Mass in 2006
Large (non-industrial) appliances [i=1]	**6634.58** [tons/year and m. capita]
Small (non-industrial) appliances [i=2]	**4893.61** [tons/year and m. capita]
Cooling and freezing appliances [i=3]	**1943.98** [tons/year and m. capita]
Display devices [i=4]	**2022.05** [tons/year and m. capita]
Lighting equipment [i=5]	**145.78** [tons/year and m. capita]

Table 2: New EEE masses sold per year and million capita in a central European country

The historical development of EEE masses sold per year differs in each segment. Changing average weights per unit complicate the deduction of new EEE masses from sold units. Figure

8 shows the history of unit sales of cooling and freezing appliances (segment i=3) per year and million capita in a central European country (Kronberger 2007, Pröll 2007). Data on unit weights is available from manufacturers and industry associations. Published data on average unit weights can be found for example in (SENS 2005, SWICO 2006, Sander et al. 2006).

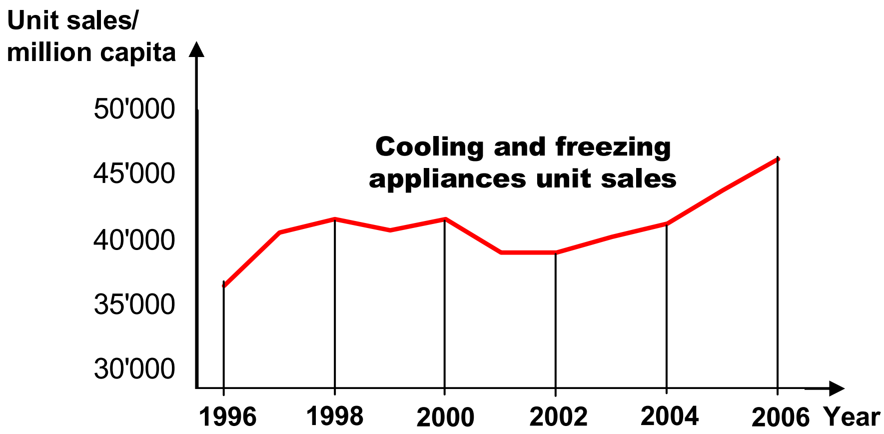

Figure 8: Historical development of unit sales for cooling and freezing appliances

The EEE stock in use as well as the respective characteristic use span differs considerably according to the analyzed economic region. Combining the household equipage with the average household size and the average weight of a type of EEE allows calculating EEE stocks in use. Table 3 shows a set of consistent data for selected appliances in a central European country (own analysis, partly based on Huber 2006, Destatis 2007, Brüning 2007, Sander et al. 2006).

Segment	Type of EEE	Equipage/ household [%]	Households/ million capita	Average weight [kg]	EEE in use/million capita [tons]	Average use span [years]
i=1	Dish washing machines	62.5	470889	43	12'655	11.5
i=1	Dryers	34.6	470889	35	5'702	13.6
i=3	Refrigerators	117.1	470889	42	23'159	14.4
i=3	Freezing appliances	83.0	470889	54	21'105	18.0
i=4	Televisions (CRT)	146.3	470889	20	13'778	11.3

Table 3: EEE stocks in use and average use spans in a central European country

Data on household equipage and size is usually provided by national statistical bureaus. Data on the average life span and use span can also be found in (SWICO 2006, EAR 2005, Lee et al. 2004).

2.3 Waste from Electrical and Electronic Equipment (WEEE)

2.3.1 Definitions and Terminology

"Waste electrical and electronic equipment or WEEE means electrical and electronic equipment which is waste within the meaning of Article 1(a) of Directive 75/442/EEC, including all components, sub-assemblies, and consumables which are part of the product at the time of discarding" (COM 2003).

This leads to the definition of waste as "any substance or object the holder discards, intends to discard or is required to discard" provided under the Waste Framework Directive (European Directive 2006/12/EC, COM 2006a) which repealed the European Directive 75/442/EC.

Furthermore, the following WEEE terms are relevant in this thesis and defined as follows: EEE reaches its end-of-life when the physical transition of a product from the consumer sphere into the sphere of an end-of-life manager occurs. At this point, it is either perceived obsolete and not worth storing, e.g. because of out-dated technology or changing consumer preferences, or damaged and not functional anymore. The transition from the consumer sphere to the end-of-life manager sphere turns obsolete or damaged EEE into WEEE which can be steered into several end-of-life fates. The consumer sphere is defined as the sphere where the consumer exerts major influence on the EEE flows. Likewise, the end-of-life manager sphere is characterized by the end-of-life managers' major influence on (W)EEE flows.

The classification of end-of-life fates for WEEE in this thesis is as follows: direct landfill or incineration, latent landfill via the household waste bin, illegal dumping, (illegal) export, material recycling, remanufacturing, repair, or reuse. More information on end-of-life fates and technologies can be found in (Huisman 2003).

2.3.2 Modeling WEEE Stocks and Flows

The consumer's choice about when and how to discard EEE is captured via several flows in the model. In terms of the timing, EEE can either be directly discarded after use (Disposed EEE I, $deI_{i,t}$) or discarded after storage (Disposed EEE II, $deII_{i,t}$). Both flows together add up to the overall disposed EEE in a year. They are dependant via time delay functions on new EEE and obsolescent EEE and influenced by the average use and storage spans. The storage ratio captures the relation between obsolescent but stored EEE and EEE directly disposed.

In terms of how to discard EEE, the consumer faces different options and already bears responsibility for the end-of-life fate of EEE to some extent. In principle, the consumer has the choice to dispose of a product via dedicated collection facilities (Dedicated collection, $dc_{i,t}$), to put it (illegally) into the household waste bin (Disposal via household waste, $hw_{i,t}$), or to dump it somewhere illegally (Illegal Dumping, $id_{i,t}$). Disposal behavior can be influenced by several aspects, among them consumer awareness, the cultural anchorage and acceptance of recycling in a society, the density and accessibility of collection points, and monetary incentives (deposits or return premiums) and disincentives (end-of-life disposal fees). Aside from the average use and storage spans $aus_{i,t}$ and $ass_{i,t}$, the effects of these parameters are captured in the illegal dumping ratio $idr_{i,t}$, the household bin ratio $hwr_{i,t}$, and the dedicated collection ratio $dcr_{i,t}$. Figure 9 shows the complete structure of stocks and flows in the consumer sphere.

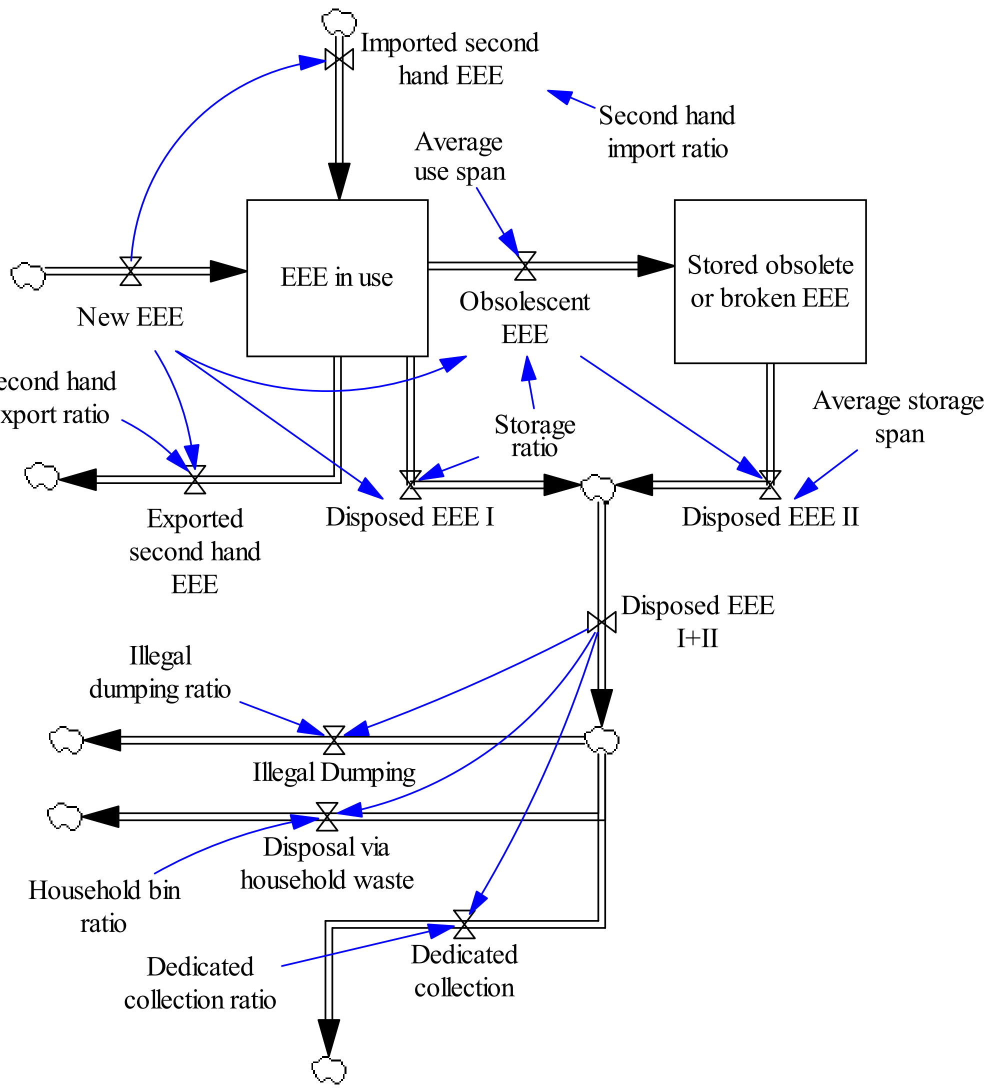

Figure 9: EEE and WEEE stocks and flows in the consumer sphere

If collected via dedicated facilities, WEEE enters the end-of-life manager sphere with the following options for further treatment: landfill, incineration, material recycling (Material recycling, $mr_{i,t}$), or remanufacturing, repair and reuse (Remanufacturing, repair, and reuse, $rr_{i,t}$). Furthermore, legal or illegal WEEE export is possible, the latter usually in conjunction with an unknown fate and treatment strategy of WEEE. Legal WEEE export with proven recycling and utilization performance is subsumed under material recycling in the model, illegal WEEE export (Illegal export, $ei_{i,t}$) is modeled as a separate stream due to its considerable magnitude. Landfill and incineration (Landfill and incineration, $la_{i,t}$) are modeled as one flow since they are both banned under modern recycling policies and both considered as environmentally inferior end-of-life treatment strategies. Each flow has a corresponding steering ratio ($mrr_{i,t}$, $rrr_{i,t}$, $eir_{i,t}$, $lar_{i,t}$) as indicated in Figure 10.

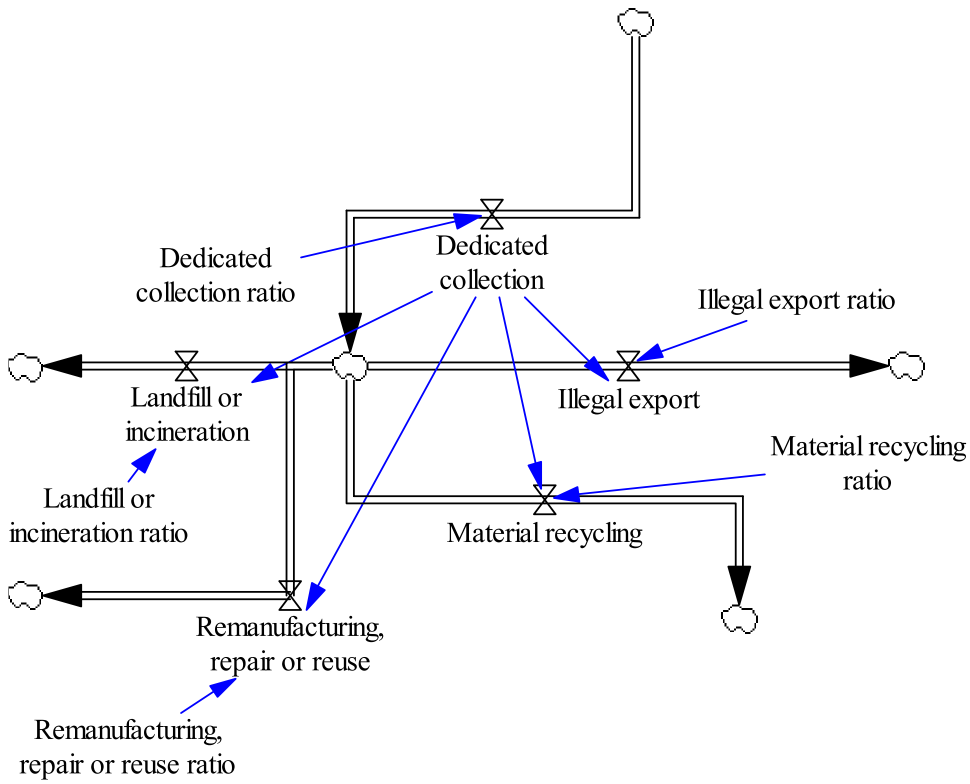

Figure 10: WEEE flows in the end-of-life manager sphere

2.3.3 WEEE Characteristics in Practice

Numerous estimates and studies have been undertaken in order to quantify “WEEE” in different analyzed economies (Griese et al. 1997, BVSE 1999, COM 2000, Ivisic 2001, BFUB 2001, Knoth et al. 2001, ETC/WMF 2002, Lee et al. 2004, OECD 2005, Magalini 2007a). However, different or inconsistent definitions cause issues in terms of reliability and comparability of the data. Table 4 provides an estimate of current annual WEEE amounts in a central European country, corresponding to disposed EEE in the model presented here (Disposed EEE I+II). The estimate is based on model simulations and has been discussed and cross-checked in expert panels.

Segment	Disposed EEE Mass 2006
Large (non-industrial) appliances [i=1]	**6080.00** [tons/year and m. capita]
Small (non-industrial) appliances [i=2]	**5947.53** [tons/year and m. capita]
Cooling and freezing appliances [i=3]	**1843.20** [tons/year and m. capita]
CRT devices [i=4]	**3172.40** [tons/year and m. capita]
Lighting equipment [i=5]	**146.67** [tons/year and m. capita]

Table 4: EEE disposed per year and million capita in a central European country

Few data is available on the EEE end-of-life fates influenced by the consumer. Illegal dumping plays an insignificant role in industrialized countries as the experience in Europe (Schwarzenbach 2006, Leitzinger 2006, Bornand 2006, Hediger 2006) and Japan (Tasaki et al. 2005) indicates. Table 5 shows study results from Japan that revealed low illegal dumping activity. The numbers for Europe should be of the same magnitude or even lower because WEEE is returned free of charge in Europe whereas consumers pay end-of-life fees in Japan.

Fate of discarded household appliances	TV sets	Refrigerators and Freezers	Washing machines	Air Conditioners
Illegaly-dumped (discovered only)	0.9%	0.4%	0.3%	0.2%

Table 5: Illegal dumping ratio of selected appliances in Japan (Tasaki et al. 2005)

Disposal via household waste plays, however, a significant role for small appliances. Due to the size, the share of EEE disposed via the household waste bin in other segments is insignificant. Data on the amounts of WEEE found in household waste is provided by (Waber 2001). Based on her research, the household bin ratio for segment i=2 in a central European country can roughly be estimated around 30%.

Dedicated collection must be distinguished into “officially tracked WEEE” and “inofficially and informally (but still separately) collected WEEE”. Officially tracked WEEE amounts are often published by producer responsibility organizations (e.g. Elkretsen 2006, Elretur 2006, SENS 2006, SWICO 2006, Recupel 2005). Inofficially collected WEEE especially plays a role in segments that include appliances with a high content of valuable materials such as the large household appliances. In such case, informally collected WEEE can outweigh officially tracked WEEE.

Illegal exports are more likely to arise for informally collected WEEE. In general, (illegal) exports play a significant role (Huijbregts 2007, Roman et al. 2002). No reliable data is officially available on these flows and published estimates on the (illegal) export ratio range from 25% up to 80% (Klatt 2001, BAN 2002, BAN 2005). Often, shipments are labeled as second hand EEE although they contain only damaged equipment (Zonneveld 2007). Study results from Japan covering the export ratio of four major appliances are presented in Table 6.

Fate of discarded household appliances	TV sets	Refrigerators and Freezers	Washing machines	Air Conditioners
Exported	52%	15%	17%	32%

Table 6: Export ratio of selected appliances in Japan (Tasaki et al. 2005)

Direct landfill and incineration have been prevailing among the end-of-life options for WEEE in the last decades (COM 2000). Data from the US Environmental Protection Agency suggests that around 91% of consumer electronic waste was still landfilled or incinerated in 2003 in the US (IAER 2003). However, both are banned under modern recycling laws and do not play a significant role in Europe or Japan nowadays.

Reuse, repair and remanufacturing of EEE are deemed first best end-of-life options according to current waste hierarchies (COM 2003, Williams et al. 2001, Lund 1996). However, LCAs reveal that this does not necessarily hold for several types of equipment (Skerlos et al. 2003, Legarth et al. 2003, Brüning 2007). Sometimes, a scenario with a new product in conjunction with the disposal of an old device performs better from an environmental point of view than a reuse or repair scenario. Since all life cycle stages are taken into account for life cycle assessments, a significantly lower energy consumption of a new product in the use phase can outweigh the environmental benefits of reuse. The potential of reuse and remanufacturing has been assessed by several researchers (Huisman 2003, Steinhilper et al. 2001, Kernbaum et al. 2007). However, reuse, repair and remanufacturing can only expand the use span of EEE – at some point, these appliances are going to occur as non-reusable waste which can only be recovered through material recycling. Material recycling currently represents the prevailing recycling process in developed countries and is comprehensively portrayed within the next chapter.

2.4 The Recycling Value Chain

This chapter presents activities, material, and monetary flows involved in take-back and material recycling. Furthermore, the economics of the recycling value chain are analyzed and put into a formalized structure, using activity-based costing as a method to allocate costs to products and services (Kaplan et al. 1999). Figure 11 shows the major steps of the recycling value chain that correspond to the constituents of the economic model.

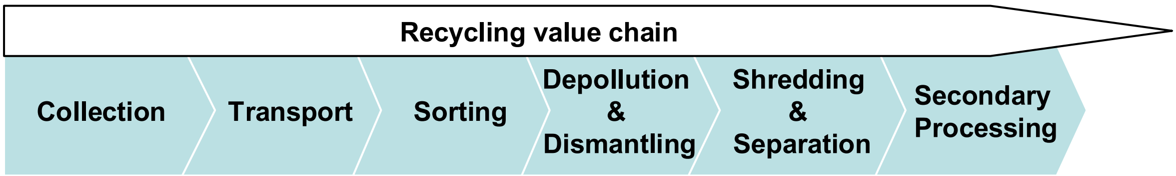

Figure 11: Recycling value chain

Each single step is subsequently portrayed, providing data and information on observed practice and the structure of the derived economic model. Some of the data presented has been generated during the research project with the support of different sources and research activities. The sections in this chapter only contain a quick explanation of the data background since providing complete information would disturb a smooth reading.

2.4.1 Collection

WEEE can be collected in several ways. Consumers usually have control and ownership of EEE at the end-of-life and their behavior and commitment is crucial for high dedicated collection rates. In principle, drop-off and pick-up collection can be distinguished. Pick-up is convenient for the consumer since it only requires providing WEEE in a suitable way. Drop-off, however, requires taking an obsolete product and bringing it to a collection point. A further distinction is to be made between continuous and temporary services. Temporary services require considerable advertising to raise awareness for an event in order to achieve reasonable participation levels. Continuous services do not need specific advertising for each event and consumer awareness increases the longer the services are established. Figure 12 classifies different collection options and highlights the ones that are mainly found in practice.

	Drop-off	Pick-up
Continuous	• **(Municipal) garbage collection stations** • **Take back in retail stores** • Collection points provided by social networks • Collection points at several businesses • **Direct take back at recycler's site**	• Curb side with other municipal solid waste • **Bulky waste curb side collection**
Temporary	• **Collection events** • Mobile collection vehicles ("Elektromobil") • Return through contractual agreement (B2B) • Direct return to OEM via mail	• On demand at households • On site disassembly of systems on demand (industrial EEE)

Figure 12: Collection options

An indicator for collection costs can be derived from PRO reports where collection costs are usually separately revealed (e.g. Elkretsen 2006, Elretur 2006, SENS 2006, SWICO 2006, Recupel 2005). However, these costs must be thoroughly analyzed in terms of their calculation basis and system-specific factors that may distort the picture. The costs of collection are usually characterized by a large share of fixed expenses and an insignificant amount of variable costs. Table 7 shows the cost structure for a municipal garbage collection station and corresponding data estimated for a central European country. It was assumed that the collection station is organized in an economic way. The cost structure is based on a generic cost framework similar to (Caudill et al. 2003) and the presented data was developed with the help of several experts and a telephone and email survey covering administrative bodies from 24 collection stations in Germany and Switzerland. Cost structures and corresponding data have been determined for all of the five highlighted collection options in Figure 12 and were used for the policy simulations presented in chapter 6.

Municipal garbage collection station (j=1)	
Staff cost (overhead)	
Amount employees	3.5
Working hours [hours/week]	37
Annual working hours employee [hours/employee and year]	1'776
Labor cost per hour [€/hour]	15
Allocation factor (average effort relation WEEE collected/total waste collected)	0.25
Total annual WEEE-related staff cost (overhead) [€]	**23'310**
Infrastructure cost (overhead)	
Office and social rooms [m2]	40
Annual rent offices and social rooms [€/m2]	92
Annual cost offices and social rooms [€]	**3'680**
Annual cost handling equipment (forklift,...) [€]	**5'000**
Annual cost infrastructure and maintenance (fences, locks,...) [€]	**4'000**
Other annual operating cost (office equipment, energy, water,...) [€]	**2'500**
Access routes [m2]	200
Switching space [m2]	150
Additional drop-off area [m2]	100
Annual space rental per m2 [€/m2]	7.5
Annual space cost [€]	**3'375**
Allocation factor (average effort relation WEEE collected/total waste collected)	0.25
Total annual WEEE-related infrastructure cost (overhead) [€]	**4'638.75**
WEEE-specific infrastructure cost	
Amount containers for circling [units]	7
Depreciation period [years]	10
Cost container [€/unit]	5'200
Annual container cost [€]	**3'640**
Space per container [m2/unit]	25
Annual space cost per m2 [€/m2]	7.5
Amount containers on site [units]	5
Annual space cost container area [€]	**937.5**
Total annual WEEE-specific infrastructure cost	**4'577.50**
Total WEEE collection cost per station [€]	**32'526.25**

Table 7: Collection costs for the archetype "municipal garbage collection station" (j=1)

In the economic model underlying simulations in this thesis, WEEE collection and its costs are captured with "collection archetypes" for the five main collection options. Therefore, the following parameters were defined: Each collection point type j (j=1,2,3,4,5) features fixed costs $fccp_j$ and variable costs $vccp_j$. Each class j is characterized by a collection point density/event frequency cpd_j and a class-related share sco_j of the WEEE take-back volumes (dedicated collection). These parameters allow calculating collection costs per ton $cc_{j,t}$ as follows:

$$cc_{j,t} = vccp_j + \frac{fccp_j}{\dfrac{\left(\sum_i dc_{i,t}\right) \cdot sco_j}{cpd_j}} \qquad (1)$$

According to (1), the collection costs per ton $cc_{j,t}$ are highly dependent on the collected volumes per collection point. These are captured with the combination of the collected WEEE amount (*dedicated collection,* $dc_{i,t}$) and the density of collection infrastructure (*collection point density,* cpd_j) . Figure 13 shows the relation of the collection costs per ton and the collected volumes per collection point for class j=1 (municipal garbage collection station) in a central European country. The average collection costs in an analyzed economy (*average collection costs per ton,* acc_t) are calculated as follows:

$$acc_t = \sum_j sco_j \cdot cc_{j,t} \quad (2)$$

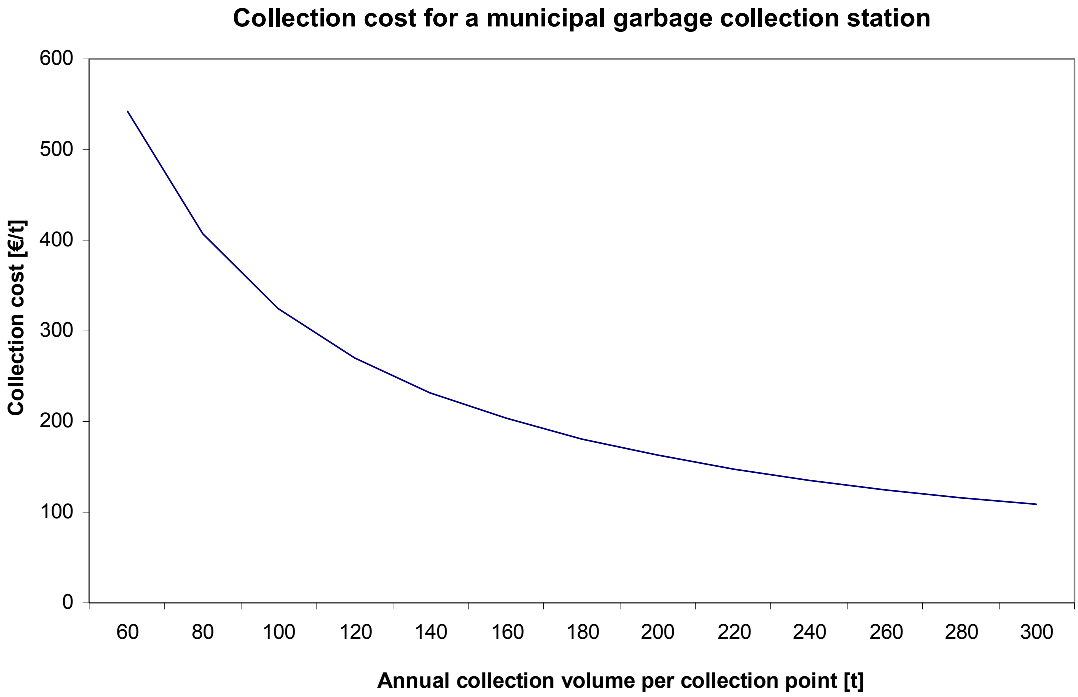

Figure 13: Relation of collection costs and annual collection volume

2.4.2 Transport

Models for reverse logistics networks in the WEEE context have been analyzed in (Krikke 1998, Spengler 1998, Fleischmann 2000, Walther 2005). WEEE that passes through the recycling value chain needs to be transported several times. Usually, consumers transport WEEE to a collection point where it is accumulated. The accumulated WEEE is then further transported to a treatment facility. After separation, materials are transported to secondary processors until they enter their final reapplication technology.

For the purposes of this thesis, a reasonable estimate of average transport costs per ton atc_i from a collection point to a recycling facility is to be developed. The estimate should be able to reflect variations in both the collection point density cpd_j and the amount of recycling facilities RF_i in order to allow for sensitivity analysis in simulations. The latter is also reflected in the recycling facility density rfd_i which is defined as the amount of facilities per km^2 and measured in [units/km^2]. Costs for WEEE drop-off which are borne by the consumer were not included in the economic model and transportation costs of secondary materials were reflected in material fraction prices that include material pick-up at the recycler's site.

A first constituent of the average transport costs is transportation equipment used. Numerous types of storage and transportation equipment can be used for WEEE materials, among them pallet boxes, gaylords, skeleton transport boxes, and various steel containers of different sizes and styles. Standard equipment often found in central Europe is the 38m^3 roll-off steel container that can be used both on the road and on the railway. Dependent on the lorry type, one or several of these containers can be carried. The amount of WEEE transported per such container depends on the segment and can be captured with segment-specific load factors. Table 8 shows average load factors lf_i for 38m^3 roll-off steel containers for the different segments that were calculated as weighted average from a small sample of data provided by recyclers (own analysis).

Segment	Load factor [t/container]
Large (non-industrial) appliances (i=1)	5.2
Small (non-industrial) appliances (i=2)	9.5
Cooling and freezing appliances (i=3)	2.9
CRT devices (i=4)	6.5
Lighting equipment (i=5)	*

Table 8: Segment-specific load factors

Further, transport costs are mainly dependent on the transport distance; however, average times for loading and unloading, average waiting times, driver labor cost, fuel cost, road toll, and the type of truck must be taken into account as well. Overheads like maintenance, insurances, taxes, truck depreciation and financing costs, order management, and tour organization and planning affect the cost calculation, too. Figure 14 shows transport costs and the respective transport cost factor tcf as a function of the transport distance for a suitable truck in a central European country. Cost structure and data underlying Figure 14 have been determined with the help of several experts (Wainer 2006, Smekal 2006).

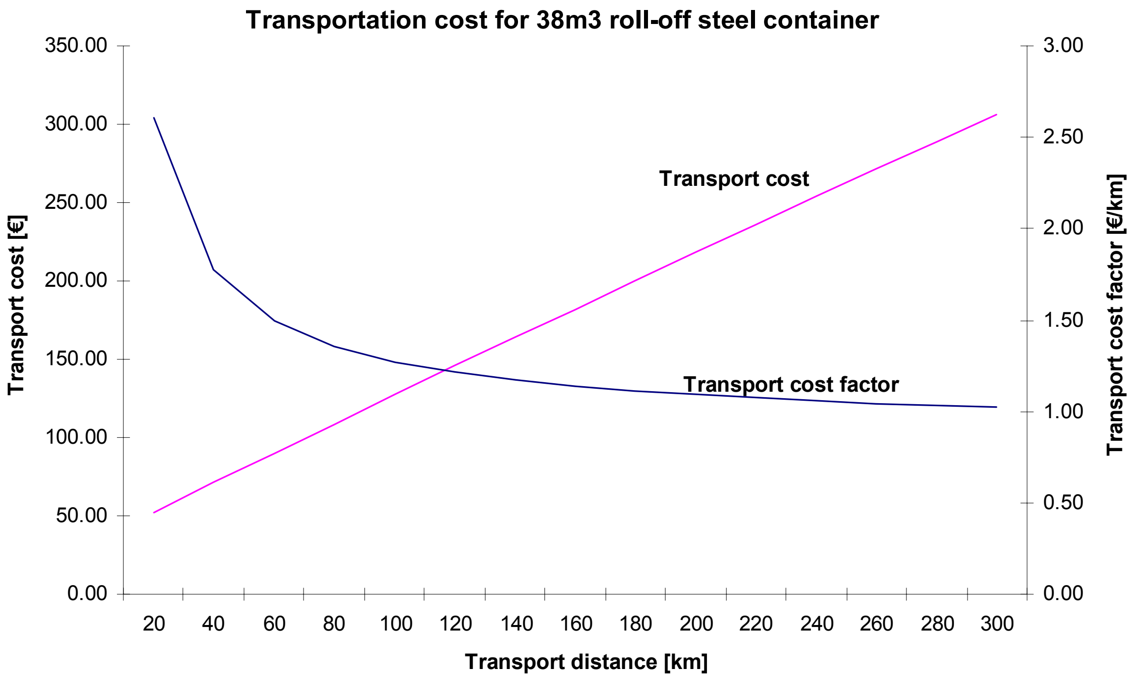

Figure 14: Transportation costs for $38m^3$ roll-off steel container

In order to estimate the average transport distance between collection points and recycling facilities in an analyzed economy, the following simplifying assumptions were made:

- The recycling facility density in a region is highly correlated with the population density and the WEEE density in the region.
- The service area worked by each recycling facility has the form of a circle.
- WEEE is homogenously distributed in the service area.
- The relation of the area only serviced by one facility to the area serviced by several facilities is 2 to 1.
- The relation of the actual transport distance and the beeline distance can be captured with a constant detour factor v (Gudehus 1999).

With these assumptions, a reasonable estimate of the average transport distance atd_i is:

$$atd_i = v \cdot \frac{2}{3}\sqrt{\frac{1.5}{\pi \cdot rfd_i}} \qquad (3)$$

The possibility of potential empty runs on the way back from the recycling facility must be taken into account as a last step for the calculation of the average transport costs per ton atc_i. The resulting average transport distance per tour $atdt_i$ is calculated with the utilization factor uf which equals 1 if no empty runs occur and 2 if one run is completely empty. Altogether, the average transport costs per ton in each segment i can be determined as follows:

$$atc_i = \frac{atdt_i \cdot tcf(uf, atd_i)}{lf_i} = \frac{uf \cdot atd_i \cdot tcf(uf, atd_i)}{lf_i} \qquad (4)$$

2.4.3 Sorting

In principle, sorting of WEEE is possible on several levels of the value chain. Sorting already occurs at collection sites where some sort of “value screening” for reusable or resaleable equipment is not uncommon. If not completely harvested there, reusable equipment or valuable components are separated by social enterprises and companies that focus on manual dismantling. These companies usually derive a considerable share of their revenues from either reuse of complete equipment or components with substantial value. Non-industrial consumer equipment, however, tends to be of high age and does not have the same reuse potential than some of the B2B appliances for which special component harvesters exist (Dickenson 2007). Actual WEEE with non-significant reuse potential is sorted in order to prepare it for material recycling and to maximize the (material) value recovery. As a first step, this involves sorting WEEE according to generic practical treatment streams for material recovery (i=1 to i=5). Furthermore, recyclers apply internal sorting strategies in order to generate batches for a processing in their plant. Some equipment (e.g. IT equipment) tends to have higher precious metals content and is therefore separately processed, partly with a different setup in the shredding and material sorting stage.

Sorting is also required for accounting and reporting purposes. Some customers ask for evidence of destruction of specific equipment, some want to be informed about the bill of materials produced from their equipment, or legal compliance can require reporting material flows or recycling rates calculated on their basis. A distinction of waste streams managed by different authorities can also be required to allow for separate billing of different institutions or customers.

Sorting times and costs depend on the respective set h of process requirements spr_h for the sorting tasks. (Walther 2005) provides data of sorting times and costs for different EEE and distinguishes identification, registration and handling in the sorting process of a manual dismantling company. Sorting costs are calculated with average sorting process times per appliance, an average weight of the appliance and an hourly labor cost rate. For the purposes of this thesis, sorting times and costs are modeled similarly, using the labor cost rate lr and average sorting coefficients ($scc_{i,h}$, [t/hour]) per segment that are differentiated according to the actual sorting requirements. No sorting costs are assumed for the generic sorting performed by the consumer on a collection site. The default setting for a recycler's sorting activities is driven by a focus on material recycling and profit maximization. Average sorting costs per ton $asc_{i,h}$ are accordingly calculated as follows:

$$asc_{i,h} = \frac{lr}{scc_{i,h}(spr_h)} \quad (5)$$

2.4.4 Depollution & Dismantling

Approximately half of all discarded EEE in Western Europe is shredded and subsequently separated into several material streams without any dismantling (van Rossem 2003). Dismantling is usually performed for (Boks 2002)

- a sound recovery of hazardous components or substances (depollution): the removal of hazardous components is usually mandated and specified by environmental authorities and a legal framework.
- the recovery of valuable components. Examples for valuable components are video cards, electric motors or PC components. Printed circuit boards are often separated due to their high precious metal contents.
- the removal of unwanted parts. Sometimes, the presence of unwanted components substantially compromises the processing speed or the concentration of targeted metals in a fraction. Large plastics housings are a typical example.

Figure 15 exemplifies the material flows in the dismantling and depollution stage.

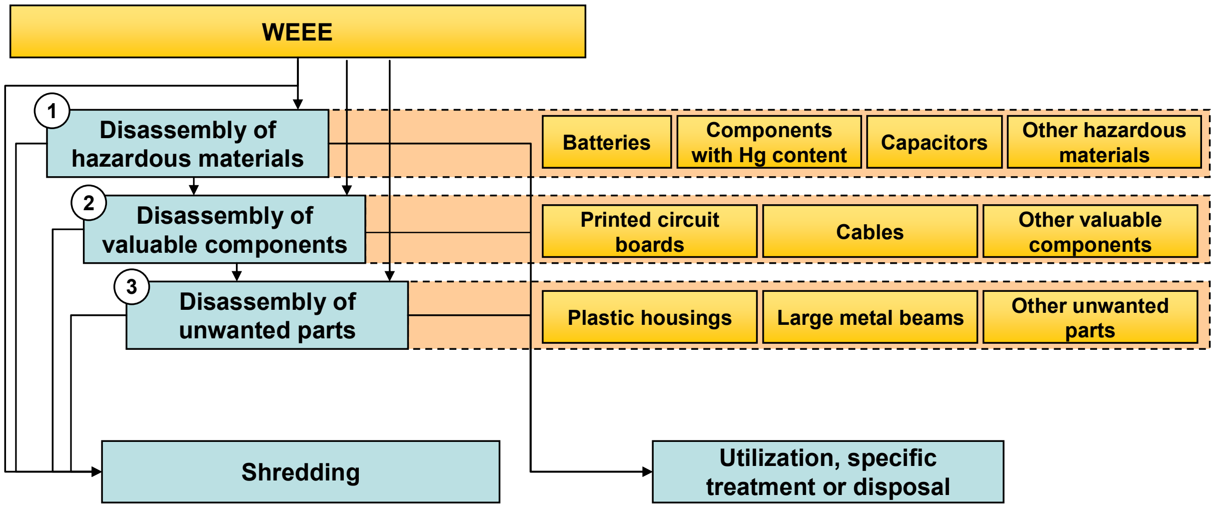

Figure 15: Material flows in the dismantling and depollution stage

(Supported) manual dismantling operations are still prevailing in industry practice. Fully or semi-automated dismantling is still a big challenge for the recycling industry although several systems have been realized as pilot and testing facilities (Seliger et al. 2000, Sony 1996, Knoth et al. 2002, Stobbe et al. 2002). Such facilities struggle with the wide variety of EEE designs that turns the application of standardized methods into a complicated venture.

The amount of time spent for dismantling and depollution and the respective costs depend on the accuracy of the screening for valuable, unwanted or hazardous components. The first two are usually performed with high accuracy due to economic incentives. However, economic disincentives exist for a sound and comprehensive depollution due to the considerably big efforts for the achievement of high depollution levels. Within this thesis, the achievement of such levels is measured with the depollution quality level $dql_{i,k}$, defined as quotient of the mass $hmr_{i,k}$ of hazardous materials or components k actually recovered and the expected maximal mass of recoverable hazardous materials or components $ehmr_{i,k}$ in a segment.

$$dql_{i,k} = \frac{hmr_{i,k}}{ehmr_{i,k}} \qquad (6)$$

The values for $ehmr_{i,k}$ in each segment vary considerably with the composition of WEEE inputs and no reliable data is available since no recycler can ensure 100% recovery. Table 9 shows an estimate (mass shares of pollutants relative to input mass) for depollution results in segments i=1,2,3 for a central European country that serves as an indication of the magnitude of potential hazardous material flows (own analysis, partly based on Kasser 2006, Hug 2006, Franov 2006, SENS 2005, Fraunhofer 2006, Sander et al. 2006).

Large appliances (i=1)	
Fraction description	**Share**
Mercury components	0.0020%
(PCB suspect) capacitors	0.636%
Electrolyte capacitors	0.240%
(PCB suspect) heat transmission oils	0.010%
Heat transmission oils	0.010%
Appliances containing (free) asbestos	1.000%
Components containing asbestos	0.100%
LCDs > 100cm2	0.010%
Printed circuit boards	0.200%
Small appliances (i=2)	
Fraction description	**Share**
(PCB suspect) capacitors	0.160%
Appliances containing (free) asbestos	0.0360%
Mercury components	0.080%
Batteries	2.000%
Other hazardous waste	0.100%
Cooling and freezing appliances (i=3)	
Fraction description	**Share**
Condensation water	0.330%
Mercury components	0.01050%
PCB suspect capacitors	0.0010%
R11+R12 (step 2)	0.546%
R12, R22, R134a, R502 (step 1)	0.228%

Table 9: Estimated maximal depollution results per segment in a central European country

The effort put into increasing depollution levels is not a linear function of the depollution quality level. Some hazardous components are easy to identify and disassemble whereas others are rarely discovered or cause tremendous difficulties during their separation. "Harvesting" the latter ones requires spending considerable time on screening WEEE and respective expertise of the dismantler. In the research work performed, depollution costs were either modeled per fraction or per segment, depending on how depollution tasks are tackled in practice and data accessibility. In principle, both fraction and segment-specific depollution cost functions were modeled in the form of exponential functions. Segment-specific depollution cost functions often involve the average segment-specific dismantling time adt_i which is dependent on the depollution quality level. In the case of fraction-specific-depollution functions, the modeling of average depollution costs adc_i requires to sum up depollution costs from all fractions. Equations (7) and (8) illustrate the calculation of depollution costs.

$$adc_{i,(k)}(dql_{i,(k)}) = adt_{i,(k)}(dql_{i,(k)}) \cdot lr = a_{i,(k)} \cdot b_{i,(k)}{}^{c_{i,(k)} \cdot dql_{i,(k)}} \cdot lr \qquad (7)$$

$$adc_i(dql_i) = \sum_k adc_{i,k}(dql_{i,k}) \qquad (8)$$

Disassembly times have been analyzed or reported in (Brouwers 1995, Huisman 2003, Walther 2005, Herrmann 2003, Kühn 2000). Figure 16 shows an example of an average dismantling time function for small non-industrial appliances, based on equation (7) and gauged with expert estimates.

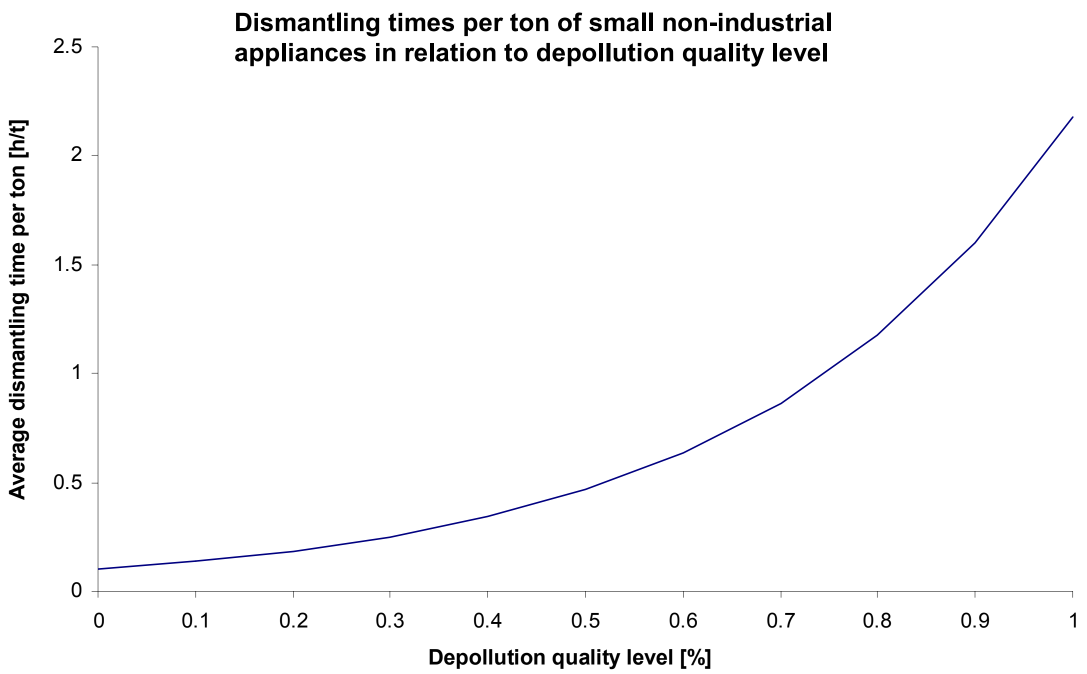

Figure 16: The relation of average dismantling time and depollution quality level

Using equation (7) yields variable costs of depollution. However, these activities are performed within a recycling facility where overheads and other fixed costs occur. The magnitude and modeling of these costs are presented in the next section in conjunction with the introduction of WEEE plant archetypes.

2.4.5 Shredding & Separation

WEEE is a complex composite of mainly metals and plastics. After depollution and dismantling of components, WEEE is shredded into small pieces which are separated afterwards. Most recycling facilities perform the shredding with coarse shredders (car shredders, large hammer mill), smaller-sized shredders with one, two or four shafts, or rotating chain mills. Slowly-rotating mills are found among older facilities.

Separation technologies are roughly based on particle-specific or material-specific properties of the shredded WEEE (Walther 2005). Modern separation technologies often combine both approaches. Examples for particle-specific sorting techniques are sieving or air separation. Traditional examples for material-specific sorting techniques are magnetic separators, eddy current separators, or flotation devices. Optical sorting gains more and more attention. Process descriptions for shredding and separation are published in (Walther 2005, Magalini 2007b, Huisman 2003, Morf et al. 2004)

In this thesis, shredding and separation in a material recycling facility is modeled with defined WEEE plant archetypes. The archetypes represent state-of-the-art turn-key facilities with basic equipment and lean management structures; allowing an analysis of costs and revenues derived from each WEEE segment.

The main cost and revenue categories associated with an operation of the WEEE recycling archetype plant are depreciation and amortisation from investments $daic_{i,t}$, financing costs $fic_{i,t}$, other fixed costs $ofc_{i,t}$, variable costs from operations vco_i, and costs for material disposal mdc_i. Revenue sources are recycling service remunerations $rsr_{i,t}$ and secondary material revenues smr_i. The magnitude of each category depends on several strategic decisions, e.g the financing strategy (buy or rent), or the treatment depth and corresponding size of the machinery park. Table 10 displays investment, depreciation and financing costs related to the operation of a basic archetype plant located in a central European country. The plant can potentially process WEEE from segments i=1, 2, 4.

Basic facility		
Machinery investment	€	4'500'000
Planning cost and licensing of operations	€	40'000
Plant assembly	€	50'000
Depreciation period	years	8
Annual depreciation facility with linear method	€	**573'750**
Real estate		
Developed real estate (Facility, Office, Storage space)	m^2	15'000
Real estate rental fee per year and m^2	€	36
Annual cost real estate	€	**540'000**
Office buildings and workshop		
Investment office buildings and social rooms	€	200'000
Investment workshop	€	1'000'000
Depreciation period	years	15
Annual depreciation office buildings and workshop	€	**80'000**
Working equipment		
Forklifts, cranes and other handling equipment (boxes, containers,…)	€	250'000
EDV equipment	€	30'000
Other equipment	€	20'000
Depreciation period	years	3
Annual depreciation working equipment		**100'000**
Total plant investment	**€**	**6'090'000**
Investment cost		
WACC		18%
Financing cost	€	**1'096'200**
Plant-based annual investment and financing cost	**€**	**2'389'950**

Table 10: Investment and financing costs of an archetype plant

Table 11 allows developing an idea of the magnitude of other fixed costs and aggregated variable costs from operations. These costs are again influenced by treatment depths and the efficiency of internal material flows.

Staff administration		
1 plant manager (business)	€	100'000
1 plant manager (technical operation)	€	70'000
1 employee quality assurance	€	42'000
1 employee mechanic	€	35'000
1 office assistant	€	30'000
Staff cost administration	€	**277'000**
Maintenance and insurance		
Total cost insurance	€	129'608
Total cost non-operational maintenance	€	204'000
Total maintenance and insurance cost	€	**333'608**
Total other fixed costs	**€**	**610'608**
Variable staff costs	€/t	34.01
Variable other operational costs	€/t	3.15
Total variable operational costs	**€/t**	**37.17**

Table 11: Other fixed costs and variable operational costs of an archetype plant

A key driver for the economics of a WEEE recycling facility is its capacity utilization fcu_i. WEEE in segments i= 3, 5 is often treated in separate plants whereas WEEE from segments i=1, 2, 4 can in principle be processed with the same equipment. In practice, big amounts of large household appliances are still treated in car shredder whereas WEEE from segments i=2, 4 is found in specialized WEEE plants. In the system model, an average capacity utilization acu is defined according to (8), based on the treatment capacity of the archetype plant fcy_i and the amount of WEEE available per facility for material recycling in the covered segments.

$$acu_{i,t} = \frac{\dfrac{\sum_i mr_{i,t}}{RF_i}}{fcy_i} \qquad (8)$$

The facility capacity can be calculated with basic performance parameters according to equation (9). It depends on the shredding equipment capacity, the amount and length of shifts per work day, work days in a year, the availability of the plant, and the relation of input materials from each segment. Table 12 shows the calculation of a facility capacity based on basic performance parameters of an archetype plant able to process WEEE from categories i=1, 2, 4.

Basic performance parameters		
Shredding equipment capacity sec	[tons/hour]	8.50
Percentage shredder input sii	[%]	80
Total input material	[tons/hour]	0.10625
Working days per year wd	[days]	248
Working hours per shift hs	[hours]	8
Shredding shifts per day ssd	[units/day]	3
Working hours per year	[hours/year]	5'952
Facility availability fa (breakdowns, service,….)	[%]	90
Effective working time	[hours/year]	535'680
Facility capacity per year fcy	[tons]	56'916

Table 12: Basic performance parameters of an archetype plant

$$fcy = fa \cdot ssd \cdot hs \cdot wd \cdot \frac{\sec}{sii} \qquad (9)$$

2.4.6 Secondary Processing

Secondary processing and reapplication options of materials after shredding and first separation are manifold. A comprehensive overview of secondary processes and the destination of fractions can be drawn from the "reptool", a monitoring instrument developed by the WEEE forum and introduced in practice in several European countries (Gabriel 2006). In order to provide a generic overview, the following typical fractions can be distinguished:

Ferrous metal fractions are used in iron smelters. Actual steel scrap can be used in the basic oxygen furnace (takes up to 30% scrap) or the electric arc furnace (takes up to 100% scrap).

However, the electric arc furnace is sensitive to the presence of residual elements like zinc, copper, chromium and molybdenum which can cause defects in the finished steel at levels measured in tens of parts per million (Wernick et al. 1998). Some steel mills use their own system of quality-defined fractions (Fuchs 2006, Rytz 2006), others rely on classifications such as, for example, the European steel scrap list.

The final destinations of most **non-ferrous metals** are copper smelters, aluminum smelters, zinc smelters, and lead smelters. Before entering this final technology, non-ferrous metals have previously undertaken further concentration processes. **Aluminum** is usually further concentrated through flotation processes (Huisman 2003), cables are stripped to recover the contained **copper**, or specialized recyclers apply optical sorting technologies to separate different types of **heavy metals**. Dependent on the purity, such fractions can also directly be processed in a reverberatory furnace or used in aluminum foundries. Furthermore, several refinery steps in modern copper smelters separate copper and other **precious metals** that are concentrated on printed circuit boards (BOL 2005).

Glass, mostly from CRTs, can be utilized in several ways. The prevailing options are direct glass-to-glass recycling and its utilization in lead smelters because of its considerable lead content. In addition, glass can be used as a replacement for feldspar in the ceramic industry or as replacement for sand in the building industry. A comprehensive description of secondary recycling options for CRT glass is provided by (Bipro 2006).

Plastics can be recycled on a material level, used in gasification plants/ for methanol production, or incinerated with energy recovery. Recycling on a material level requires the separation of plastics that contain brominated flame retardants and a sorting of different types. Automated processes for plastics recycling are only mastered by few companies; however, current economic conditions encourage the spread of technologies to recycle plastics on a material level.

Hazardous materials that are separated in the depollution stage are recycled, incinerated, or disposed of in special landfills. For example, batteries are recycled, mercury from mercury-containing components is recovered, CFCs and asbestos are incinerated, and PCB suspect capacitors are disposed of in special landfills.

The costs for material disposal mdc_i and revenues from secondary materials smr_i for a primary recycler depend on the produced output per ton input in each segment. The average material disposal costs per ton $amdc_i$ and the average revenues from secondary materials

$asmr_i$ in each segment i are calculated with the output mass share of each fraction k $fms_{i,k}$ and an average disposal price/ secondary material price of each fraction fp_k.

$$amdc_i = \sum_k fms_{i,k} \cdot fp_k \qquad (10)$$

$$asmr_i = \sum_k fms_{i,k} \cdot fp_k \qquad (11)$$

The amount of materials to be recovered from different WEEE segments differs considerably due to differing input characteristics and different processing strategies. Table 13 shows an estimate of expected outputs of an archetype material recycling facility for WEEE segments i=1,2 and 3 in a central European country (own analysis, partly based on Kasser 2006, Hug 2006, Franov 2006, SENS 2005, Fraunhofer 2006).

Output per ton input in segment i=1	
Fraction description	**Share**
Ferrous metals	62.100%
Non-ferrous metals	2.100%
Cables	0.484%
Motors (with copper spools)	3.106%
Heavy metals	3.400%
(Quality-defined) plastics	0.000%
Stainless steel	4.600%
Sieving material	8.500%
Shredder light fraction	6.480%
Residual waste	8.552%
Output per ton input in segment i=2	
Fraction description	**Share**
Residual waste	8.000%
Shredder light fraction	6.350%
Printed circuit boards	1.000%
Cables	1.800%
Ferrous metals	29.780%
Non-ferrous metals	3.845%
Plastics	32.500%
Transformers, motors, other metal-containing parts	15.050%
Stainless steel	1.250%
Output per ton input in segment i=3	
Fraction description	**Share**
Ferrous metals (>99% pure)	47.000%
Aluminum/copper	5.400%
Compressor	21.320%
Cables and wires	0.380%
Oil	0.480%
Plastics (approx. 80 % polystyrene)	12.459%
PUR foam (insulation)	9.760%
Non recyclable items (e.g. food residues)	1.560%
Glass wool and "non-PUR"	0.170%

Table 13: Estimated average output per ton input in a central European country

Recycling service remunerations $rsr_{i,t}$ depend on the contracting body paying for the recycling service. For the purposes of this thesis, average recycling service remunerations $arsr_{i,t}$ can be derived as a residual from all other defined variables, using industry-specific weighted average costs of capital $WACC$. The exact design of the equation determining the $arsr_{i,t}$ depends on the financial responsibilities of different players on the value chain and the policy design. Examples under different policies will be analyzed and illustrated in the case scenarios in chapter 6. This, however, requires understanding different policy instruments which will be described and analyzed in the following chapters.

3 Regulation and Extended Producer Responsibility

This chapter illustrates the need for regulating electronics recycling and the aspects regulation should focus on. Furthermore, it explains the background and features of extended producer responsibility and provides insight into its application fields and function. An overview of the development of WEEE recycling regulation across the globe is provided afterwards, followed by a composition of goals relevant for the analysis and evaluation of EPR approaches.

3.1 The Need for Regulatory Action

Achieving sustainable development has been acknowledged as one of the fundamental challenges to be tackled in the 21st century. Early works in the area of industrial ecology suggest that a cycle economy in the electronics industry is a crucial contribution to sustainable development (Costanza et al. 1997, Allenby et al. 1994). Obsolete electronic products contribute to a growing volume of toxic inputs into the local waste streams and the manufacturing of new products entails a large demand for natural resources. Accordingly, the closing of material loops and the appropriate treatment of hazardous materials are two key aspects to be addressed when assessing the need for regulations. An analysis of existing economic or market incentives is required in order to come to a conclusion where regulation is necessary and where market forces already achieve favorable results (Costanza et al. 1997).

The following observations allow developing an understanding of the main relevant issues in the context of WEEE recycling[3]:

- Product design plays a significant role for the closing of material loops (Herrmann 2003). Design strongly impacts the recyclability of a product's materials or the reusability of some of its components (end-of-life perspective). Further, product design also accounts for possible secondary material content in new products (beginning-of-life perspective). Producers do not face economic incentives to improve product design in this regard since no economic advantages arise from environmentally favorable product design.
- In developed countries with high labor costs and established worker health and safety standards, recycling of WEEE including its collection, transportation, and treatment

[3] A more comprehensive description of these issues can be found in (Bohr 2005), embedded in an industrial ecology framework for sustainable manufacturing.

requires additional financing to be economically viable. This is not the case for third world countries with no established treatment and disposal standards. Due to the economics, WEEE export to such countries takes place to a tremendous extent, resulting in the release of toxics into ecosystems and the exposure of workers to dangerous substances without flanking protective measures (BAN 2002, BAN 2005).

- A further key concern in the recycling process of WEEE is a sound depollution. Recyclers do not face economic incentives for sound depollution and appropriate disposal of the toxics. In contrary, strong economic disincentives to do so exist.
- Markets for secondary materials are established for metals; however, still underdeveloped for other secondary materials from WEEE. This compromises the achievement of high recycling rates and recycling quotes which determine the performance in the closing of material loops.

Several million tons of WEEE every year and growth rates around 11-13% per year call for immediate action. The concept of extended producer responsibility has widely been acknowledged as an appropriate answer to the abovementioned issues (Lee et al. 2004, Lindhqvist et al. 2003, Mayers et al. 2005, Roine et al. 2006). EPR's features and background are described in the next section.

3.2 Extended Producer Responsibility (EPR)

3.2.1 Definition, Aim, and Application Fields

Extended Producer Responsibility (EPR) is a relatively new and market-oriented regulatory instrument. The Organization for Economic Cooperation and Development defines it as "an environmental policy approach in which a producer's responsibility, physical and/or financial, for a product is extended to the post-consumer stage of a product's life cycle" (OECD 2001).

An even broader interpretation comes from (Lindhqvist et al. 1990) who claim, that EPR is "an environmental protection strategy to reach an environmental objective of a decreased total environmental impact from a product, by making the manufacturer of the product responsible for the entire life-cycle of the product and especially for the take-back, recycling and final disposal of the product."

EPR's conceptual goals are characterized by a shift from end-of-pipe aproaches to preventive measures, an enhancement of life-cycle thinking, and the attempt to set incentive mechanisms for industry to improve products and processes (Tojo 2003). A review of the guiding principles of EPR programs reveals two key features: First, responsibility for and financial burden connected to the treatment of products after their actual use phase shall be shifted toward the producers. This forces manufacturers to reflect a product's end-of-life management costs in their business models. Producers are likely to pass these costs on to consumers via the product price. Second, EPR emphasizes the creation of a product design feedback loop to manufacturers. This covers all three conceptual goals and shows the feedback loop's central role in the EPR concept. The feedback is supposed to spur improvements in green design or, more specific in the context of the WEEE directive, design for end-of-life. In the broader, entire life-cycle-oriented EPR interpretation, the feedback is meant to set incentives for producers to generally incorporate environmental considerations into the design of products, including the use phase. This eventually leads to fewer overall environmental impacts of these products. Finally, guiding EPR principles further advocate for "focusing more on the results than on the means of achieving them" and "implementing policies that avoid economic dislocations".

The EPR concept can be applied to almost any product. So far, typical product groups covered by EPR programs are cars, packaging materials, paper, tyres, solvents, batteries, and electrical and electronic equipment. An EPR approach is further characterized by applied policy instruments and its organizational form. Both are briefly described within a dedicated section.

3.2.2 Policy Instruments

Various policy instruments are applicable under the EPR umbrella that can be classified according to types and focal point on a product's life cycle. With respect to the latter, it is possible to distinguish instruments focused on green production, a product's green performance characteristics during its use span, and instruments targeted toward end-of-life aspects. This is usually also reflected in separate legislatory approaches. In the case of electrical and electronic equipment (EEE), the RoHS stipulations concern green production (ROHS 2003), the EuP directive focuses on a product's use phase (COM 2005), and the WEEE directive targets end-of-life issues (COM 2003). Nevertheless, an integrated approach is often necessary to ensure that the set of mandated rules creates holistically favorable incentive structures. (Walls 2006) further distinguishes administrative, economic, and informative instruments. Table 14 provides an overview with examples of EPR-based policy instruments.

	Green Production	Use span	End-of-life
Administrative	• Targets for recycled content • Product standards	• Utilization mandates	• Landfill restrictions • Collection and recycling targets • Environmentally sound treatment standards
Economic	• Material/product taxes • Subsidies • Tax incentives	• Material/ product taxes • Subsidies • Tax incentives	• Material/ product taxes • Advance disposal fees • Deposit-refund systems • Tradable recycling credits
Informative	• Marking/labeling of products • Information requirements towards consumers	• Marking/labeling of products • Information requirements towards consumers	• Marking/labeling of products • Reporting to authorities

Table 14: EPR-based policy instruments

When applying or introducing such instruments, it is important to assess the economic as well as the environmental importance of the regulated field. The relative environmental importance of different life cycle stages is hard to measure although attempts to do so exist. Figure 17 presents an attempt to measure the relative importance of different life cycle stages of an average consumer electronic product, based on the ecoindicator 99 methodology (Goedkoop 2001)

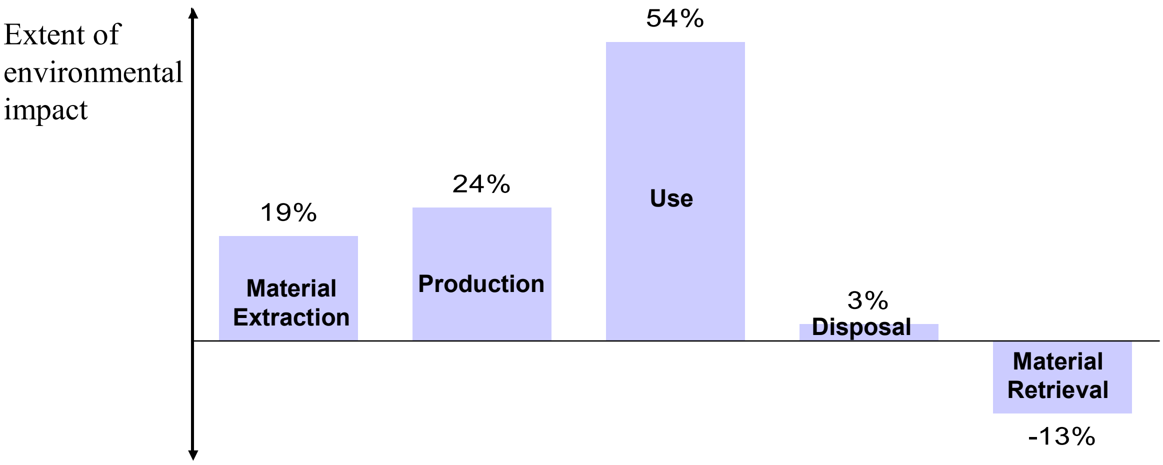

Figure 17: Environmental impacts of EEE from life cycle stages (modified, Huisman 2003)

The measurement is based on several assumptions and can only serve as a coarse indication. In the graph, the environmental burdens of the four first life-cycle stages sum up to 100% and improvements through material retrieval or recovery are measured as a negative number that is computed in relation to the sum of the first four stages. According to these calculations, the importance of end-of-life management is relatively humble in terms of environmental impacts. However, the eco-efficiency[4] of measures mandated via regulation in different life-cycle stages should first be taken into account. The total impact of a life-cycle stage is therefore not the best indicator for the focus of environmental policy. A well thought-out recycling regulation might be able to realize highly eco-efficient decreases in the environmental impact of EEE, dependant on the economic context.

3.2.3 Organizational Forms

End-of-life management responsibilities are in the focus of this thesis, requiring the definition of two ranges with respect to organizational forms of EPR: Individual vs. collective responsibility and competitive vs. collaborative approaches to fulfill EPR obligations.

Individual producer responsibility (IPR) represents the original conceptual idea of EPR. Here, a direct connection between producers and their individual waste is established. The IPR can start on different levels of the value chain, dependant on the degree of individual responsibility. In its extreme form, producers are financially and physically responsible for the individual collection, transport, and sound treatment of their own wastes. The reference point on the value chain where an identification and allocation of individual wastes takes place is an indicator for the degree of individual producer responsibility.

No identification and allocation of individual wastes at all characterizes the collective producer responsibility (CPR). Under CPR, producers completely loose the connection to their wastes. CPR allows producers to build groups or service units and manage WEEE collectively. This creates economies of scale on several value chain levels but means that waste remains “anonymous” and producers can not face product design incentives in a direct form.

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[4] For attempts to define eco-efficiency, see (Huppes et al. 2005, Kuosmanen 2005, Ekins 2005, Brattebo 2005, Ehrenfeld 2005). In this thesis, eco-efficiency is thought of as a measure that relates environmental benefits like an amount of pollutants thoroughly recovered and treated to the economic burden necessary to achieve the environmental benefit.

A second parameter of organizational forms is the degree of competition under an EPR approach. Again, it is important to be aware of the fact that measures of the "degree of competition" can be analyzed on each level of the value chain. In collaborative approaches, one producer responsibility organization manages take-back and recycling on behalf of the complete industry. Competitive approaches are characterized by competition between several service providers that compete in an open market for take-back service mandates from producers. Figure 18 illustrates the range of organizational forms and exhibits the positioning of some countries in their EPR approach.

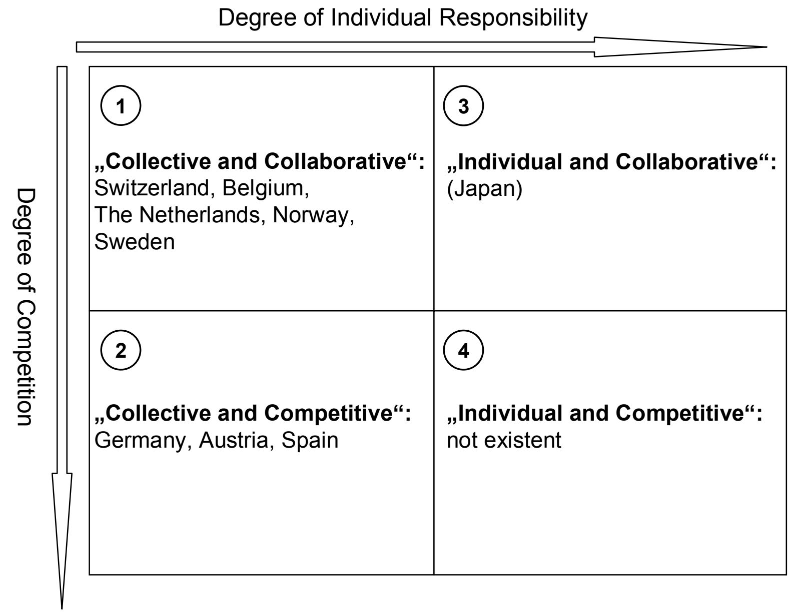

Figure 18: Organizational forms of EPR

3.3 The Evolution of EPR Regulation across the Globe

The establishment of specific take-back regulations for WEEE was, for the first time, discussed in Germany in 1991. However, no specific EPR-driven rules were implemented. Pioneering EPR initiatives in the area of WEEE recycling were to be observed in Switzerland in 1992 with the establishment of a voluntary take-back and recycling system for refrigerators (SENS 2005). In addition, take-back and recycling of IT and telecommunication equipment under producer responsibility was achieved in Switzerland in 1994 (SWICO 2001). These initiatives were followed by a governmental decree in 1998 (VREG 1998) in order to clearly define responsibilities of producers, importers, retailers and consumers. Prompted by environmental authorities, the Netherlands, Belgium, Norway and Sweden prepared and established producer responsibility organizations for WEEE recycling between 1997 and 1999. In each of the five countries, a collaborative approach was chosen to organize take-back systems; therefore, competition among end-of-life managers was not intended but the advantages from economies of scale and easier system organization for such solutions were perceived more valuable.

Parallel to these first steps, EU institutions started discussions on a European directive for WEEE management under EPR, originally as a joint directive with the stipulations today known as ROHS. After vigorous debates over several years, the final version of the Directive 2002/96/EC of the European Parliament and of the Council on Waste Electrical and Electronic Equipment (WEEE) (COM 2003) was released on January 27th 2003 and was published in the official journal of the European Union on February 13th. Member states were required to bring into force the laws, regulations and administrative provisions necessary to comply with the directive by August 13th 2004 (Article 17). After a long transition period, most countries have now started respective take-back systems.

In Asia, Taiwan introduced specific EPR legislation in 1998, followed by Japan where several recycling laws including EPR were enacted in the following years. Korea implemented their EPR legislation in 2003, other Asian countries like China, India, and Singapore have started discussions about regulating the treatment of WEEE under the EPR umbrella. (Hieronymi 2001, Ueno 2003, Ronningen 2005, Panfeng 2004). Several initiatives exist to further the legislative development in these countries and partnerships with experienced countries have been established. (UNEP 2006) provides an overview of current policy initiatives in China.

In North America, several states in Canada and the United States have implemented e-waste programs but EPR legislation or product stewardship supported by specific legislation is rather rare (Gregory et al. 2006). Initiatives to enact EPR legislation on a nationwide level have not been successful so far. In the US, several bills were or currently are under debate with a rather low chance of implementation, e.g. the "Electronic Waste Recycling Promotion and Consumer Protection Act", or the National Computer Recycling Act". A variety of national dialogues such as the National Electronics Product Stewardship Initiative (NEPSI) or the STEP initiative have been trying to tackle e-waste issues and are working on ideas for a nationwide solution. California has almost fully implemented the Electronic Waste Recycling Act which covers CRTs and laptops (SB20/SB50). The Maine program covers the same categories. Other US states have smaller or voluntary programs and/or pending legislation, e.g. Maryland that mandates fees for computer manufacturers to foster computer take-back and Massachusetts and Minnesota that have banned CRTs from landfill disposal. In Canada, Alberta and Ontario have released legislation whereas British Columbia, Manitoba, Quebec and Saskatchewan still seem to be in the process of developing respective regulations. (Perchards 2005, NCER 2005, Hieronymi 2004, Darby et al. 2004) Figure 19 provides an overview of EPR regulation across the globe representing the status by the end of 2006.

EPR legislation status

Europe	Asia	North America
Switzerland, Norway, Sweden, Belgium, The Netherlands: Countries with several years of EPR experience before the WEEE directive implementation **Rest EU-25:** Most countries have started operational systems according to the WEEE directive **Italy and UK:** Systems to start in early 2007	**Japan and Taiwan:** Early adopters around 2000 **Korea:** Implementation in 2003 **China, Singapore, India:** EPR Regulations under development	**Canada:** Alberta and Ontario released regulations in 2004, in other states under development **United States:** • California and Maine with direct recycling laws, Massachusetts and Minnesota with CRT landfill bans • 28 states with pending legislations or legislation under development

Figure 19: Overview of EPR policy making across the globe

All these EPR approaches are characterized by different scopes and a variety of goals. The latter are subsequently portrayed in brief.

3.4 Goals of EPR Regulation for WEEE Recycling

Before analyzing, evaluating, or even designing a policy or an implementation approach, it is vital to clarify goals. Most attempts to define policy-related goals remain on a fairly abstract level because the system's complexity is high. This notion can be supported with a broad array of documents on existing legislation, such as the European WEEE directive, the Japanese Fundamental Law for Establishing a Sound Material-Cycle Society, or the Massachusetts Act to Require Producer Responsibility for Collection and Recycling of Discarded Electronic Products (COM 2003, EAJ 2000, MAS 2005). However, a concise analysis requires to clearly identify and determine goals and to put them in concrete terms. The first step is tackled in this section and reflected in the following goal framework that has been devised in accordance with the stakeholder discussions and interviews during the industry survey.

Environmental goals:

- **Ensure appropriate treatment of hazardous materials:** This is reflected in a sound depollution of WEEE with appropriate disposal of hazardous components and substances and high worker health and safety standards.

- **Spur the closing of material loops** which is possible via a broad scope of products to be recycled, high recycling quotes (defined as amount of recycled products to products becoming obsolete), high recycling rates (defined as percentage of secondary materials produced per waste input), and high material reapplication levels.
- **Reward design for environment**: This includes design for end-of-life (using materials that are easy to recycle, ease of disassembly, avoidance of hazardous materials,…), design for beginning-of-life (allowing the use of secondary materials in production,…), and life-cycle optimized design in general (energy-efficiency,…).

Economic goals:

- **Ensure fair allocation of burdens,** requiring sanctions in case of system abuse and against free riders, a fair distribution of burdens between players and stakeholders, a level playing field in participating economic areas, and a fair burden allocation among producers. The latter is reflected in the avoidance of cross-subsidization, an alignment of burdens with virgin material consumption, and a differentiation of burdens according to a product's end-of-life performance.
- **Minimize compliance costs,** e.g. leverage competitive structures in order to spur efficient services among the whole value chain, avoid financial burdens that do not successfully serve an environmental purpose (dependent on the approach, this could be financial guarantees & provisions, sorting & identification or tracking costs, inefficient monitoring,…).
- **Keep EPR approach simple and clear**, meaning clearly defined responsibilities, transparency, the avoidance of jurisdictial vulnerability, low organizational and communication efforts, and easy system monitoring & control.

Economic incentive structures in existing markets partly support or destabilize the achievement of these goals. EPR regulation should be applied to thoughtfully amend these incentive structures in order to create overall favorable incentives within a system. EPR approaches are analyzed and evaluated in this regard in the next chapter.

4 State-of-the-Art in Policy-Making

This chapter features examples of EPR policy-making and explains the underlying EPR system design. Presented EPR policies and their implementation approaches are afterwards analyzed and evaluated in terms of effectiveness and efficiency, revealing their strenghts and weaknesses and the need for an application of better-focused alternative approaches. Key learnings from the analysis of current implementation practice are summarized in policy design requirements that guide the development of alternative approaches in chapter 5.

4.1 Policy-Making and EPR Implementation Practice

4.1.1 The WEEE Directive

The European WEEE directive is regarded as a raw model for legislation and several countries across the globe consider the adoption of similar legislation. The main features of this framework are as follows:

The directive 2002/96/EC of the European Parliament and of the Council on Waste Electrical and Electronic Equipment (WEEE) (COM 2003) was released on January 27th 2003 and published in the official journal of the European Union on February 13th. The directive's purpose is "the prevention, reuse, recycling and other forms of recovery of WEEE so as to reduce the disposal of waste" (Preface). The directive is based on the principle of extended producer responsibility but its provisions can be interpreted differently in the implementation among member countries. Basically, producers or parties on their behalf are responsible for establishing collection schemes and ensuring appropriate treatment of WEEE (Article 5). Furthermore, member states shall encourage the design and production of EEE which take into account and facilitate dismantling and recovery, in particular the reuse and recycling of WEEE, their components and materials (Article 4).

Take-back opportunities must be provided for consumers free of charge and distributors are required to accept product returns if a new device of the same kind is purchased. Producers are allowed to set up and operate individual or collective take-back systems for WEEE from private households, provided that these are in line with the objectives of the directive. A collection goal of 4 kg per capita per year is set to ensure a minimal system performance.

Producers are required to provide, at a minimum for the financing of collection, treatment, recovery and environmentally sound disposal of WEEE from private households. For products put on the market later than August 13th 2005, each producer shall be responsible for the financing of the operations relating to the waste from his own products. Historic orphaned waste must be collectively financed, based on market shares of producers. Furthermore, each producer shall provide a guarantee when placing a product on the market showing that the management of all WEEE will be financed. In addition, products have to be clearly marked in order to allow an identification of the producer (Article 8, Article 11).

The scope of the directive comprises all sorts of WEEE, classified in 10 product categories. The treatment performance is addressed in Article 7 where goals for material recovery on a weight basis are set. A distinction is made between component, material and substance reuse, recycling, and recovery. Mandatory recycling rates are defined for both specifications, and the respective values and categories are presented in Table 15.

Product Category	Reuse and Recycling	Recovery
1 Large household appliances 10 Automatic dispensers	75%	80%
3 IT and telecommunication equipment 4 Consumer equipment	65%	75%
2 Small household appliances 5 Lighting equipment 6 Electrical and electronic tools 7 Toys, leisure and sports equipment 9 Monitoring and control instruments	50%	70%
5 Lightning equipment	80%	80%
8 Medical devices	Not defined	Not defined

Table 15: Product categories and recycling and recovery rates

In addition, an important practical part of the directive is the compulsory selective treatment for certain and in most cases potentially hazardous WEEE. Several hazardous components like PCB-containing capacitors and all fluids have to be removed from WEEE. The definition of additional quality standards by member states is encouraged. Furthermore, a certification of treatment facilities is mandated and annual quality inspections are required. Table 16 shows the depollution requirements set in annex II of the directive.

WEEE Directive Annex II: Depollution Requirements	
PCB-containing capacitors, Mercury containing components, Batteries, Printed circuit boards > 10cm^2 , Toner cartridges, Plastic containing brominated flame retardants, Components which contain asbestos,	Cathode ray tubes, Gas discharge lamps, CFCs, HCFS, HFCs and HCs, Liquid crystal displays > 100cm2, Components containing refractory ceramic fibres, Components containing radioactive substances, Electrolyte capacitors containing substances of concern.

Table 16: Requirements for the removal of hazardous parts from WEEE

Additional rules are required to define the funtioning of an EPR system, e.g. the concrete organizational form, the division of responsibilities, or the timing and anchorage of financial burdens. The WEEE directive as well as other similar legislation only serves as a framework for policy-making. Two implementation examples representing most common EPR system design practice are subsequently presented.

4.1.2 Collective & Competitive: EPR Implementation in Germany

The German EPR legislation for WEEE, the “ElektroG”, has verbally adopted most of the text of the WEEE directive. Goals that were only vaguely addressed by the directive, such as the product design feedback (Article 4), were not further put in concrete terms in the legislation. Industry was granted the choice to assume producer responsibility collectively. The determination of the EPR system design was left to the electronics industry that founded a private regulatory authority for this purpose, the “Stiftung Elektro-Altgeräte Register” (further referred to as EAR). German public environmental authorities merely ask for compliance with the basic rules set out in the legislation (Theusner 2005). The key features of the current German system are as follows:

Municipalities and producers share the responsibility for waste management, the former being responsible for WEEE collection and the latter being responsible for WEEE transport, treatment, and quality assurance. Municipalities are not obliged to provide a defined collection infrastructure but consumers can dispose of WEEE free of charge via municipal garbage stations or other offered forms of collection. Municipalities are, however, obliged to collect WEEE in 5 groups defined in the ElektroG. They can opt to market each group individually or hand them over to responsible producers. Therefore, WEEE must be consolidated at so-called municipal hand-over points. Such hand-over points are to be equipped with collection boxes by producers or their service providers. Hand-over points run by municipalities notify the central public authority when a full box at their collection site is ready for pick-up. When EAR receives a notification, a producer/importer is chosen from a database that tracks the obligation fulfillment status of each producer/importer, the latter reputedly being calculated on the basis of market shares. The fulfillment status goes up with each take-back prompt followed. Direct WEEE collection by a retailer with subsequent transport and treatment can be credited to the account of producers/importers in terms of their fulfillment status. Independent of municipal collection, informal collection by private businesses and component harvesting can be witnessed to a considerable extent. These activities are not tracked by EAR.

In order to calculate market shares, EAR collects sales data from producers and importers and calculates market shares in each of the ten categories defined by the directive. A proprietary algorithm serves to match the market shares from ten product categories with five collection groups. An identification or distinction of individual, historic, or orphaned wastes does not occur in the system.

The allocation of take-back prompts is centralized and under the control of EAR. Each producer/importer faces randomized and changing take-back prompts all over Germany. This shall avoid cherry-picking (Theusner 2005). As a result, take-back service providers act as end-of-life managers in the German market, offering take-back networks covering the whole country. Several of these system providers compete for the mandates from producers and importers and the competitive mechanism is deemed responsible for the comparably low recycling service remunerations in Germany (Hieronymi 2007).

Quality assurance is required by producers and importers and each registered company is supposed to individually provide evidence of the legal compliance of their recycling service partners. How to provide such evidence, for example the achievement of recycling rates and the realization of depollution requirements as mandated in the directive, is still debated among experts (Hornberger 2007, Rhein 2007).

Figure 20 illustrates the German system design and shows players, material and monetary flows.

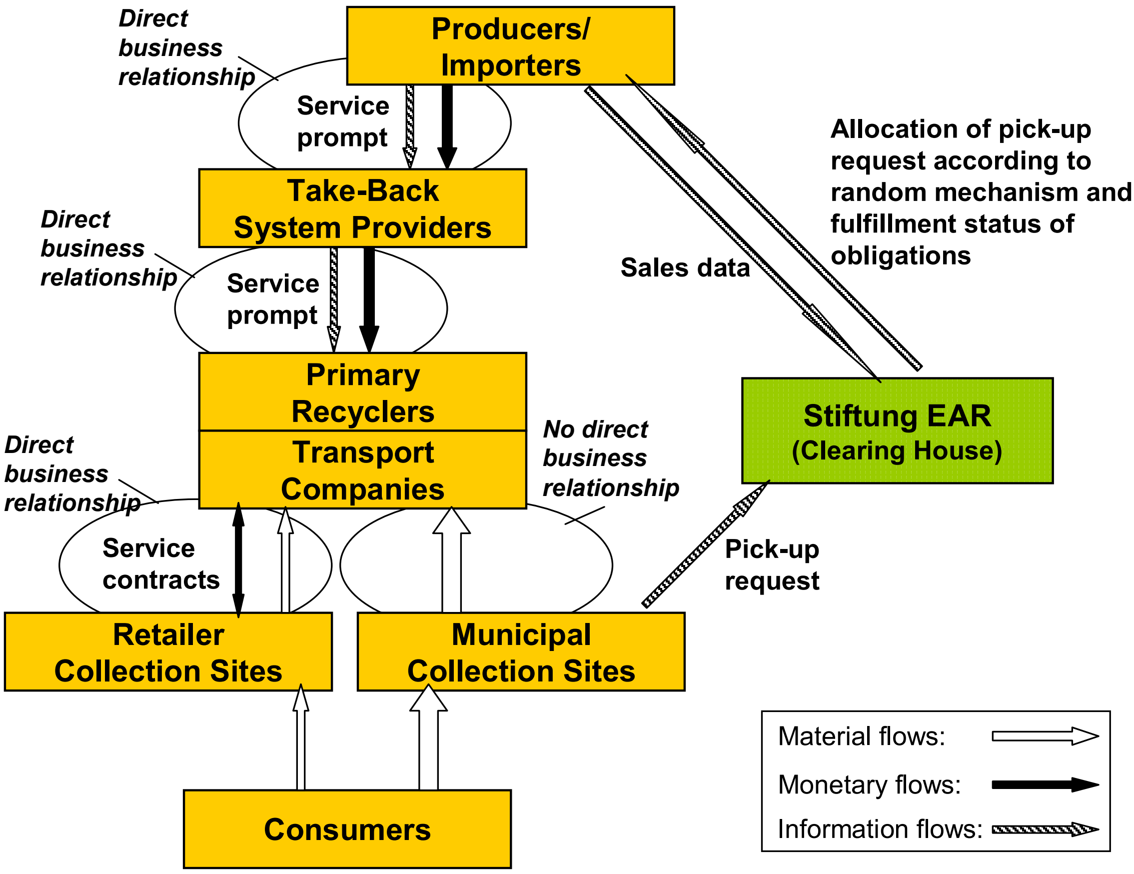

Figure 20: The German EPR system design

4.1.3 Collective & Collaborative: EPR Implementation in Switzerland

Switzerland features a collaborative approach to fulfill the entire industry's collective responsibility. The responsibility for WEEE management is shared by two institutions, each governing the whole market in their WEEE segments. The Swiss Economic Association for Information, Communication and Organization Technology (SWICO) formed an organizational structure to organize take-back and recycling of office equipment and some leisure electronics in 1994. Since then, the scope of covered products has been increased. The Foundation for Waste Disposal (SENS) began with the recycling of refrigerators in 1992. Today, SENS manages all large household appliances, several categories of small equipment, and lighting equipment.

The SWICO approach represents the typcial collaborative system practice in Europe and is therefore further portrayed. Figure 21 provides on overview of the system. At the time of purchase of a new product, Swiss consumers pay a visible advanced recycling fee. The fee is collected at the point of sale and transferred to the producer responsibility organizations SWICO and SENS. SWICO uses these funds to pay for waste collection, transport, treatment, its own administration, and to build up provisions for future liabilities. The VREG provisions mandate consumers to seperately return WEEE to collection points (VREG 1998). Retailers are obliged to take back any old device if they sell the same kind and manufacturers and importers must take back their own brands. Aside from these two return options, consumers can fulfill their "return obligation" via municipal and SWICO collection sites. Altogether, the consumer can use a broad take-back infrastructure and a dense network of collection points across the country (SENS 2005).

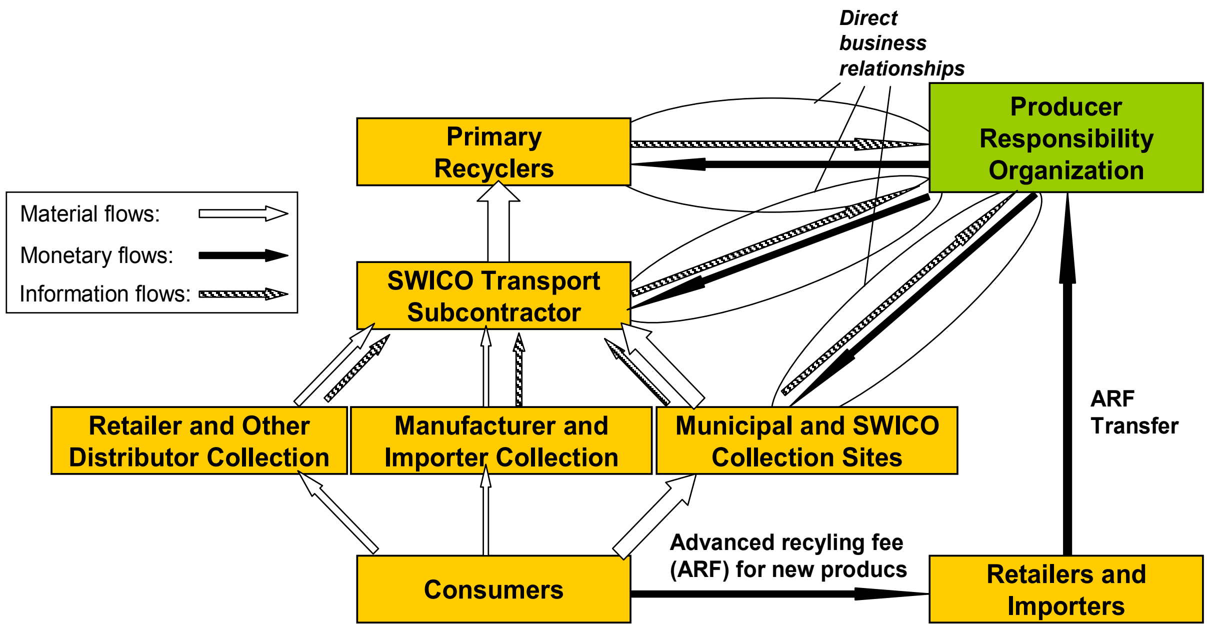

Figure 21: A Swiss EPR system design

WEEE is collected by SWICO's transport subcontractor from all collection points over the country. The logistics company then transports the respective amounts to subcontracted recyclers according to SWICO's allocation planning. The transport subcontractor weighs collected WEEE and enables SWICO to thoroughly pursue material flows in the take-back system. Both transport subcontractor and recycling partners are determined every 2 years with a call for tender process. Recyclers are prompted to offer treatment prices in relation to the overall WEEE volume they potentially obtain. After the determination of recycling partners and volumes, the waste streams are centrally allocated and managed through SWICO. In 2006, SWICO had 13 treatment partners and some of them collaborated with further subcontractors, e.g. social institutions with manual dismantling operations. Treatment practice is monitored by SWICO with regular audits that are conducted by technical expert groups.

Furthermore, annual material flow data sheets are submitted by each treatment partner and depollution figures are tracked.

The Swiss case serves well to illustrate the impact of the timing of fees on dedicated collection. Switzerland introduced advanced recycling fees in 2003. In previous years, consumers had to pay when disposing of an item. Figure 22 shows the collected WEEE amounts over time for all categories covered under the systems from SENS and SWICO.

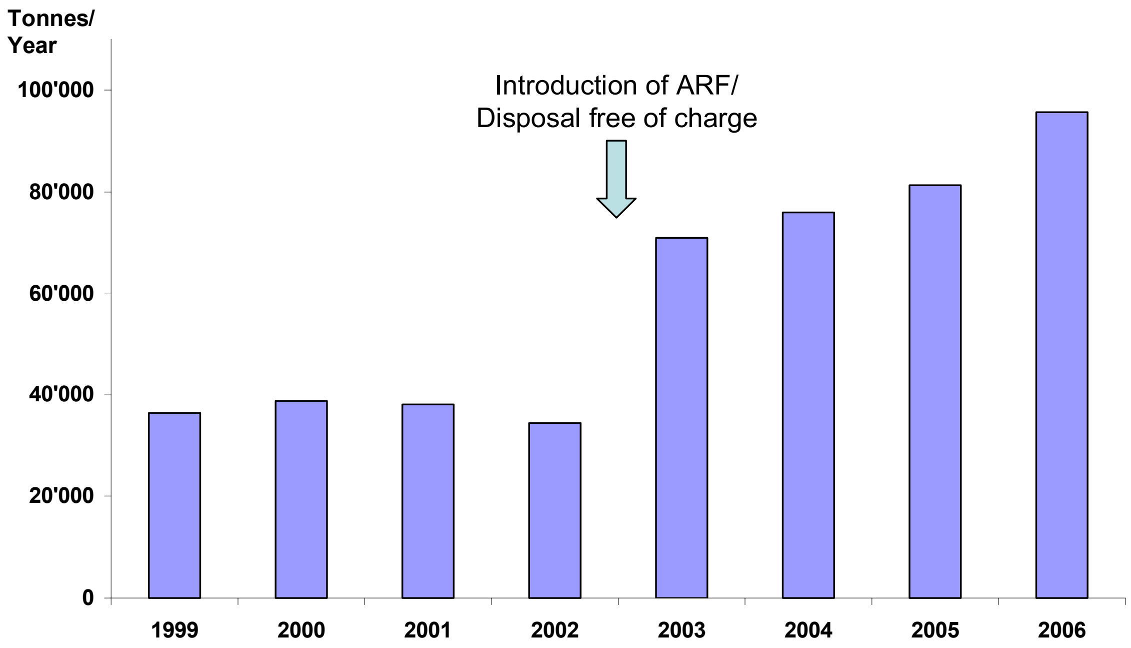

Figure 22: Development of collected WEEE over time in Switzerland

As opposed to the stipulations in the EU WEEE directive, Swiss producers are not individually required to provide guarantees for the future end-of-life treatment of EEE that they currently put on the market. There is a mutual understanding that SWICO and SENS cover such risks collectively on behalf of the producers. Current fees are used for the treatment of current WEEE streams in some sort of generational contract.

4.2 Policy Analysis and Evaluation

A concise evaluation of a regulation's effectiveness benefits from quantitative and traceable figures or ratios that measure the achievement of goals. The evaluation of policies in this thesis relies on both qualitative and quantitative analysis. Frameworks for an assessment of EPR systems for WEEE recycling have been compiled in (Bohr 2005, Tasaki et al. 2005, Widinski et al. 2003). The assessment in this thesis is structured according to the goals presented in the previous chapter. Concrete quantitative indicators to measure the achievement of goals are presented and embedded in respective sections on each goal that are presented successively.

4.2.1 Appropriate Treatment of Hazardous Materials

Appropriate treatment of hazardous materials is manifested in a sound depollution of WEEE with subsequent appropriate disposal of toxics and in the adherence to appropriate worker health and safety standards. A sound depollution is addressed with annex II of the WEEE directive. All components and materials of concern have to be removed according to the WEEE directive which translates into a depollution quality level $dql_{i,k}$ = 100% for all inputs. Recyclers control and decide on their depollution practice; however, economic disincentives can discourage the adherence to high depollution quality levels. An analysis of a recycler's decision-making helps to support this notion:

In principle, it is reasonable to assume that each individual recycler strives for a maximization of his profits. In doing so, he will account for all costs and revenues relevant in the decision-making context. This further implies that he optimizes the choice about a certain depollution quality level $dql_{i,k}$ according to dependant costs and revenues. The balance of these costs and revenues is further depicted as decision-relevant profit DRP. Dependant on the anchorage of monetary flows in an EPR system, some of the decision-relevant costs and revenues vary with dql_i whereas others are independent from dql_i and insofar irrelevant for his decision-making.

Potentially relevant revenues in this decision-making context are recycling service remunerations rsr_i and secondary material revenues smr_i. Under current EPR systems, a recycler receives recycling service remunerations as gate fees when accepting WEEE material for processing (input-based remuneration). This implies that rsr_i do not directly influence the recycler's decision on dql_i. In principle, secondary material revenues are as well independent from dql_i. Secondary material revenues can be affected if the recycler does not achieve a certain minimal depollution quality level, dependent on the hazardous material and WEEE segment. However, this level is rather low and no relation between smr_i and dql_i exists above this limit.

Potentially relevant costs are depreciation and amortisation from investments $daic_i$, depollution costs adc_i, other fixed costs ofc_i, variable costs from operations vco_i, costs for material disposal mdc_i, and expected monetarized economic disadvantages from non-compliance $edfn$. The expected monetarized economic disadvantages from non-compliance represent fines under a command & control regulation or a recycler's fear to suffer from any other economic disadvantages resulting from not being compliant with regulatory requirements (e.g. reputational damage). The $edfn$ are assumed to depend on the probability $pncn$ to be checked and discovered as non-compliant (strongly correlated with the monitoring and control efforts in a system) and the actual extent of non-compliance, manifested in the depollution quality level dql_i applied by a recycler.

Analyzing a recycler's operations reveals that only depollution costs adc_i, expected economic disadvantages from non-compliance *edfn*, and costs for material disposal mdc_i are affected by dql_i. Costs for material disposal mdc_i increase with dql_i because more depollution means more separated mass of hazardous materials that need to be disposed of appropriately. The decision-relevant profit is displayed in equation (12):

$$DRP = (rsr_i + smr_i) - (daic_i + adc_i(dql_i) + ofc_i + vco_i + mdc_i(dql_i) + edfn(pncn, dql_i)) \quad (12)$$

In a mathematical framework, the condition for profit maximization is:

$$\frac{\partial DRP}{\partial dql_i} = 0 = \left(\frac{\partial adc_i(dql_i)}{\partial dql_i}\right) + \left(\frac{\partial mdc_i(dql_i)}{\partial dql_i}\right) + \left(\frac{\partial edfn(pncn, dql_i)}{\partial dql_i}\right) \quad (13)$$

However, the function can be analyzed graphically for a more intuitive understanding of a recycler's incentive structures. Figure 23 shows a typical curve for the relation of adc_i, mdc_i and the sum of adc_i and mdc_i to the depollution quality level dql_i. The minimal depollution level to avoid considerable losses in secondary material revenues smr_i is displayed as well. The graph shows that a minimal dql_i is optimal if no monitoring and enforcement efforts existed.

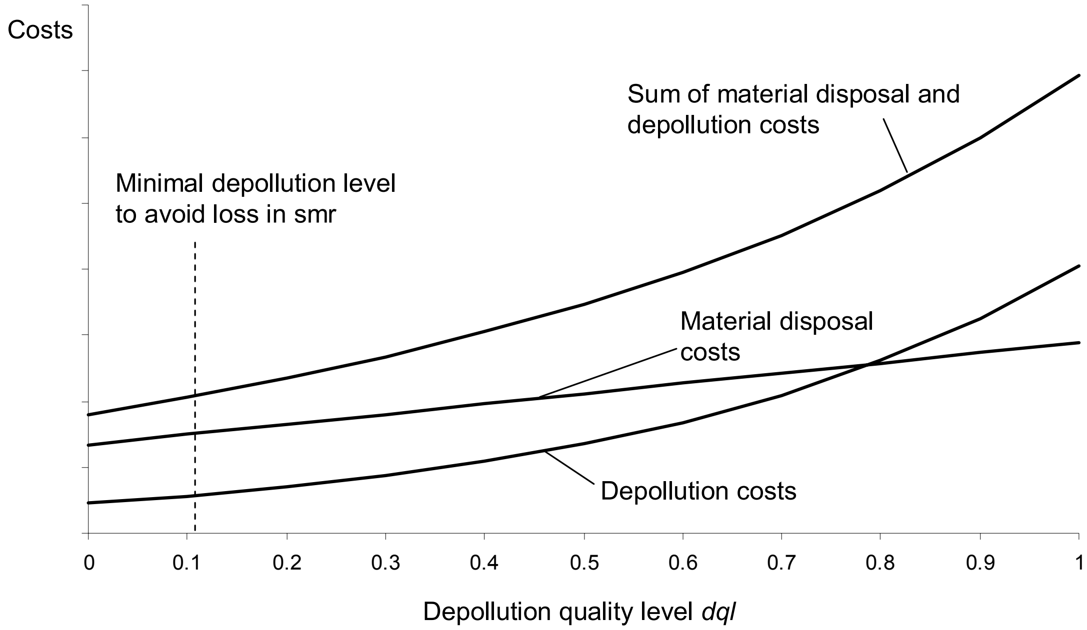

Figure 23: Depollution and material disposal costs

Figure 24 shows an exemplified relation between *edfn* and dql_i. The graph is based on the following assumptions: Monitoring and enforcement efforts of a responsible control body are reflected in a probability *pncn* that characterizes the likelihood of a recycler to be assessed and discovered as non-compliant. The higher the probability *pncn*, the higher are the expected monetarized economic disadvantages from non-compliance *edfn* for a recycler. Such economic disadvantages for recyclers can result, for example, from fines, reputational damage, or a loss of future mandates to recycle WEEE. This relation is visualized with a cohort of functions displaying *edfn* for different values of *pncn*. Further, *edfn* increases with decreasing depollution quality level dql_i. The slope of this function is assumed to increase for decreasing depollution quality levels which is why the relation is covered with an exponential curve. Other curve types would be possible as well to describe the relation.

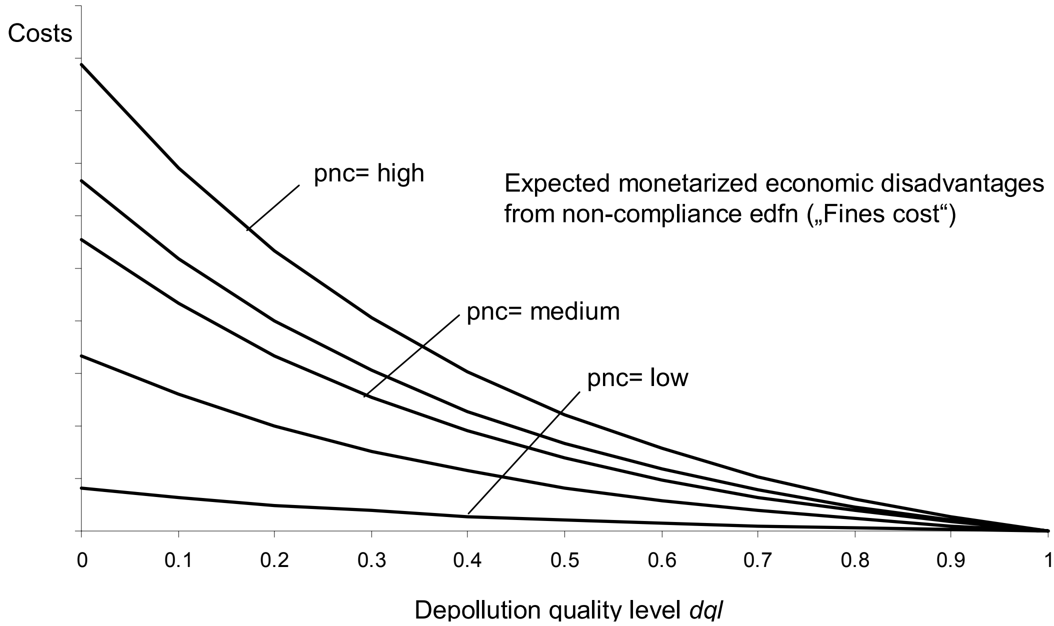

Figure 24: Expected monetarized economic disadvantages from non-compliance

Both figures are consolidated in Figure 25 that shows the recycler's profit maximum (respectively cost minimum) in relation to the quality level dql_i and the monitoring efforts. Two scenarios are visualized, one with low monitoring efforts and one with high monitoring efforts.

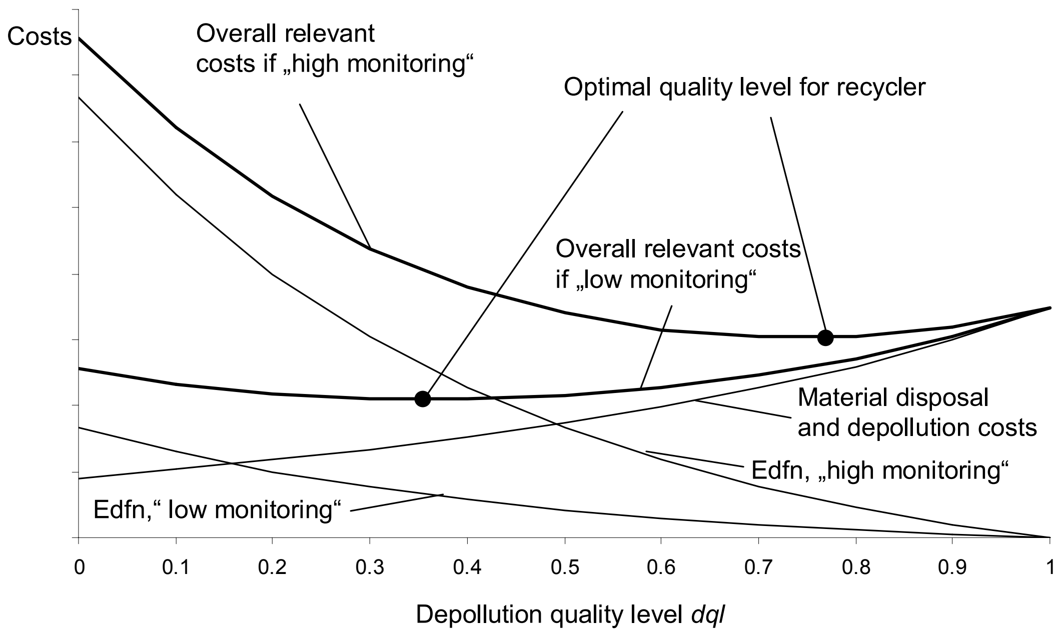

Figure 25: Optimal *dql* if recycler strives for profit maximization

As Figure 25 reveals, monitoring and enforcement is crucial for the achievement of quality standards under a command & control approach such as the WEEE directive. However, most countries analyzed during the industry survey do not have effective monitoring systems for depollution quality. Furthermore, the independence of a controlling body is rarely guaranteed.

Worker health and safety standards are not explicitly addressed by the WEEE directive or corresponding country-specific e-waste legislation; however, these standards are normally under the control of several public authorities and appropriately enforced in developed countries. Unfortunately, this is not the case for countries that are typically receiving obsolete electronics such as Nigeria or China (BAN 2002, BAN 2005). Minimizing illegal exports would help alleviating this issue. This aspect will be revisited in several of the next sections and will there be investigated in more detail.

4.2.2 The Closing of Material Loops

The performance of an EPR system in closing material loops depends on several levers. A good regulation approach should attempt to pull the most eco-efficient ones in order to achive a defined target. The performance in closing material loops *PCML* will be introduced and defined in this chapter as such a target. Potential levers to influence this performance are the product scope of the underlying legislation, the dedicated collection amount of WEEE, the recycling rate, defined as the ratio of produced secondary materials to the original input, and the reapplication level of materials.

Figure 26 exemplifies levers, material flows, and their estimated size in a central European country. A comprehensive product scope currently gives rise to a potentially available disposed EEE mass of 17 kg per capita. This flow is drained by illegal dumping (0,2 kg per capita) and disposal via household waste (1,8 kg per capita). The dedicated collection amount of 15 kg per capita is to be distinguished into wastes officially tracked by responsible bodies and wastes informally collected. Officially tracked wastes usually enter controlled material recycling facilities whereas informally collected wastes are likely to enter the most economically rewarding pathway. As an estimate, 0,5 kg per capita is currently reused or remanufactured, 6,5 kg per capita is recycled under tracable material recycling operations, and 8 kg per capita are (illegally) exported with no evidence of utilization.

The performance of material recycling operations is measured with the recycling rate. An average material recycling rate *rec* = 70%[5] is a realistic base estimate for European material recycling operations that leads to 4,55 kg per capita of secondary materials from tracable

[5] The definition and measurement of recycling rates is highly contentious. The estimate stated here is based on the methodology and classification of the reptool (Gabriel 2006).

material recycling operations. Wastes in (illegal) export streams are likely to achieve a certain recycling or reuse performance as well. However, cherry-picking presumably leads to a considerably lower recycling rate since no infrastructure is in place to recover problematic materials. If one assumes an illegal export recycling rate *eirr* = 10% for these WEEE flows, *sme* = 0,8 kg of secondary materials per capita are generated from (illegaly) exported wastes.

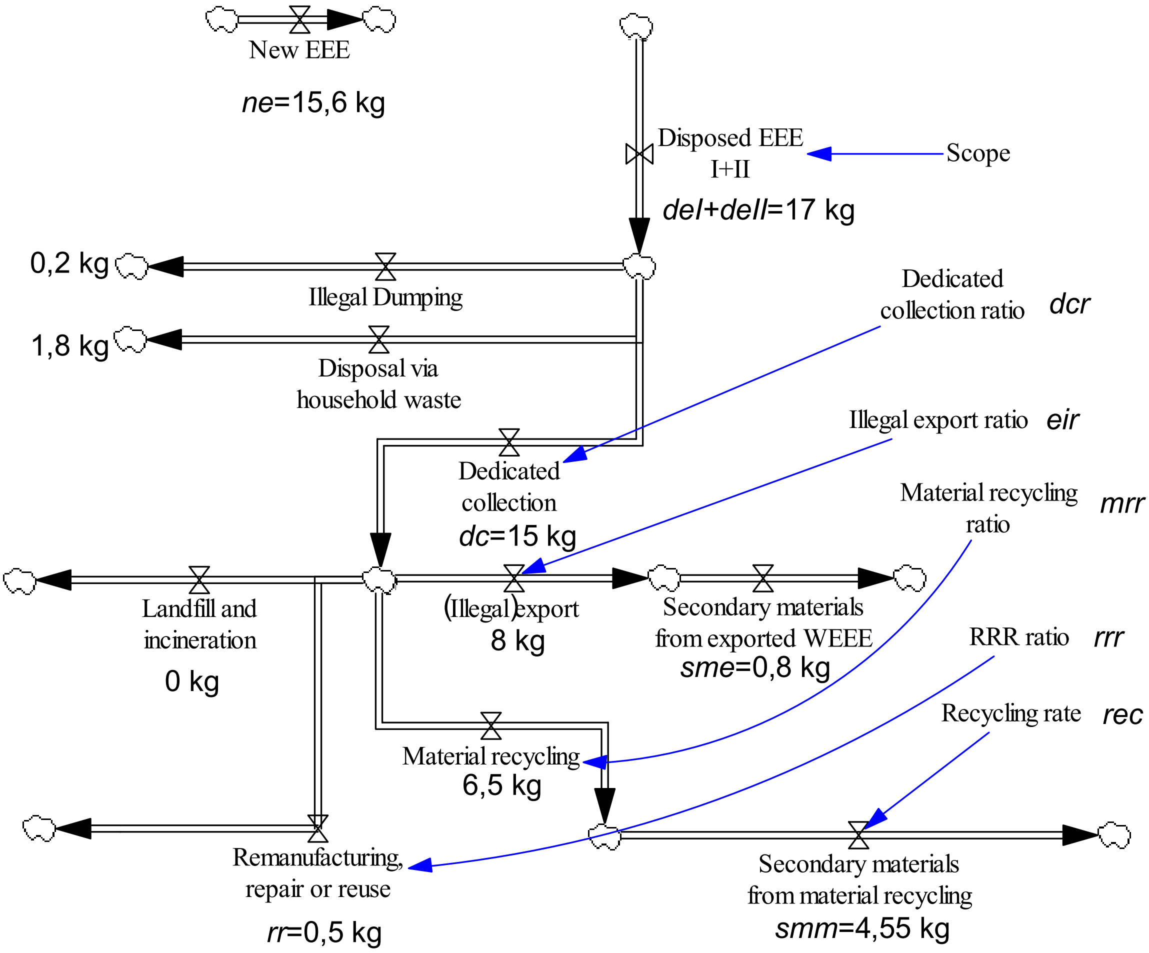

Figure 26: Measuring the performance in closing material loops

An evaluation of a regulation's effectiveness in fostering the closing of material loops should now assess the regulation's impact on the different levers and the overall performance *PCML*. *PCML* should be an indicator for the amount of materials produced/generated under EPR in relation to the amount of resources consumed by the EEE industry. This relation can be measured with the mass of secondary materials or products produced per mass of materials used in production (*new EEE*) as indicated in equation (14).

$$PCML = \frac{sme + smm + rr}{ne} = \frac{(deI + deII) \cdot dcr \cdot ((eir \cdot eirr + mrr \cdot rec) + rrr)}{ne} \tag{14}$$

In principle, the large product scope of the WEEE directive is welcome and ensures a broad basis of materials available for recycling under extended producer responsibility. The directive covers almost any type of EEE with negligible exceptions (Article 2.2, COM 2003). Furthermore, the consumer's right to dispose of WEEE free of charge (Article 5, COM 2003) should help to steer obsolete EEE into dedicated collection channels. Free of charge disposal is likely to decrease illegal dumping, decrease the household bin ratio, and also decrease the average storage span and the storage ratio which makes obsolete EEE earlier available for utilization. However, the collection target for WEEE under producer responsibility, namely 4 kg per inhabitant, is very low. Measured in relative terms, the collection target set by the directive currently translates into a collection rate cr = 4kg/17kg = 23.5% for a central European country. As opposed to the collection target, mandated recycling rates (Table 15) are rather high. However, due to dynamic feedbacks from different levers, such high targets for one lever must not necessarily benefit the overall goal, measured with *PCML*. In order to understand such feedbacks, it is helpful to analyze a recycler's decision-making:

Any recycler that collects or obtains a range of WEEE faces different options for its utilization. When deciding for an option, collection costs, transport costs and sorting costs have usually already been incurred. These costs are insofar irrelevant for his decision-making. In accordance with the material flow classification in this thesis, a recycler's options are to reuse, remanufacture or repair EEE, recycle it on a material basis, or export it illegally (legal exports are summarized under material recycling). Based on the decision-relevant costs and revenues, he will evaluate these options for a range of WEEE at his disposal in terms of their decision-relevant profitability:

- In the non-industrial sector, the profitability of reuse, repair or remanufacturing is limited due to the quality of returned WEEE materials and only few selected appliances are potential candidates for a respective utilization. In such case, profit can be derived from the resale value *rsv*, lessened by repair or remanufacturing costs *rrc*.
- The economics of material recycling differ between segments and WEEE batches and further depend on the processing costs which are influenced by the depollution quality level and the mandated recycling rate alike. Relevant cost positions for this option are depollution costs *adc*, variable costs from operations *vco*, costs for material disposal *mdc*, and expected monetarized economic disadvantages from non-compliance *edfn*. Relevant revenues are secondary material revenues *smr*.

- The decision-relevant profit from illegal export consists of WEEE sales revenues *wsr*, lessened by shipment costs *shc* and expected monetarized economic disadvantages from illegal export *edfi*.

Equations (15)-(17) show the decision-relevant profits for each option:

$$\text{DRP(Reuse)} = rsv - rrc \qquad (15)$$

$$\text{DRP(Material recycling)} = smr - adc - vco - mdc - edfn \qquad (16)$$

$$\text{DRP(Illegal export)} = wsr - shc - edfi \qquad (17)$$

These costs and revenues differ among recyclers in practice for several reasons. Examples are variations in the distance to a harbor that cause different shipment costs, or different business relationships to players in export destinations that give rise to different achievable WEEE sales revenues. Furthermore, the expected monetarized economic disadvantages from non-compliance edfn should be considerably higher for officially collected wastes than for informally collected wastes. However, the basic impacts of policy-making are similar for each recycler. Figures 27-30 visualize the decision-making framework of a recycler. The x-axis displays a continuous range of different WEEE types at the recycler's disposal. These are arranged according to increasing WEEE sales revenues per ton from left to right. The y-axis shows the decision-relevant profit for each option in relation to the WEEE type. Figure 27 shows a decision-relevant profit curve for reuse and decision-relevant profit curves for illegal exports. For the sake of simplicity, WEEE sales revenues per ton are assumed to increase linearly from left to right. Similar to the analysis of *edfn* in 4.2.1, the expected monetarized economic disadvantages from illegal export *edfi* depend on monitoring and enforcement efforts of a responsible control body. Those efforts are reflected in a probability *pnci* that characterizes the likelihood of a recycler to be checked and discovered as an illegal exporter. The higher the probability *pnci*, the higher are the expected monetarized economic disadvantages from illegal export *edfi* for a recycler and the lower is the respective decision-relevant profit *DRP*. Again, this relation can be visualized with a cohort of functions displaying *edfi* for different values of *pnci*.

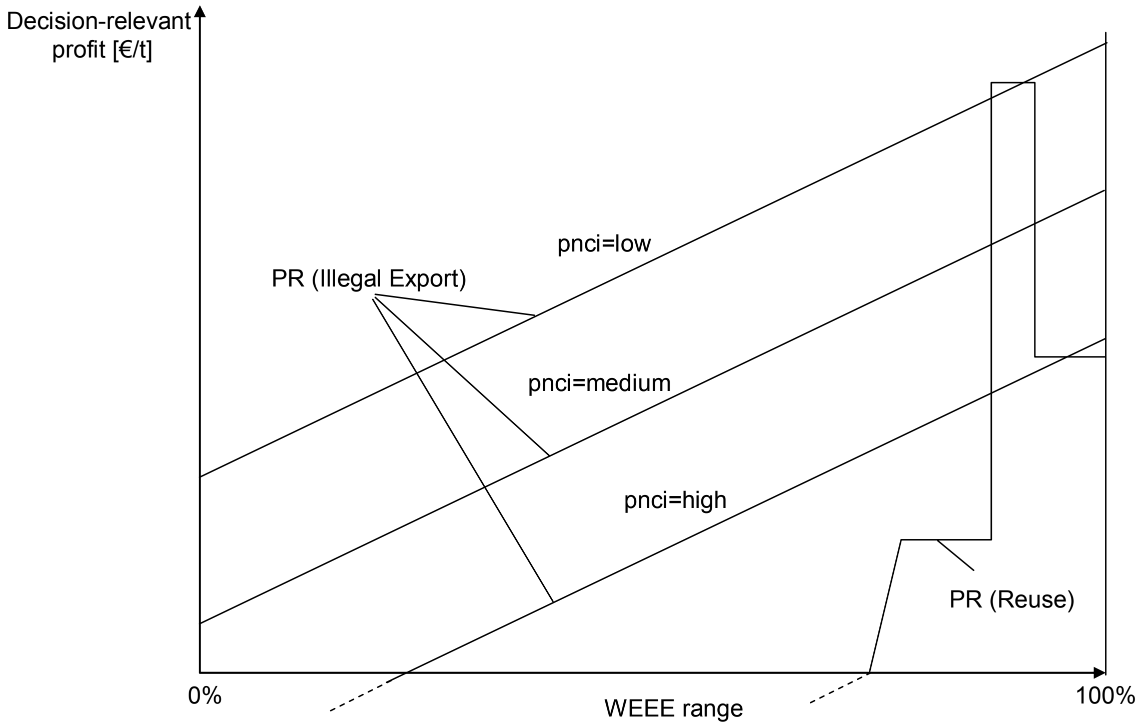

Figure 27: Decision-relevant profits from illegal export and reuse

Figure 27 must be amended by the decision-relevant profit achievable via the material recycling option in order to understand a recycler's utilization decisions. For the sake of simplicity, it shall be assumed that the recycler obtains a fixed amount of WEEE and that he has a defined archetype plant that achieves average secondary material revenues at his disposal. Furthermore, recycling service remunerations are assumed to reflect the processing economics for material recycling among the range of WEEE for mandated treatment rules such as depollution requirements and the recycling rate. Given these assumptions, the profit curve for material recycling according to equation (16) can be approximated with a horizontal line. Figure 28 shows all three profit curves for a fixed set of framework parameters that leads to a similar material flow balance as presented in Figure 26. A recycler will strive for maximal decision-relevant profits among the whole WEEE range and will apply utilization options accordingly. The trajectory of the maximal decision-relevant profit is displayed with a thick line.

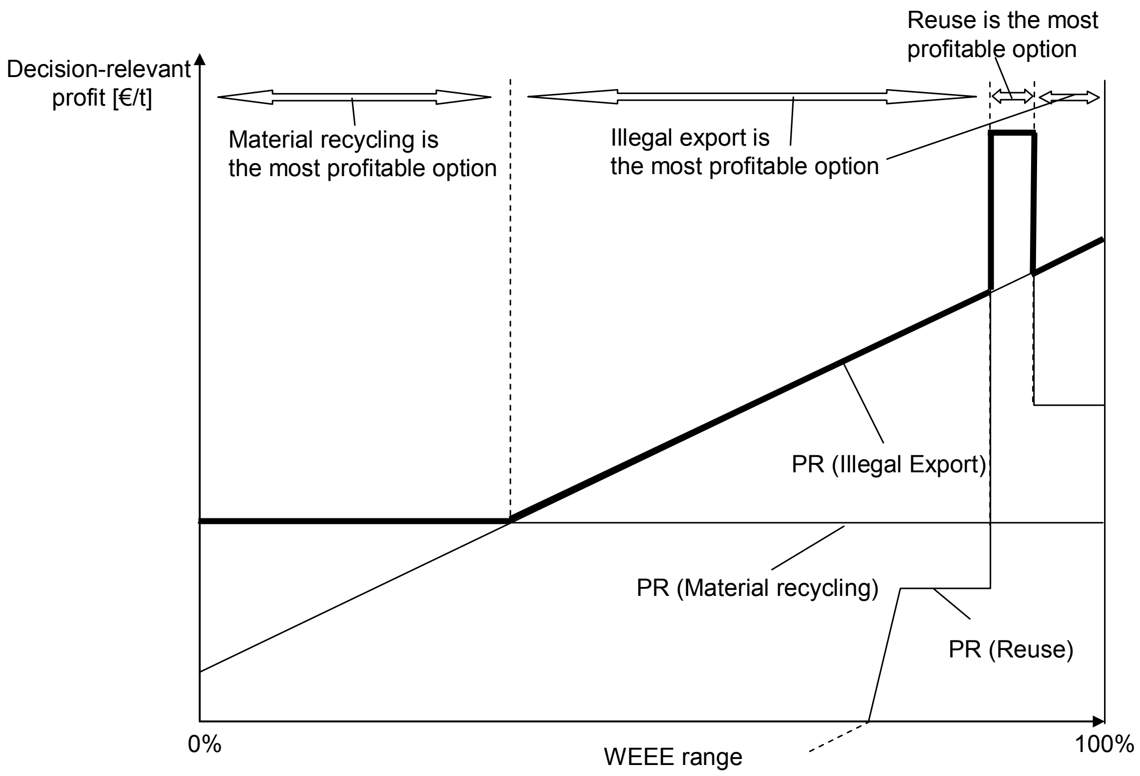

Figure 28: Optimal utilization decisions of a recycler

Now, it is possible to analyze the impact of regulatory policies on a recycler's decision-making and on *PCML*: Mandating recycling rates above the economically optimal recycling rate makes material recycling more costly if any kind of quality monitoring and enforcement exists (*edfn*>0). Recycling service remunerations should reflect these "quality requirements" and go up accordingly. However, recycling service remunerations do not impact the decision-making of a recycler in current EPR systems because these systems work with input-based service remunerations (gate fees). When making the decision on the utilization strategy, a recycler already obtained recycling service remunerations and he will only account for the edfn from monitoring efforts of his principal or public authorities. Recycling service remunerations are insofar irrelevant for the decision-making which is also the reason why these revenues are not found in equations (15)-(17).

As a result, the decision-relevant profit of material recycling decreases and the horizontal line in Figure 28 is shifted downwards. Figure 29 reveals that this increases the share of the WEEE range exported illegally whereas the share going into material recycling is decreased. The same effect occurs if depollution quality or recycling rates are more strongly monitored and enforced. This increases pncn and edfn which decreases the decision-relevant profit of material recycling, resulting again in a downward shift of the respective horizontal line. The overall effect on *PCML* can not be clearly determined. Mandated recycling rates are likely to increase the amount of secondary materials from material recycling; however, they shift WEEE input away from the material recycling option. The relative profitability of material recycling compared to illegal export is decreased with mandated recycling rates, leading to more illegally exported WEEE and less secondary materials due to the low recycling rate in this option.

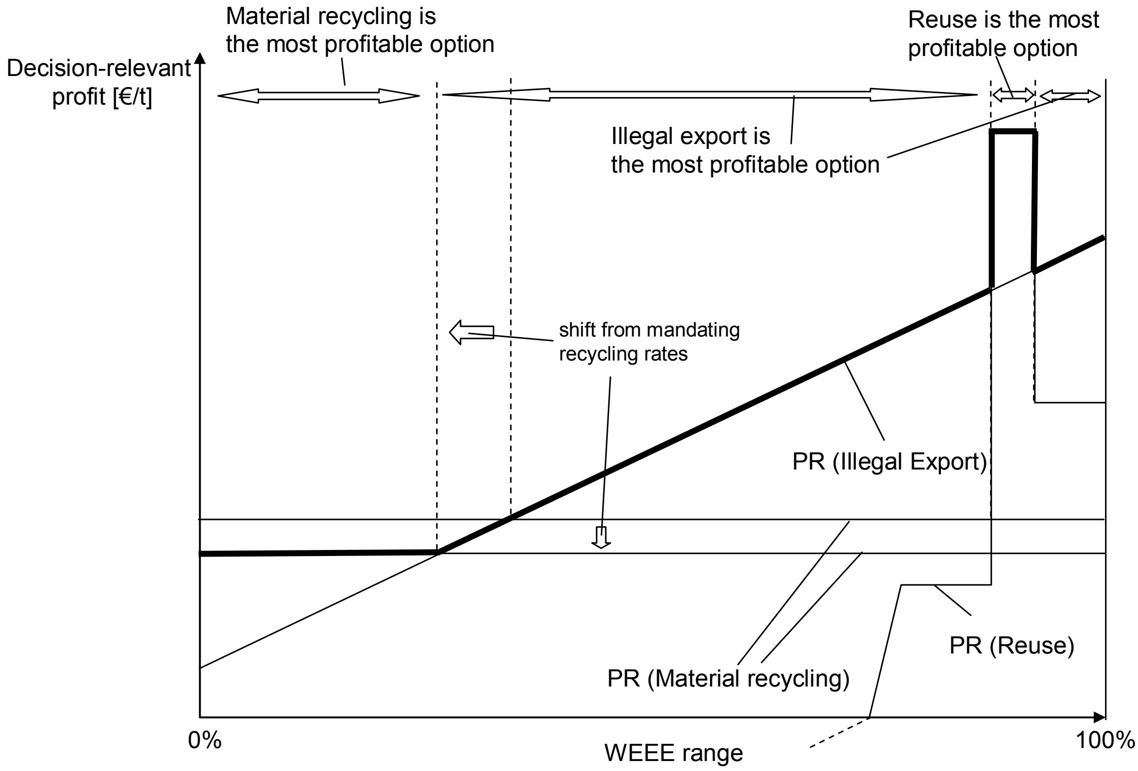

Figure 29: The effect of mandated recycling rates

The WEEE directive does not explicitly address illegal export. However, the interplay of monitoring and enforcement efforts towards recycling quality and illegal export has substantial impact on *PCML*. A considerable focus of monitoring and enforcement efforts in current EPR systems lies on recycling rates due to the WEEE directive. The discovery and avoidance of illegal exports is fought with less attention. Setting such priorities is highly questionable as shown with an impact analysis in Figure 30. Lower efforts for border control give rise to a low *pnci* and higher decision-relevant profits from illegal export. This shifts the profit curve of illegal exports up, increases the mass share of WEEE illegally exported, and lowers *PCML*.

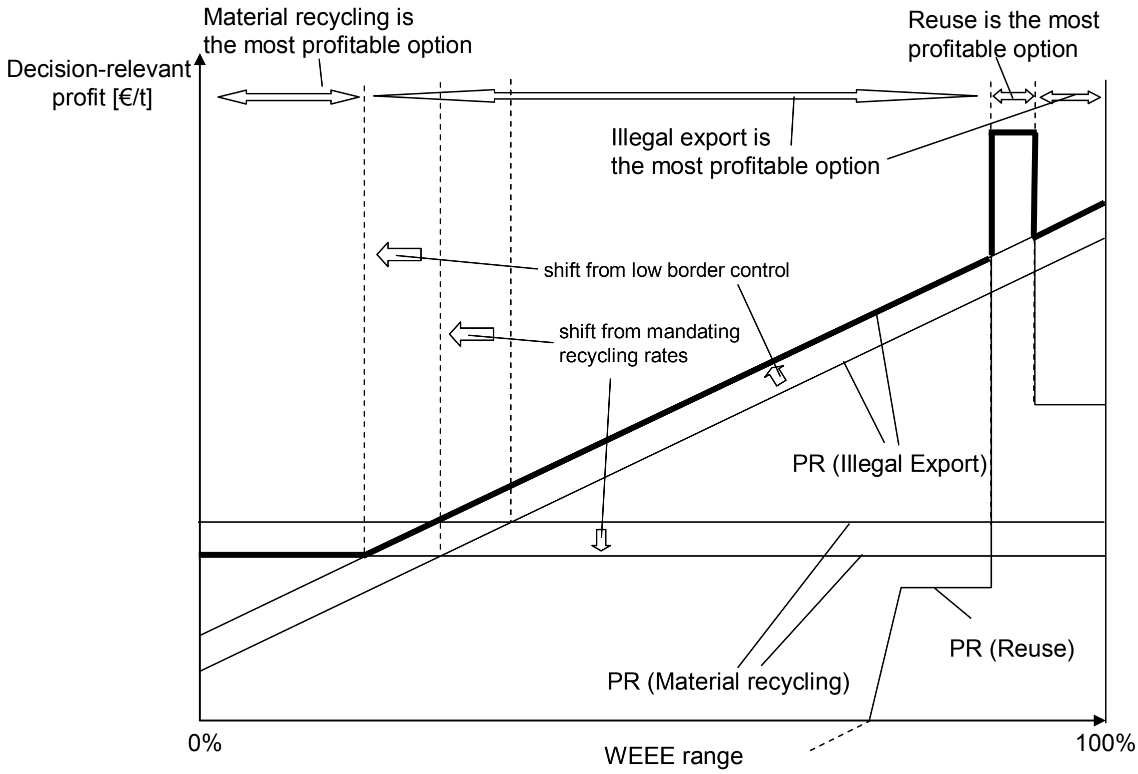

Figure 30: The effect of different monitoring and enforcement efforts

The analysis shows that such priorities are counterproductive for high *PCML* results. In addition, higher shares of illegal export are less beneficial for the achievement of appropriate depollution levels as indicated in the previous section. A shifting of the monitoring and control focus is apposite.

The fourth *PCML* lever that has been supressed in the quantitative analysis so far is the reapplication level of materials. A quantified analysis of this lever is difficult because attempts to operationalize this aspect are still in their infancy (Kasser 2006, Hug 2006). The reapplication level of materials is only indirectly addressed by the WEEE directive with rates for material recycling and material recovery. The respective classifications of some final technologies are debated among experts (Gabriel 2006, Hornberger 2007). Research on the environmental benefits of different material reapplication levels has been conducted by (Huisman 2003); however, tracking and assessing each material reapplication technology is difficult in a dynamic market. In principle, recyclers control and decide on the reapplication level of materials and economic incentives do partly spur high material reapplication levels due to higher achievable fraction prices.

4.2.3 Incentives for Environmentally Favorable Product Design

"Member states shall encourage the design and production of electrical and electronic equipment which take into account and facilitate dismantling and recovery, in particular the reuse and recycling of WEEE, their components and materials." (WEEE directive, Article 4, COM 2003).

Article 4 of the WEEE directive addresses a core constituent of the actual EPR concept: the establishment of a product design feedback loop to the manufacturer. However, the directive itself does not give a concrete recommendation how to implement such a design feedback. (Physical) individual producer responsibility as defined in (Tojo 2003) has been discussed as a means to achieve a design feedback (van Rossem et al. 2006). However, if producers do orient their decision-making on economic profit, the concept is highly unlikely to spur any design changes due to the following reasons (Bohr et al. 2007a):

First, a large time gap between between "design investment" and potential "financial return" must be bridged since a product's use span ranges from several years up to 15 years. Second, the analysis of material flows in the end-of-life sphere shows that the amount of appliances in take-back is likely to be considerably lower as the amount originally put on the market. Third, if producers are to benefit from their design efforts, a respective treatment and tracking of their WEEE is necessary. This must involve expertise and recycling relevant information being exchanged between recyclers and producers. Identifying items in returning product streams is required as well as sorting from the stream of products that are not specifically designed. All these activities involve considerable costs. Fourth, since some WEEE has a trading value, producers will have substantial difficulties to steer their products into their own take-back channels under current economic conditions for recycling.

Other forms of burden differentiation according to product design would be appropriate in order to provide design incentives. The criteria for the burden differentiation could take end-of-life issues or more general life-cycle design considerations into account. However, in the absence of other commonly acknowledged means, member countries have neglected article 4 while transposing the directive into national law (Perchards 2005, van Rossem et al. 2006). Producers have set up collective take-back systems with no burden differentiation according to product design. Either one or several system operators assume a collective responsibility on behalf of producers, treating "anonymous" wastes whose recycling costs are split among producers according to their current market shares. Collective systems make sense because they allow leveraging economies of scale in several areas. However, as has been pointed out in (Lee et al. 2004, Stevels 2003), such collective responsibility does clearly not provide a design feedback if no additional measures are taken. Insofar, the WEEE directives as well as public authorities responsible for implementation in Europe have failed to address a key concern of EPR. The lack of respective incentive structures compromises the credibility of the whole regulatory approach.

4.2.4 Clear and Simple Rules

Transparent, simple, and unified rules and targets benefit the efficiency of interactions between different players in a system. Regulation policy should define clear responsibilities and quantifiable target indicators that can be determined and monitored with reasonable effort. More generally, the interplay of market forces and regulatory stipulations should create a clear and simple system design.

The WEEE directive doesn't serve this purpose very well. Vaguely defined stipulations have created a versatile and intransparent EPR system landscape across Europe. Producers face a complex construct of different rules in different countries because the stipulations of the directive were interpreted differently on a national basis. In addition, target indicators and responsibilities were not skillfully defined. First, the terms "producer" and "put on the market" require a sound definition. National authorities have not consequently tackled this task. Second, the WEEE segmentation defined in the directive is highly unfortunate. The directive defines 10 product categories and mandates recycling and recovery rates in each category. These categories do neither represent collection practice nor treatment practice. As a result, the directive's segmentation gives rise to several costly activities that are only required for accounting purposes and that do not create any environmental benefit. Usually, WEEE is officially collected in 5 boxes. Due to the mismatch of monitoring and collection, players are required to continuously sample the composition of collection boxes in order to allow for an allocation of burdens to producers according to the 10 product categories. Furthermore, the directive requires evidence of achieved recycling rates in 10 product categories. Running batches according to the 10 product categories is nonsense from a processing economics point of view. WEEE is treated in 5 or 6 groups due to technical and economic framework conditions. However, the determination of recycling rates from 5 or 6 treatment streams and their allocation to 10 product categories is complex and can not be determined on a sound basis.

The actual determination of recycling rates deserves clarification, too. A consistent determination of recycling rates requires a sound methodology and a clear classification of final reapplication technologies which is not provided by the directive. Problems emerge with the arbitrary classification of various technologies as material recycling, material recovery, or disposal in different countries. Several experts vigorously debate these aspects and different classifications can be reasonably supported (Kasser 2006). A non-harmonised approach distorts competition and allows for arbitrage across national borders within the EU. The most economic and simple approach would be to collect and monitor WEEE according to material recycling treatment practice and to prescribe a consistent methodology for the determination of recycling rates. This means that collection, monitoring, and treatment would be aligned according to practical requirements.

The rules governing concrete system design and organization fall under the responsibility of national authorities. Complex and poorly thought-out systems are insofar a national phenomenon and can not be blamed on a framework directive. The German system from Stiftung EAR serves well to illustrate some adverse impacts from poor system design. As already described, Stiftung EAR applies a centralized allocation mechanism for take-back obligations. Such a system obstinately ignores the benefits of regional partnerships. Instead of relying on cooperation with a restricted number of recycling and logistic partners, producers have to organize take-back all over the country. This gives rise to considerable additional efforts for communication, price negotiation and coordination of logistics, and destroys economies of scope and scale. Other issues arise as well, e.g. with respect to the ownership of collection equipment. However, the most critical concern in such a system is the missing business relation between recycler and waste collector. Under current market conditions, WEEE is exploited at collection sites and valuable parts are taken off and sold independently if no direct business relation exists between collectors and recyclers. This can compromise the achievement of environmental goals to a considerable extent, e.g. if compressors are already removed from refrigerators without proper removal of CFCs from the cooling circuit. Furthermore, the calculation basis for environmentally sound WEEE processing is distorted since a recycler is suddenly deprived of valuable income streams.

4.2.5 Minimal Compliance Costs

Compliance costs comprise costs for collection, transport, and treatment of WEEE as well as other costs such as transaction costs or the need for provisions in the balance sheet. A clearly defined market environment with transparent responsibilities is a precondition for low compliance costs. Furthermore, EPR systems that leverage competitive structures on several value chain levels are likely to achieve minimal compliance costs. Both competitive and collaborative approaches have their strengths and weaknesses in this regard. Producers in small countries such as Belgium, Switzerland, Norway, Sweden, and the Netherlands often work with only one producer responsibility organization (PRO) that rules the market and coordinates and finances take-back, logistics, and recycling (collaborative approach). This allows for regional partnerships and well-established teams to perform the necessary tasks in a take-back system. Furthermore, economies of scale can be leveraged and competition among recyclers and transporters can be initiated with calls for tenders. However, a lack of competition on the system provider side may lead to inefficiencies or high prices because of the considerable market power of the PRO. Nevertheless, producers are liberated from WEEE recycling management and planning. Competitive approaches as to be found in Germany, Austria or Spain partly seem to achieve lower costs (Hieronymi 2007). However, the coexistence of several system operators curbs economies of scale and requires a coordination of their activities in a centralized clearing house.

An unambiguous impact on producer's compliance costs results from mandated financial guarantees and the distinction between historical, orphaned and new WEEE. The usefulness of these stipulations deserves a closer assessment. The directive foresees a distinction of historical WEEE and new WEEE; the latter defined as WEEE from products put on the market after the 13th August 2005. The treatment of historical WEEE shall be financed by producers with collective systems and producers are supposed to "contribute to these systems according to their current market shares" (Article 8, COM 2003). Furthermore, producers are supposed to provide recycling guarantees or recycling insurances for new WEEE. These are supposed to cover the product's recycling costs at end-of-life or to finance the recycling of an orphaned product if a producer goes out of business.

A lot of recyclers apply throughput-oriented shredding strategies and minimize tracking and identification efforts. An identification of producers does not serve any purpose for the recycler because new WEEE is not differently processed than historic WEEE. Accordingly, the impact direction of the WEEE directive has been chosen in an unfortunate way. It is not mandating individual take-back that spurs individual treatment and product design for end-o-life but mandating individual treatment that spurs individual take-back. Given the considerable efforts necessary to identify a certain producer's WEEE, the recycler is highly likely to process any WEEE under the declaration of historic WEEE. Under current economic conditions, such practice pays for itself in several WEEE segments if high capacity utilization is achieved. Recyclers will insofar pull WEEE into their plant regardless of who the producer might be. The transition between new and historic WEEE is accordingly not going to be tracked in practice. This allows for a first insight:

Any EEE producer is extremely unlikely to see his individual product occurring as WEEE (whose treatment needs to be financed) in the future. High efforts connected to the identification of products and the economics of recycling turn this case into a theoretical construct. If individual responsibility was mandated, producers are highly likely to provide evidence of a take-back structure without any WEEE amounts in take-back and accordingly without any financial obligation. A policy-maker should be aware of this.

4.2.6 Fair Allocation of Burdens

Current EPR systems are driven by the idea that the determination of a fair burden allocation among producers requires to analyze the type and origin of the occuring waste as to allocate the costs to the respective producer. Yet, the idea of fairness in the burden allocation is not very well served in this context. Analogous to the principles in tax law, a fair burden allocation among producers could be based on either an ability-to-pay principle or an environmental benefits principle. However, waste occurring in a system/country does not have much in common with the profits (ability-to-pay principle) or the environmental burden (environmental benefits principle) a producer generates via the sale of his products. A critical review of the “relative share approach” is apposite. Several aspects distort the picture. First, some products are more likely to end up in municipal waste bins (especially smaller devices) or are directly exported (Klatt 2001, Tasaki et al. 2005). Furthermore, the amount of WEEE in the official system collection bins is dependant on arbitrary consumer behavior and storage effects in households. This gives rise to considerable cross-subsidization effects between product categories according to their occurrence at dedicated official WEEE collection sites. Furthermore, producers face disincentives to inform consumers about WEEE take-back under the relative share approach. The less WEEE that is officially collected, the less is the resulting monetary obligation for producers – which means that producers do not have incentives to advertise and promote WEEE recycling.

A fair burden allocation among stakeholders is reflected in a balanced accounting for the interests of the public sector/municipalities and the private sector/producers. The allocation of the responsibility for collection (municipalities or producers) is an example for a contentious issue in this regard. The perception of fairness on these grounds is a question of political and societal values and can not be evaluated in terms of effectiveness or efficiency. Therefore, it is not further analyzed in this thesis.

4.3 Key Learnings for Alternative EPR Approaches

The analysis of current policy-making and implementation practice allows deriving recommendations for alternative policy approaches. The following key learnings from policy evaluation can be transferred into requirements for alternative EPR system designs.

- A comprehensive high quality recycling is compromised by economic disincentives for depollution. Command & control approaches are not capable to tackle this issue economically due to high monitoring and enforcement costs. An alternative system design should refocus on depollution quality and ideally include built-in economic incentives.
- Market forces spur recycling of some materials whereas others need additional incentives to be recycled. Focused incentives for certain eco-efficient reapplication strategies of materials are an effective way to boost recycling rates. A general recycling rate target without a clearly defined methodology for its determination is powerless.
- Input-based remuneration mechanisms (gate fees) do not spur or reward high quality recycling but solely reflect a “get rid of waste”-perspective. An alternative system design should align a producer’s financial obligations with the quality of recycling.
- Decentralized quality responsibility can deteriorate industry-wide quality standards considerably. Quality assurance is only possible with know-how. Centralized authorities can build up such competences and leverage economies of scale in monitoring and enforcement.
- Producer responsibility should not end at the harbor. Low collection targets and frequent illegal exports compromise the performance in closing material loops. Regulation should set a clear overall target and enable involved players to optimize the interplay of different levers to achieve the target. In the case of WEEE recycling, such a policy is likely to encourage higher collection rates and to discourage illegal exports.
- Current systems do not reward producers for good design because compliance costs are not aligned with design characteristics. Instead, they are often measured on a general per unit basis or via relative market shares. However, costs for sound treatment are lower for specific product designs, e.g. if equipment can easily be depolluted. The recycling process benefits from such advantages in practice. Indeed, substantially different depollution times of individual devices in the same product

category can occur at recycling facilities. However, tracking these advantages in practice involves substantial additional costs that outweigh those benefits. Therefore, EPR systems should directly differentiate a producer's burden according to the design of new products and minimize those sorting or identification efforts in practice that bear no economic or environmental benefits (Bohr et al. 2007a).

- The definition of targets and respective monitoring should be aligned with practical collection and treatment streams. Product categories in the WEEE directive should be reclassified.
- A centralized allocation mechanism by a clearing house as applied in Germany is highly inefficient. A liberal approach with decentralized decision-making and direct business relationships between various players is likely to be more effective in achieving EPR's goals. Competition can and should be spurred with different instruments.
- Producers are neither interested nor proficient in waste management. EPR systems should liberate producers from waste management and focus on the design feedback and appropriate financing of end-of-life treatment.
- Product recycling guarantees are a very awkward instrument to set design incentives and also represent a considerable burden in the balance sheet of a producer. Furthermore, a closer analysis reveals that individual guarantees lack of a substantiated economic basis. A different interpretation of the producer's obligation can set stronger design incentives and allow for some sort of generational contract that enables authorities to forgo burdensome producer guarantees.

5 Alternative EPR Approaches

Free market tools provide promising alternatives for a regulation of electronics recycling. Several scientists have been advocating for a broader use of such tools to tackle environmental issues (Housman 1994, Costanza et al. 1997). This is in line with a strong emphasis on internalizing externalities in a cost-efficient way (Coggins et al. 1994). Especially certificate markets have been subject to vigorous academic interest, but their suitability for electronics recycling has not been demonstrated yet.

Tradable permits/certificates have been analyzed as policy instruments from the 1960s on. Based on early works (Coase 1960, Demsetz 1964, Dales 1968 and Crocker 1966), it has been formally proven (Montgomery 1972) that no alternative regulatory scheme can achieve a given environmental standard at a lower cost than a permit-trading scheme. Since then, tradable permits have been used in the case of SO2 emissions, water emissions, municipal solid waste recycling credits, fishery quotas, and land development rights (Boyd et al.2003). To date, there is little experience in the application of certificate markets in the area of solid waste recycling. The only practical implementation analyzed in literature is the application of tradable permits for packaging waste recycling in the United Kingdom (Salmons 2002). Drawing on this experience, a generic report has been prepared for the EU commission analyzing the possibility of using a similar system for WEEE recovery (ERM 1999). Such certificate markets must, however, be carefully designed for the specific characteristics of the system and mechanisms to balance a respective certificate market are not obvious (Elmer et al. 2005). Awareness of the manifold design options of these instruments is still not widespread among decision-makers and stakeholders (Salmons 2002). Accordingly, the tendency to reject policy initiatives based on a stereotypical notion of certificate markets compromises the implementation of such policy tools in practice (Godard 2002a, Godard 2002b).

Electronics recycling systems are complex. A direct transfer of a policy designed for packaging waste into a policy tool to shape electronics recycling systems is inappropriate. This chapter will familiarize the reader with incentive-oriented certificate-based regulation variants that were specifically designed for WEEE recycling. First, material recovery certificates will be introduced as a conceptual basis. Afterwards, system variants with tradable material recovery certificates will be presented and contrasted with the characteristics of existing regulatory approaches.

5.1 The Concept of Material Recovery Certificates

5.1.1 The Role of MRCs in the Waste Management Sphere

Waste management is focused on finding suitable disposal options for wastes. This management focus is also reflected in payment mechanisms. If a suitable disposal option can be offered, the offerer usually receives mass-based gate fees (input-based remuneration) for wastes entering the respective disposal option. WEEE management works similar in current EPR systems.

In the context of industrial ecology, the waste management focus has been supplemented by additional goals such as the quality of a recycling process (Richards et al. 1994). Important process quality measures are depollution quality level, recycling rate, and the level of reapplication of materials. Such quality measures are mainly determined by the outputs of a recycling process. Several regulatory appraoches have adressed these measures with respective targets and the definition of suitable disposal options. Yet, this hasn't changed payment mechanisms. However, the anchorage point of financial flows is a crucial aspect for economic incentive structures in a system. As the analysis in chapter 4 revealed, gate fees are not a suitable instrument to set focused incentives in order to boost recycling rates or depollution quality levels.

5.1.1.1 Functioning and Determination of MRCs

Material recovery certificates (MRCs) are driven by the idea that the actual recycling service is characterized by the processing level/ treatment depth of a recycler (how much of the incoming material is fed back into the loop and which level of reapplication) and by the depollution performance (how many hazardous wastes are diverted from their reentry into ecosystems). MRCs are based on outputs produced and serve as recycling service remunerations instead of gate fees. A recycler uses material recovery certificates to refinance his operations and he issues (and sells) MRCs for a certain amount of generated secondary material fractions. A functional MRC system requires

- a comprehensive classification of fractions occurring in the secondary materials industry,
- a definition of anchorage points in the secondary materials system,
- scoring factors for fractions to set focused economic incentives and to allow for an aggregation into a unified "currency",

- and a definition of market segments for incoming WEEE.

Expertise for a suitable classification of secondary materials has been developed in systems with centralized producer responsibility organizations. Experiences of different system operators have ended up in a unified reporting tool developed by the WEEE forum (Gabriel 2006). The reporting tool features a fraction classification based on the European Waste Catalogue (EWC) which was refined for the purposes of reporting and monitoring in the area of WEEE recycling. A subset k of fractions f_k from the reporting tool is used as a classification basis in this thesis.

The determination of MRC anchorage points in the secondary material system and the quantification of scoring factors are driven by the following considerations: Where can materials easily be traced in the system and from which point can be ensured that materials enter the intended final technology? Where do established and economically attractive reapplication strategies exist (no need to anchor MRCs) and where are additional incentives necessary to make recyclers recover materials in an environmentally sound or favorable way? The material flows via certain anchorage points are to be tracked by recyclers in order to be able to issue MRCs. The amount of MRCs per fraction is determined by the scoring factor sf_k and the mass $mf_{i,k}$ of the fraction. Segment-specific MRCs can be issued for each segment i; however, the scoring factors per fraction should be identical in order to obviate and discourage cheating and abuse. The determination of the MRC supply $mrcs_i$ per ton input is portrayed in Figure 31:

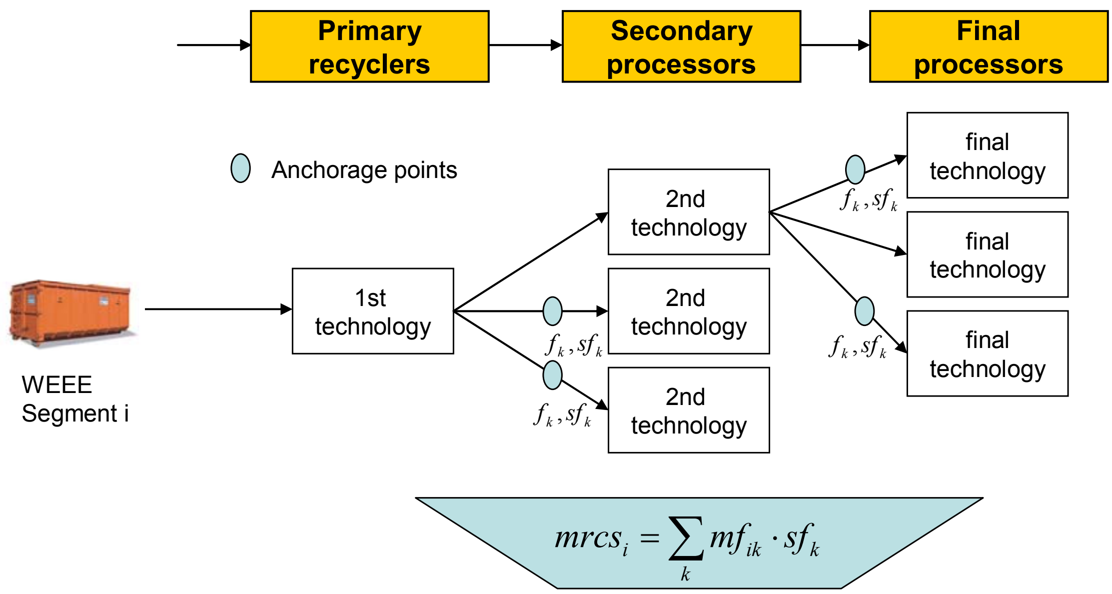

Figure 31: The determination of MRCs

Ideally, the scoring factors are chosen in a way that monitoring is feasible and appropriately focused in practice, thorough depollution and recovery of hazardous components is stimulated, and high recycling rates are stimulated. The assessment of the pricing situation of secondary material fractions during the survey revealed that market-driven incentives exist to recover some fractions in the highest quality possible while the sound recovery of other materials is discouraged by market conditions. The purpose of the weighting factors is not to provide an environmental weighting of fractions although such a setup can be chosen if a sound environmental assessment framework is available. Instead, the proposed weighting factors are designed to set overall economic incentives for recyclers in a way that a profit maximization strategy will lead a recycler to sound recycling and the adherence to depollution and recovery standards as defined by the WEEE directive. With perfect information on all costs and revenues of a recycler, the scoring factors sf_k could be determined according to equation (18). The rationale underlying equation (18) is that the relation of individually allocable costs of a fraction with an anchorage point ($sf_k \neq 0$) to the overall depollution and disposal costs equals the contributional share of the fraction on material recovery certificates. For a set K' of fractions with $sf_{k'} \neq 0$, fraction price fp_k, and output mass share of fraction k in segment i $fms_{i,k}$, this translates into

$$\frac{fms_{i,k} \cdot fp_k + adc_{i,k}}{\sum_{k'} fms_{i,k} \cdot fp_k + \sum_{k'} adc_{i,k}} = \frac{fms_{i,k} \cdot sf_k}{\sum_{k'} fms_{i,k} \cdot sf_k} \quad (18a)$$

$$\Leftrightarrow sf_k = \frac{fms_{i,k} \cdot fp_k + adc_{i,k}}{\sum_{k'} fms_{i,k} \cdot fp_k + \sum_{k'} adc_{i,k}} \cdot \frac{\sum_{k'} fms_{i,k} \cdot sf_k}{fms_{i,k}} \quad (18b)$$

5.1.1.2 The Impact of MRCs on Depollution Quality

The application of material recovery certificates as a remuneration mechanism is likely to have substantial impact on depollution quality levels in a system. Under an MRC system, recyclers obtain recycling service remunerations based on quality-defined outputs produced. Thus, the recycling service remunerations are relevant in a recycler's decision-making context because they depend on *dql*. Scoring factors for hazardous material fractions are likely to be high due to their high disposal and depollution costs. Their share on recycling service remunerations has to be accounted for when deciding on an optimal *dql*. Recycling service remunerations increase linearly with *dql* for a given scoring factor. Figure 32 shows the decision framework from Figure 25 with overall relevant costs (as before) and the new decision-relevant revenues from MRCs. The sum of both, the decision-relevant profit, is now maximal with a much higher *dql*.

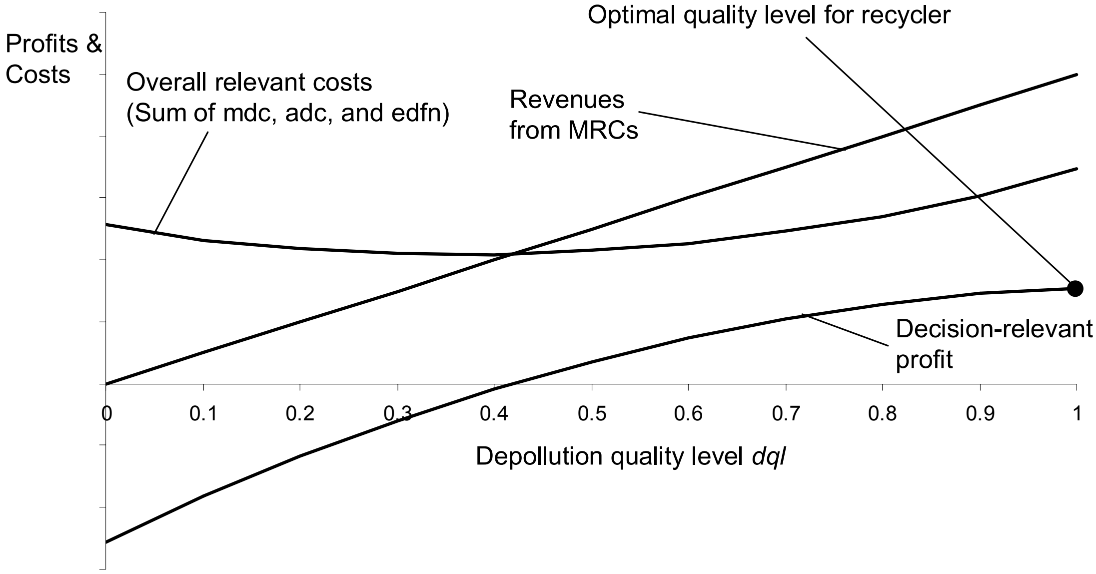

Figure 32: MRCs and depollution quality

Equations (12) and (13) are amended accordingly:

$$DRP = \left(rsr_i(dql_i) + smr_i\right) - \left(daic_i + adc_i(dql_i) + ofc_i + vco_i + mdc_i(dql_i) + edfn(pncn, dql_i)\right) \quad (19)$$

$$\frac{\partial DRP}{\partial dql_i} = 0 = \left(\frac{\partial rsr_i(dql_i)}{\partial dql_i}\right) + \left(\frac{\partial adc_i(dql_i)}{\partial dql_i}\right) + \left(\frac{\partial mdc_i(dql_i)}{\partial dql_i}\right) + \left(\frac{\partial edfn(pncn, dql_i)}{\partial dql_i}\right) \quad (20)$$

5.1.1.3 The Impact of MRCs on Illegal Export

A recycler also faces a different decision-making context when it comes to the utilization strategies for different types of WEEE. The payment basis on outputs now accounts for a decrease of the profitability of illegal exports because no recycling service remunerations are obtained for WEEE that is not treated for material recovery. In principle, the reuse profitability is decreased in the same way; however, scoring factors could as well be defined for reuse in order to keep such utilization attractive. Figure 33 shows the decision-making framework from Figure 28 and features profitability curves of illegal exports and reuse that are shifted considerably downwards compared to their original position due to missing recycling service remunerations.

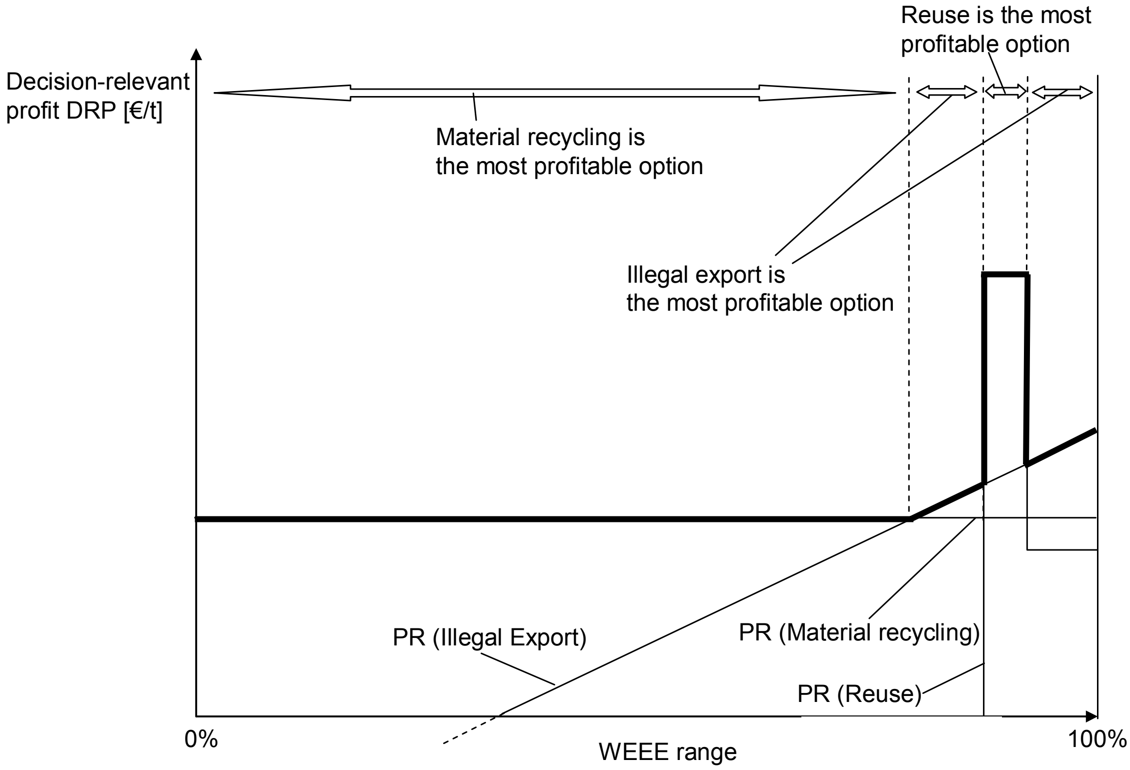

Figure 33: MRCs and utilization strategies

As a result, the share of material recycling is increased whereas the share of illegaly exported WEEE is decreased if recyclers choose the optimal utilization strategy. As Figure 33 reveals, the use of MRCs as a payment mechanism is an effective way to ensure an actual treatment and to reduce illegal shipments of obsolete electronics that contain hazardous materials. If mere WEEE trading pays off and the political willingness is not given to ban such shipments within a system, such practice certainly doesn't need nor deserve an additional subsidization as it can be the case under EPR systems with gate fees.

5.1.2 The Role of MRCs in the Producer Sphere

On the one hand, material recovery certificates represent a financing and incentive mechanism for the actions of a recycler in the waste management sphere. On the other hand, MRCs are a producer's means to demonstrate EPR compliance and MRCs can here be used as an incentive mechanism as well. The currently prevailing gate fees paid by producers in EPR systems are substituted with payments for certificates in the MRC system. Thus, the EPR burden for each EEE producer is mainly determined by the amount of MRCs a producer is supposed to acquire or finance under an EPR approach. In order to continuously monitor the fulfillment of responsibilities, all EPR approaches work with a defined accounting period t. A typcial length for such an accounting period would be 6 months or a year.

Several methods are applied under current EPR systems to allocate financial responsibilities to a producer. In Germany, a non-transparent mechanism reputedly allocates responsibilities on the basis of current relative market shares. The obligation is restricted to officially collected WEEE and illegal exports are insofar not covered by producer responsibility. These and other drawbacks of the “relative share approach” have been described in the previous chapter.

Centralized producer responsibility organizations (PROs) usually work with a fee per kg or unit put on the market. This fee is either paid in the form of an advanced disposal fee or with direct invoices to producers. Yet, although not obvious at first sight, this is a relative share approach, too. The fee is continuously adjusted over time according to the amounts of officially collected WEEE in the system. As the experience in countries like Norway and Sweden shows, this approach must not impede a sound WEEE management and high amounts of officially collected WEEE. However, considerably lower officially collected amounts of WEEE in other countries evoke doubts about the effectiveness of relative share approaches.

5.1.2.1 Static Steering and Absolute Obligations

Material recovery certificates allow for different variants in allocating financial burdens. A simple version directly connects the amount of MRCs to be acquired with the amount of units sold or the mass put on the market. A static steering lever s_i determined by a clearing house serves to define an absolute amount of MRCs to be acquired by a producer per kg or unit put on the market in each accounting period t. If obligations are separately defined in each segment i, an individual producer’s obligation $iomrc_{i,t}$ is defined according to (21):

$$iomrc_{i,t} = ne_{i,t} \cdot s_i \qquad (21)$$

Equation (21) defines a fixed, absolute obligation to recover a certain amount of materials from WEEE. A skillful determination of the static steering lever requires respective know-how as well as a political/societal compromise on a reasonable and just MRC target level. An absolute obligation set on a reasonable level is likely to discourage illegal and untraceable WEEE exports. The producer obligation to acquire a certain amount of MRCs allows pulling respective amounts of WEEE into controlled utilization strategies. Compliant recyclers with high quality utilization processes are endowed with a stronger WEEE acquisition budget on the collector market due to the refinancing mechanism via MRCs.

The logic behind the absolute mechanism is consistent with industrial ecology theory and environmental goals: The consumption of virgin materials and the creation of potential hazardous flows in ecosystems (production of new EEE) is met with a corresponding obligation to feed materials back into the loop and to fetch back hazardous materials from the cycle economy. However, such an approach is likely to spur intense debate and fights about the target. Furthermore, temporary MRC shortages on the supply side (waste management sphere) can give rise to volatile certificate prices. Such phenomena can be controlled with different instruments under an MRC approach: First, banking and borrowing of certificates allows for a shifting of burdens among periods. Second, MRCs could be generated via project-based mechanisms outside the underlying economy in order to bridge supply-side shortages. Such projects would allow producers to issue additional MRCs, a mechanism similar to the clean development mechanism under the Kyoto protocol. A consequent interpretation of a producer's responsibility would comprise such projects since it should cover all WEEE that occurs in an economy, including illegally exported WEEE.

5.1.2.2 Dynamic Steering and Relative Obligations

As opposed to a static steering of obligations, a dynamic steering allows to balance the MRC demand according to MRC supply in the underlying economy. A dynamic steering can be interpreted as a transfer of the relative share approach into a system with MRCs. The dynamic steering lever $s_{i,t}$ serves to convert the amount of MRCs generated in the waste management sphere or purchased by the PRO into individual financial obligations or contributions of single producers. A central clearing house or PRO continuously monitors MRCs generated in the waste management sphere and adjusts the prognosis for the future generation of MRCs and the dynamic steering lever accordingly. Several mechanisms can be applied in order to discourage speculation and to spur the closing of MRC deals between producers and recyclers. First, banking and borrowing options allow for a smoothing of temporary supply-side shortages or a surplus of MRCs. However, such mechanisms must not always be effective in discouraging speculation. Alternatively, the central clearing house can provide a price corridor which defines the worst case scenario for each market participant. In order to do so, the clearing house offers bid and ask prices for MRCs similar to the role of a central bank. Recyclers are allowed to sell certificates to the clearing house for the bid price ($MRCb_{i,t}$) at any time. Similarly, producers can acquire certificates from the clearing house for the ask price ($MRCa_{i,t}$) at any time. Banking and borrowing is only allowed if a temporary supply side or demand side market saturation holds (no more deals possible). In that case, the central authority is likely to adjust the dynamic steering lever accordingly. Speculation with MRCs does not pay off under such a setup.

5.1.2.3 Integration of a Virtual Product Design Feedback

Economic incentives for environmentally favorable product design can only occur if a producer's individual financial burden depends on the design characteristics of his products. This is not the case in current EPR systems, and any other physical (individual or collective) responsibility for WEEE is not likely to achieve such incentives either. However, a direct differentiation of a producer's $iomrc_{i,t}$ under the MRC approach can provide such incentives. This is referred to as a "virtual product design feedback" because the differentiation is based on a simulation or forecast of a product's characteristics and not on the actual fate of a product during its use phase and at end-of-life.

An approach to distinguish producer obligations according to the end-of-life characteristics of products has been described in (Bohr et al. 2007a). Equation (22) accounts for such a distinction with a product-individual factor *RELP* that depicts the relative end-of-life performance, based on simulations of the end-of-life behavior of products in recycling processes. An average relative end-of-life performance *ARELP* can be determined for a range of EEE sold by a producer in a market segment i. The *ARELP* is used as an additional factor in the determination of the producer's obligations and can substantially shift the burden among producers in order to set incentives for product design.

$$iomrc_i = ne_{i,t} \cdot s_{i,(t)} \cdot ARELP_i \quad (22)$$

Some researchers advocate for a stronger focus on life cycle design characteristics (Huisman 2006). In principle, a similar adjustment of individual obligations is possible according to life cycle design characteristics of a product. Furthermore, the REACH stipulations (COM 2006b) could be used to anchor individually adjusted EPR obligations according to the content of hazardous materials in new products. This would discourage the use of such substances and encourage the application of alternative materials where possible.

Defining concrete methods for the environmental evaluation of products would go beyond the scope of this thesis. Several methods are described in literature (Herrmann et al. 2002, Herrmann 2003, Ohlendorf et al. 2003, Mueller et al. 2003, Rifer et al. 2003, Herrmann et al. 2006) and the benefits of their application for a differentiation of producer obligations should be weighed against the efforts necessary to consistently do so. In order to ensure the political feasibility of such an approach, producers should be given the opportunity to voluntarily demonstrate environmentally superior properties of their products. This allows manufacturers with environmentally conscious product design to benefit from their efforts while others are given the opportunity to restrict their reporting requirements to data on sales and weight. In the latter case, a low product performance benchmark could be used to assign financial responsibilities to manufacturers that do not provide product information.

5.2 A Market with Tradable Material Recovery Certificates

5.2.1 Supporting Rationales for a Market Solution

MRCs could be used as a payment mechanism in direct contractual relations between recyclers and collectors or end-of-life managers. Alternatively, a market with tradable MRCs could be created in order to provide a platform for producers and recyclers to exchange MRCs. The general characteristics of input-based tradable certificates for WEEE recycling have been analyzed in (ERM 1999). Output-based material recovery certificates can be traded on a similar market.

One basic argument for the creation of environmental markets with tradable certificates is efficiency gains from trading (Baumol 1988). In a baseline-and-credit approach, the underlying rationale would be that a market participant who can generate (additional) credits with lower costs than another market participant can sell such credits to the latter for a price that lies between his costs and the potential costs of the other. The more heterogenous the capabilities of market participants in generating credits is, the higher the efficiency gains from trading should be. Evidence of gains from trading can be witnessed in a number of cases where tradable permit schemes were applied (Hahn et al. 1989, Tietenberg 1990). However, trading of material recovery certificates would occur between recyclers and producers, not between operators of industrial installations as is the case in emissions trading systems (ERM 1999). While it is reasonable to assume some heterogeneity among recyclers in their capability to generate material recovery certificates, trading gains are less likely to be of considerable magnitude under such a market because of similar technologies applied in the generation of MRCs.

Another argument in favor of tradable certificate markets is, however, increased competition which is likely to decrease prices for MRCs. Some researchers see the main advantage of a trading system in its stimulative effects on competition (ERM 1999). Yet, competitive EPR systems based on other means already exist in practice. However, transaction costs incurred under a competitive system are of considerable importance, too (Tietenberg et al. 2004). As will be argued in the evaluation section, a specifically designed market leads to considerably lower transaction costs incurred to ensure legal compliance if compared with other systems. This especially holds true if current competitive systems such as the German central allocation approach are used as counterfactuals in the evaluation.

5.2.2 Functioning of the Market

The setup of an MRC market approach recommended for WEEE is displayed in Figure 34.

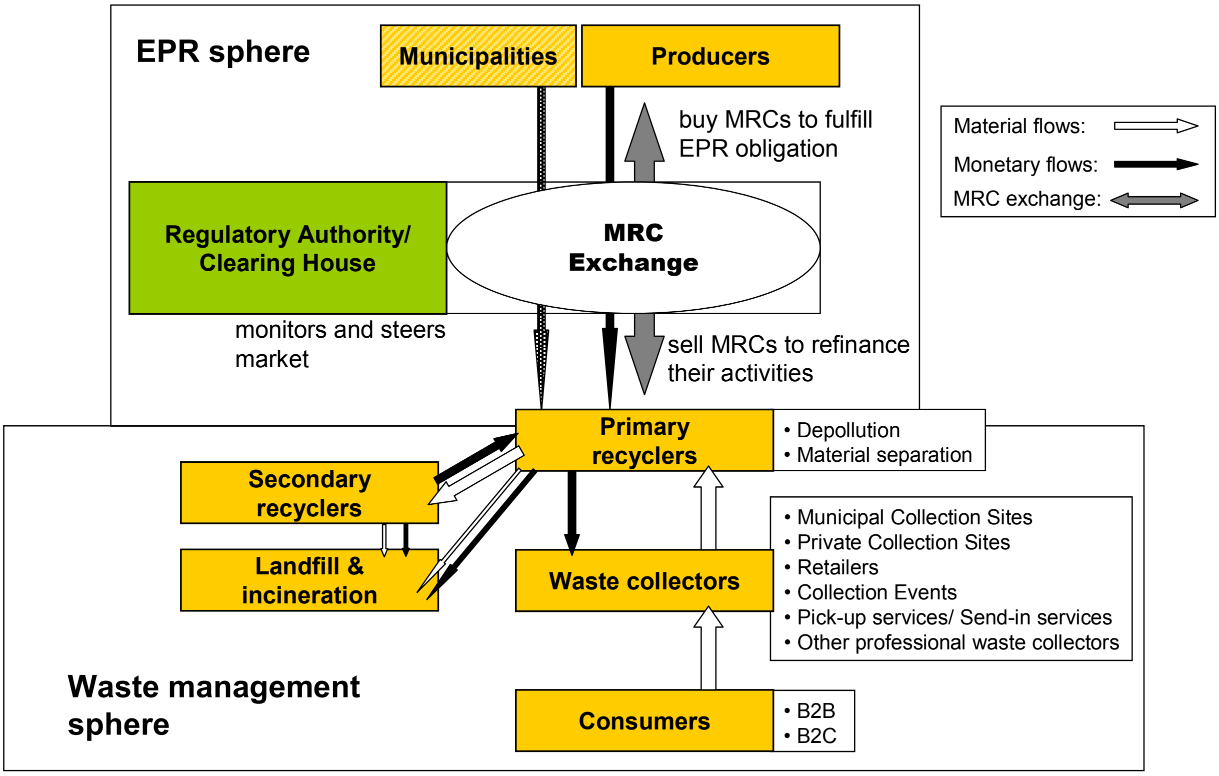

Figure 34: Material and monetary flows under a market with tradable MRCs

An EPR sphere and a waste management sphere are to be distinguished. In the waste management sphere, recyclers proactively work the waste collector market. They can either establish direct business relationships with existing collectors or create additional new collection infrastructure under their own control. Recyclers pay waste collectors for their service and negotiate how WEEE should be provided, which collection bins to use, how to organize the logistics, payment per kg collected, and other specifications. Collectors allow consumers to return WEEE free of charge. Furthermore, collectors can work with return incentives or induce the use of their take-back infrastructure by the consumer with additional advertising or other means. The WEEE transport from collector to recycler is also financed by the recycler. After treatment, the recycler markets his secondary materials and sends them on for further processing. The producer is not involved in any activities in the waste management sphere.

Recyclers cover all expenses for collection services, transport, and treatment with the payment obtained via the sale of MRCs in the EPR sphere. In order to receive such payment, recyclers compete with their MRC offers on the MRC market. Producers act as buyers on the MRC market, assuming their extended responsibility by fulfilling the obligation to acquire a certain amount of material recovery certificates (MRCs). Their obligation is based on the amount (and properties) of new products "put on the market", the latter being interpreted as total amount of products sold regardless of the end user. The market is segmented into 6 categories according to recycling treatment flows in practice as described in chapter 2. The balancing of supply and demand is steered by a regulatory authority or a clearing house. Both static and dynamic steering as described in chapter 5.1 is possible.

Municipalities can also be required to buy certificates on the market in order to share the burden with producers to some extent. This has the following background: Producers often oppose the internalization of collection costs in the EPR system as is the case, for example, in the German approach. In this regard, producers argue that waste collection is a municipal service task and related costs are to be borne by municipalities. The extent of shared responsibility is a question of political and societal values. Shared responsibilities can be incorporated in an MRC market as well and the problem of how to deal with the spatial allocation of producer responsibilities can be tackled at the same time. The latter refers to the anxiety that in a liberal market, producers preferably work with recyclers that target areas with a high WEEE potential (high population density). Municipal collection points in remote locations on the countryside could struggle to find a service partner under such a setup. An MRC market can balance these aspects in the following way: Instead of sharing costs according to value chain levels (Municipalities: collection costs, Producers: transport and treatment costs), costs are split over the obligations on the MRC market. Municipalities must acquire MRCs as well, (a certain percentage of the overall MRC supply) and their obligations should be determined according to their population size *and* density. If done so, low-density municipalities face a lower MRC obligation compared to muncipalities with same size but higher density. The reasoning is as follows: In the negotiations with recyclers, municipal collection points in rural areas would most likely achieve a lower collection fee per kg WEEE than collection sites that are preferably located. As a result, rural municipalities would probably face lower revenues for collection than urban municipalities; however, their obligation to acquire MRCs and the respective financial burden would be lower as well.

5.3 Strengths and Weaknesses of an MRC Market

The effects of the use of material recovery certificates on an appropriate treatment of hazardous materials, the performance in closing material loops, and the possibility to create design incentives were already adressed in chapter 5.1. The analysis and evaluation in this chapter focuses on the economic strenghts and weaknesses of an MRC market and its effectiveness compared to alternative EPR systems.

5.3.1 Market Segmentation, Monitoring, and Quality Assurance

The MRC market is segmented into 6 groups according to recycling treatment practice. This allows for an alignment of collection, treatment, and reporting and makes a consistent monitoring easier. Sorting activities that only occur due to accounting requirements under current systems are not necessary. A tracking of WEEE according to B2B and B2C sources or a sorting and identification of historical and new WEEE in waste streams is not necessary either. In addition, a continuous analysis of collection samples for the allocation of producer obligations becomes obsolete.

The assurance of quality standards is addressed with a risk-oriented approach that focuses on key fractions. Fractions whose environmentally sound recycling is ensured by market forces are not addressed with regulatory stipulations. The salient role of recycling rates and its undesired impacts on material flows is overridden while the responsibility of producers is refocused on depollution and “transition fractions”. “Transition fractions” include materials which are on the border to an economically viable material recycling. An anchorage point for MRCs allows turning a material recycling of these fractions into an ecomically viable activity. A centralized institution with respective knowhow is responsible to monitor the generation of key fractions. The recommended monitoring strategy refrains from sophisticated tracking systems that try to map material flows over the full value chain. Instead of tracing all parts of the value chain, WEEE sourcing and transport is left to recyclers and free market forces. The actual material flows of the recycler and the recycling site is, however, subject to an enhanced control and occasional on-site check-ups by control experts. A monitoring tool with detailed material flow reporting including inbound logistics, current mass in treatment, and outbound logistics for key fractions should be established for this purpose. In order to approve the issuance of MRCs, key material flows are reported and traced in a timely manner. The occasional on-site presence of control experts, plausibility tests based on batch experiments, the general requirement of a thorough output reporting under the MRC market, and sensitive penalties for abuse are, if consequently pursued, an effective lever towards sound treatment and accounting practice.

5.3.2 System Efficiency

As noted earlier, competitive structures are important to ensure low compliance costs. However, the system design chosen to create competition must be thoroughly assessed towards system-inherent inefficiencies that can cause transactions costs of a considerable magnitude. As observed under the competitive German approach, considerable communication and organization efforts occur with a lack of decentralized decision-making and an overregulation of material flows. The organizational properties of an MRC market allow avoiding several of these inefficiencies.

Under the MRC market, recyclers are endowed with the organizational responsibility for upstream collection, sorting and transport. Naturally, recyclers with several years of experience in waste management should be more capable to assume these organizational tasks than producers that do not have an interest in interfering with the waste management sphere. A direct business relationship of collector and recycler and the control of collection service remunerations on the one hand and competitive price pressure from the MRC market on the other hand induce a recycler to organize WEEE take-back in the most efficient way. Competition for efficient waste collection services is likely to occur. In order to ensure a certain take-back service level for consumers, legal mandates for municipal collection and retailer take-back can complement the recyclers' collection efforts. Decentralized decision-making is possible which allows leveraging grown-up regional structures with established logistic solutions and communication channels. Recyclers can influence their take-back partners to collect and sort WEEE according to their operational needs and quality requirements. Appropriate sorting can already yield productivity gains in a magnitude of 20-30%. Furthermore, communication and organization efforts for producers are minimized because they do neither face planning and coordination tasks nor physical responsibilities in the waste management sphere. This is especially relevant for smaller manufacturers from foreign countries which are liberated from an inefficient setup of "take-back power" in the respective country.

5.3.3 Treatment Performance

An MRC market sets focused and continuous incentives for a depollution and recycling performance improvement of a recycler. Fixed mandatory recycling rates do not prompt a recycler to go beyond the target. Minimal performance factors can be mandated to avoid "wasteful" mass recycling. To ensure this, simple aggregated figures derived from mass input/output monitoring on a facility level are sufficient.

5.3.4 Producer Responsibility Interpretation

Under the MRC market, a producer's responsibility is not to take care of future waste but to ensure that an amount of materials proportional to his current use of virgin material is currently fed back into material loops. Depending on which methods are used for a potential differentiation of producer burdens, this can also be interpreted as an obligation to recover an amount of hazardous materials from current WEEE that is proportional to the amount of hazardous materials used in a producer's current products. This EPR interpretation enables producers to refrain from providing guarantees or having to make provisions for future WEEE in the balance sheet which is a considerable economic burden. Furthermore, recycling insurances are obsolete.

5.3.5 Market Steering and Price Volatility

An MRC market with a static steering approach can lead to short-term economic distortions if the MRC target level is set too high or too low. The drifting down of prices in the European CO^2 market for the EUA 2007 period serves as an example where oversupply of certificates via grandfathering lead to a completely unbalanced market. The same effect would occur under an MRC market with a too low target level. Too high target levels are likely to spike prices because producers are not able to acquire a sufficient amount of certificates due to a lack of supply. A dynamic steering approach requires a knowledgeable clearing house which responds quickly to market observations in order to smoothly steer the MRC market. Uncertainties do arise from the volatility of material contents in WEEE market segments. The yield of key fractions can fluctuate temporally and spatially which complicates market steering in practice. Cheating and the occurrence of black markets for key fractions have to be discouraged with an effective monitoring system as described in the previous section. A sound and unbiased application of theoretically available instruments such as banking and borrowing and the price corridor is likely to achieve a balanced market; however, it must be expected that the concrete application of these steering instruments is likely to be under strong lobbying pressure which can impede the efficiency of the market steering.

6 Policy Simulation

This chapter features excerpts of a quantitative analysis of the impact of current command & control regulation and an MRC market regulation on WEEE recycling systems. Results from policy simulations in a defined environment for one WEEE segment (recycling of cooling and freezing appliances, segment i=3) will be shown and discussed. While the simulations reveal economy-wide costs and revenues and market prices for different scenarios, the focus of the analytical discussion in this chapter is on depollution quality levels and its impact on an economy-wide recovery of hazardous materials. The structure of the system model used for simulations will be presented first and its focus and limits will be explained. Afterwards, the constituents of the system model are illustrated in more detail and regulation designs to be contrasted and compared will be defined. The recycling process for cooling and freezing appliances will be explained and key areas of concern will be pointed out. This involves waste input characteristics, state-of-the-art recycling process technologies, and the secondary material flows for the relevant segment. Furthermore, simulation results of a case study for the German market will be presented and discussed, supported by a sensitivity analysis of key variables.

6.1 System Model Constituents

6.1.1 Model Structure

The impact of different regulatory designs on material flows and underlying economics is captured with a system model that consists of two interconnected parts: a material flow model and an economic model. Each model features several key variables whose development over time or dependency on the regulation can be subjected to a closer analysis. Both models require an input in the form of framework parameters or variables that are characteristic for the analyzed economy (see 6.1.3) or defined by the simulated regulation design (see 6.1.4). Furthermore, the simulation requires an exchange of model variables between material flow model and economic model. Environmental results such as the amount of hazardous materials recovered or the performance in closing material loops are derived from the material flow model. Economic results such as the financial flows and the overall macroeconomic costs to achieve these material flows are derived from the economic model. Figure 35 provides an overview of the modeling approach.

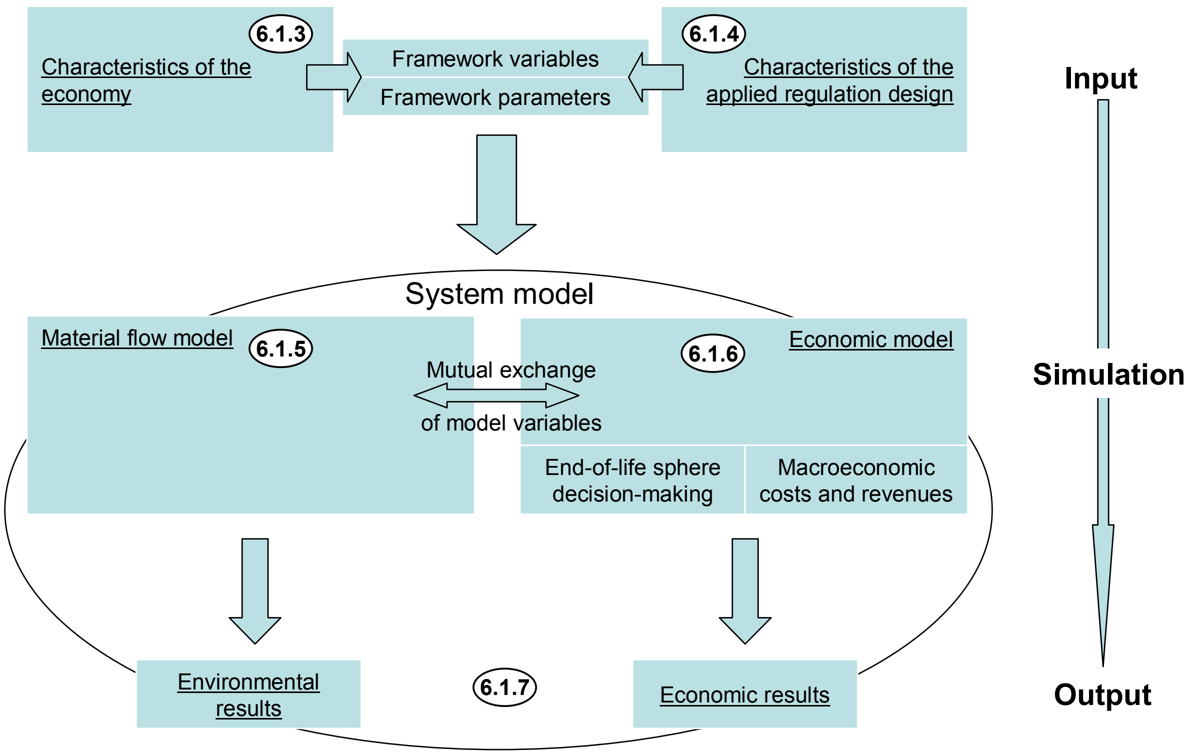

Figure 35: Model structure for policy simulation

6.1.2 Model Focus & Limits

The simulation of different regulation designs and its informational value is restricted and focused on the determination of certain key indicators. Several impacts of the differing regulation approaches are not captured in the model. The model

- is not capable to simulate dynamic effects from competition over time,
- does not reflect or quantify advantages from decentralized decision-making in collection and transport for different policy designs,
- does not reflect or quantify the level of transaction costs occurring under a regulation design,
- does not feature heterogeneous players in the end-of-life sphere,
- and does not include dynamic feedbacks in the simulation of material flows.

6.1.3 Characteristics of the Analyzed Economy

An analyzed economy is characterized by its size *ES* [capita], its areal spread *A* [km^2], an amount RF_i [units] of players with a defined archetype recycling facility in each WEEE segment i, a collection point density/frequency cpd_j [units/million capita] for each collection option j which indicates the collection infrastructure level, a share sco_j [-] that depicts the relative share of waste volumes taken back via each collection option j, a detour factor *v* [-] that indicates the road infrastructure level, a labor cost rate *lr* [€/h] reflecting the level of wages in the economy for low-skilled workforce in industrial operations, and industry-specific weighted average cost of capital *WACC* [-] asked for by recyclers, indicating the strength of competition in the economy. Economy-specific parameters used in the material flow model were already discussed and explained in chapter 2. Table 17 summarizes the characteristics of the analyzed economy and reveals framework parameters and variables and their application.

Depiction	Symbol	Unit	Characterization	Used in
Size of economy	ES	[capita]	Parameter	Mfm/Em
Areal Spread	A	[km^2]	Parameter	Em
Amount of recycling facilities	RF_i	[units]	Variable	Em
Collection point density/frequency	cpd_j	[t]	Parameter	Em
Share of collection option j	sco_j	[%]	Parameter	Em
Labor costs	lr	[€/h]	Parameter	Em
Detour factor	v	[%]	Parameter	Em
Industry-specific costs of capital	$WACC$	[%]	Variable	Em
New EEE	$ne_{i,t}$	[t/year]	Parameters	Mfm
Second hand export ratio	$sher_{i,t}$	[%]	Parameters	Mfm
Second hand import ratio	$shir_{i,t}$	[%]	Parameters	Mfm
EEE in use	$eu_{i,t}$	[t]	Initial parameter	Mfm
Storage ratio	$sr_{i,t}$	[%]	Parameters	Mfm
Average use span	$aus_{i,t}$	[years]	Parameters	Mfm
Average storage span	$ass_{i,t}$	[years]	Parameters	Mfm
Stored obsolete or broken EEE	$so_{i,t}$	[t]	Initial parameter	Mfm
Illegal dumping ratio	$idr_{i,t}$	[%]	Parameters	Mfm
Household bin ratio	$hwr_{i,t}$	[%]	Parameters	Mfm
Landfill and incineration ratio	$lar_{i,t}$	[%]	Parameters	Mfm
Rem., repair, reuse ratio	$rrr_{i,t}$	[%]	Parameters	Mfm

Table 17: Characteristics of the analyzed economy

Furthermore, several economy-specific parameters such as real estate prices, energy prices, financing costs, or country-specific taxes that are not explicitly revealed in detail underlie the cost and revenue structures of the defined archetype recycling facilities.

6.1.4 Characteristics of the Applied Regulation Design

Two regulation archetypes will be compared in the simulations: A typical command & control approach (C&C) as to be found in current European implementation practice and a regulation approach with tradable material recovery certificates (MRC).

The C&C approach features mandated recycling rates and depollution levels as defined in the WEEE directive and presented in chapter 4.1.1. Recycling service remunerations are paid for the amount of wastes treated (input-based service remunerations). The C&C approach is further characterized by efforts put into monitoring and enforcement in terms of quality and border control. These are reflected in a regulation-specific function for *edfn* and a defined level of *pncn* (quality control) as well as a regulation-specific function of *edfi* and a defined level of *pnci* (border control). The MRC approach is characterized by defined anchorage points in the secondary material system, a set k of scoring factors sf_k that define the amount of material recovery certificates to be generated per ton output, and, if applied, ask and bid prices $MRCa_{i,t} / MRCb_{i,t}$ of a clearing house. Table 18 summarizes the key regulation design inputs required for the policy simulation.

C&C Regulation Design	Symbol	Characterization
Monitoring and enforcement of quality (quality control effort)	edfn	Defined Function
	pncn	Parameter
Monitoring and enforcement of illegal exports (border control effort)	edfi	Defined Function
	pnci	Parameter
Mandated recycling rates	rec_i	Parameters
Mandated depollution quality levels	dql_i	Parameters
MRC Regulation Design	**Symbol**	**Characterization**
Scoring factors	sf_k	Parameters
MRC ask and bid price of clearing house	$MRCa / b_{i,t}$	Parameters

Table 18: Characteristics of the applied regulation design

6.1.5 Material Flow Model

The modeling of material flows has already been described in chapter 2. A stock-and-flow model implemented with the simulation software Vensim serves to illustrate the magnitude of material stocks and flows and their development over time. The model captures EEE from sales over the use phase until its end-of-life fate and allows analyzing a regulation's impact on key variables in electronics recycling systems. Figure 36 shows the default material flow model used for the simulations presented in this thesis. Depending on the segment under scrutiny, output fraction flows from material recycling are put in concrete terms and connected with steering parameters according to the output fraction mass shares in practice.

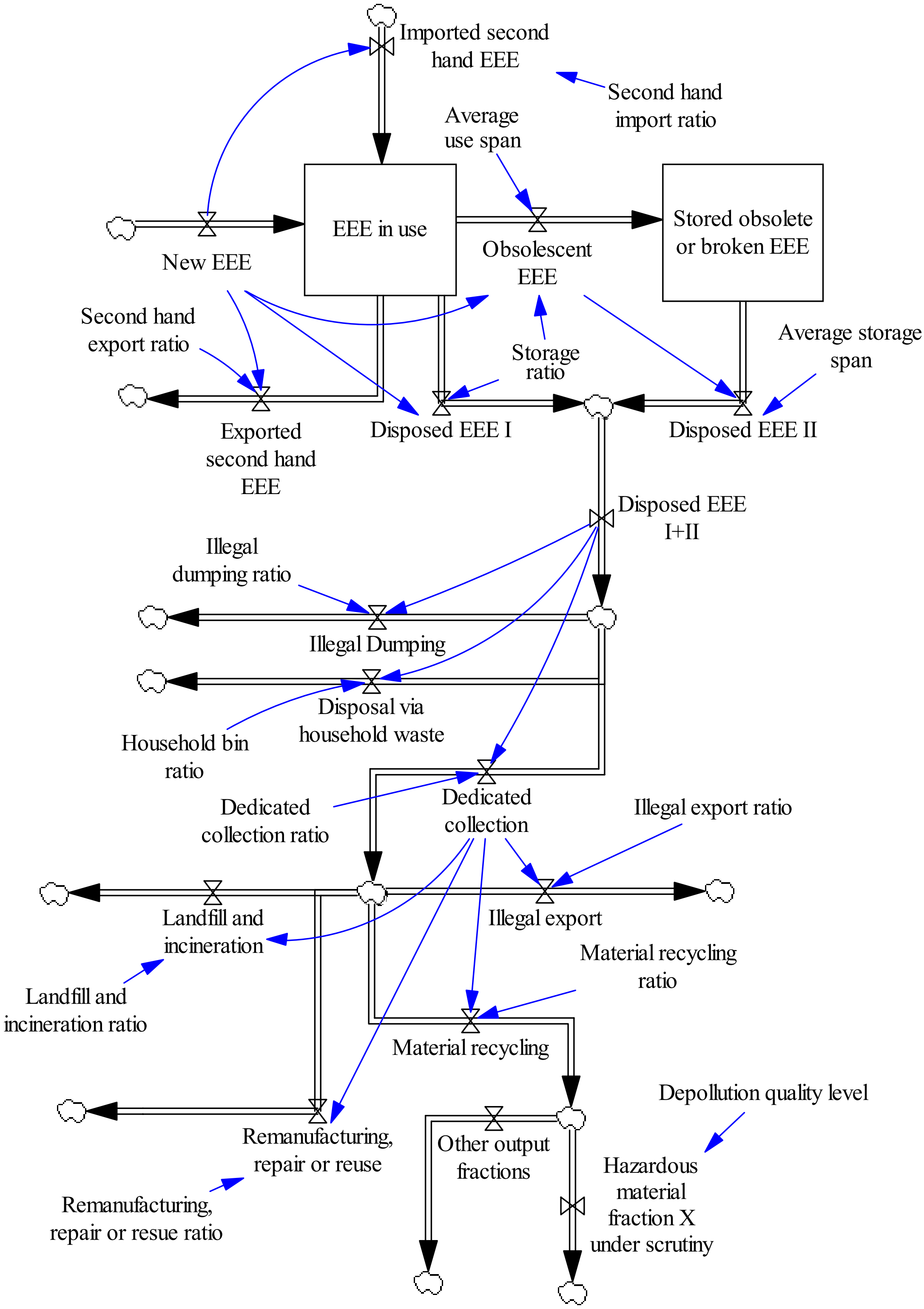

Figure 36: Material flow model for simulations

The model basically consists of stocks, flows, and auxiliary variables which steer or influence stocks and flows. Stocks are shown as boxes, flows are represented as thick arrows with valves, and auxiliary parameters or variables are identified by the thin arrows that connect them with other model constituents, indicating a functional relation. Their development over time is modeled in discrete time steps t. For the purposes of this thesis, the time step t is defined as one year, the starting time of simulations is 1980, and the ending time of simulations is 2020. The starting point of simulations was chosen as to allow for a steady state of flows in the time period of interest (2006-2020).

Each variable in the model is either internally or externally determined for each time step t. An internal determination is based on functional relations to other model variables whereas externally determined variables are set independently. The decision on an internal or external determination of variables is to be aligned with the modeling purpose. In this thesis, the area of interest is a regulation's impact on electronics recycling systems. In principle, regulation addresses several variables in the model and attempts to change them in order to steer material flows according to the goals underlying the regulation. Thus, one could model the impact of a regulation design on all (auxiliary) variables and parameters within a highly complex model structure. However, the analysis in chapter 4 revealed that the success of a regulation in achieving EPR's goals in current business environments in developed countries mainly depends on its impact on the depollution quality level and the illegal export ratio. Therefore, the modeling is focused on capturing the impact of regulation on these two auxiliary variables in an adequate way whereas other auxiliary parameters are determined independently (externally) according to researched data.

Table 19 provides an overview on variables and parameters and reveals which are internally or externally determined. Several of the auxiliary parameters are found among the characteristics of the analyzed economy. As explained in chapter 4, depollution quality level and illegal export ratio are subject to the profit-oriented decision-making of a recycler in the end-of-life sphere. These variables are insofar quasi-external in the material flow model since they are determined in the economic model.

Depiction	Symbol	Characterization	Internal	External	Further Information
New EEE	$ne_{i,t}$	Flow		X	Non-constant external parameters
Imported second hand EEE	$ie_{i,t}$	Flow	X		
Exported second hand EEE	$ee_{i,t}$	Flow	X		
Second hand export ratio	$sher_{i,t}$	Auxiliary parameter		X	Time-constant external parameters
Second hand import ratio	$shir_{i,t}$	Auxiliary parameter		X	Time-constant external parameters
EEE in use	$eu_{i,t}$	Stock	X	X	Initial values for t=0 (1980) set externally
Obsolescent EEE	$oe_{i,t}$	Flow	X		
Disposed EEE I	$deI_{i,t}$	Flow	X		
Disposed EEE II	$deII_{i,t}$	Flow	X		
Storage ratio	$sr_{i,t}$	Auxiliary parameter		X	Time-constant external parameters
Average use span	$aus_{i,t}$	Auxiliary parameter		X	Non-constant external parameter
Average storage span	$ass_{i,t}$	Auxiliary parameter		X	Time-constant external parameters
Stored obsolete or broken EEE	$so_{i,t}$	Stock	X	X	Initial values for t=0 (1980) set externally
Illegal dumping	$id_{i,t}$	Flow	X		
Illegal dumping ratio	$idr_{i,t}$	Auxiliary parameter		X	Time-constant external parameters
Disposal via household waste	$hw_{i,t}$	Flow	X		
Household bin ratio	$hwr_{i,t}$	Auxiliary parameter		X	Time-constant external parameters
Dedicated collection	$dc_{i,t}$	Flow	X		
Dedicated collection ratio	$dcr_{i,t}$	Auxiliary parameter		X	Residual of other auxiliary parameters
Illegal export	$ei_{i,t}$	Flow	X		
Illegal export ratio	$eir_{i,t}$	Auxiliary variable		X	Derivatives of economic model
Landfill and incineration	$la_{i,t}$	Flow	X		
Landfill and incineration ratio	$lar_{i,t}$	Auxiliary parameter		X	Time-constant external parameters
Remanufacturing, repair, reuse	$rr_{i,t}$	Flow	X		
Rem., repair, reuse ratio	$rrr_{i,t}$	Auxiliary parameter		X	Time-constant external parameters
Material recycling	$mr_{i,t}$	Flow	X		
Material recycling ratio	$mrr_{i,t}$	Auxiliary parameter	X		Residual of other auxiliary parameters
Depollution quality level	dql_i	Auxiliary variable		X	Derivatives of economic model

Table 19: Variables and parameters in the material flow model

6.1.6 Economic Model

The economics of the recycling value chain have already been analyzed in chapter 2.4, using activity-based costing as a means to determine costs and revenues associated with each activity. On the one hand, these costs and revenue structures serve to model a recycler's/end-of-life manager's decision-making and to determine concrete values for depollution quality level and illegal export ratio for a defined business environment. On the other hand, they allow drawing conclusions on economy-wide costs and revenues for a given set of economy-specific framework parameters.

The economic model is based on the assumption that all players involved as end-of-life managers/recyclers in the electronics recycling system have homogenous perceptions on framework conditions, apply equal treatment strategies as defined with an archetype recycling facility, are equally successful in obtaining recycling mandates, and are further confronted with equal cost and revenue structures and plant capacity utilization rates. These preconditions allow deriving average costs per ton and economy-wide costs for each identified activity.

The decision-making of an archetype recycler on the depollution quality level is characterized by the following equations:

$$\max!\ DRP = \left(rsr(dql) + smr\right) - \left(daic + dc(dql) + ofc + vco + mdc(dql) + edfn(pncn, dql)\right) \quad (23)$$

$$\Rightarrow \frac{\partial DRP}{\partial dql} = 0 = \left(\frac{\partial rsr(dql)}{\partial dql}\right) + \left(\frac{\partial dc(dql)}{\partial dql}\right) + \left(\frac{\partial mdc(dql)}{\partial dql}\right) + \left(\frac{\partial edfn(pncn, dql)}{\partial dql}\right) \quad (24)$$

The decision-making of an archetype recycler on his illegal export ratio is driven by the attempt to maximize profits via an optimized balance of *eir, mrr*, and *rrr*. This corresponds to a maximization of $\int_0^1 DRP(x)$ over the WEEE range x.

Collection, transport, sorting, depollution, and treatment can be identified as separate activities for a macroeconomic analysis. With the equations defined in chapter 2.4, average costs per ton can be assigned to each activity and economy-wide costs can be estimated with the material flow data from the material flow model for each segment i. Figure 37 provides an overview of relevant equations.

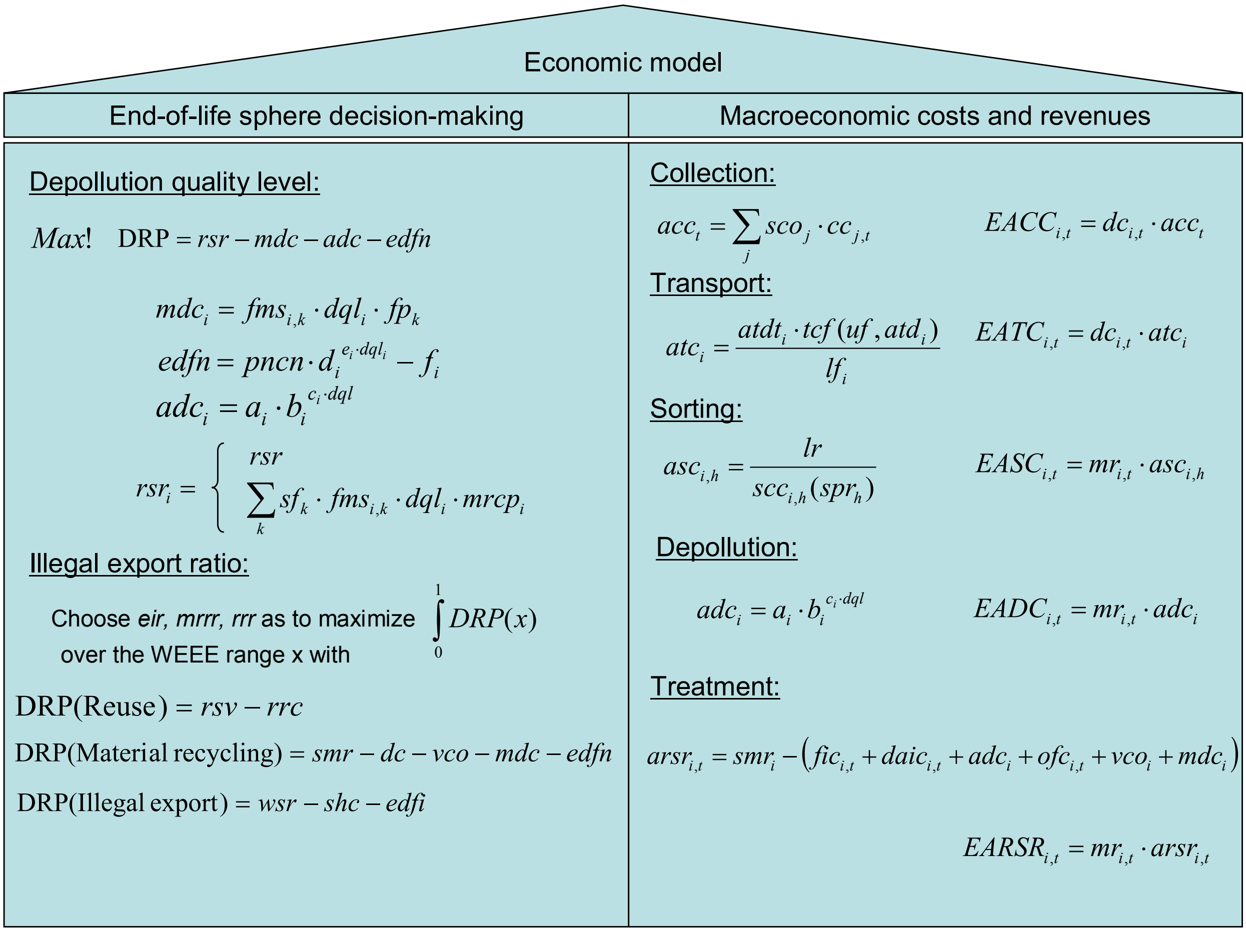

Figure 37: Overview of economic model equations

6.1.7 Environmental and Economic Results

Both models yield quantifiable results in terms of environmental and economic achievements. For the C&C approach, economic results are captured in the activity-related economy-wide costs $EACC_{i,t}$, $EATC_{i,t}$, $EASC_{i,t}$, $EADC_{i,t}$, and $EARSR_{i,t}$, and in the overall EPR-related costs $EEPRC_{i,t}$. Under the MRC market, several cost positions are reflected in the certificate prices $mrcp_{i,t}$ and the respective economy-wide MRC costs $EMRCP_{i,t}$. The indicator $bsd_{i,t}$ provides information on the apportioned absolute cost of EPR on each new unit sold. Environmental results are reflected in the performance in closing material loops *PCML* and in absolute amounts of recovered hazardous materials (fraction k) $ERHM_{k,i,t}$ in the analyzed economy. Results presented in this thesis focus on $ERHM_{k,i,t}$ which is calculated according to (25)

$$ERHM_{k,i,t} = mr_{i,t} \cdot ehmr_{i,k} \cdot dql_{i,k} \qquad (25)$$

6.2 Case Study: Cooling and Freezing Appliances in Germany

6.2.1 The German Economy

Germany's 82.300.000 million inhabitants live in approximately 38.7 million households on an area of 357.000 km^2 (Destatis 2007). As a mature economy, Germany features high household equipage saturation levels and a relatively stable development in unit sales in the cooling and freezing appliances segment. Consumer-initiated second hand import and export is of negligible importance. Figure 38 shows an estimate of cooling and freezing appliances sales tonnages in the German economy for the period 1980-2020 (own analysis, partly based on EAK Austria 2006, SENS 2006, SWICO 2006). A constant value of 170'000 tons is assumed as a prognosis for sales amounts in the period 2007-2020.

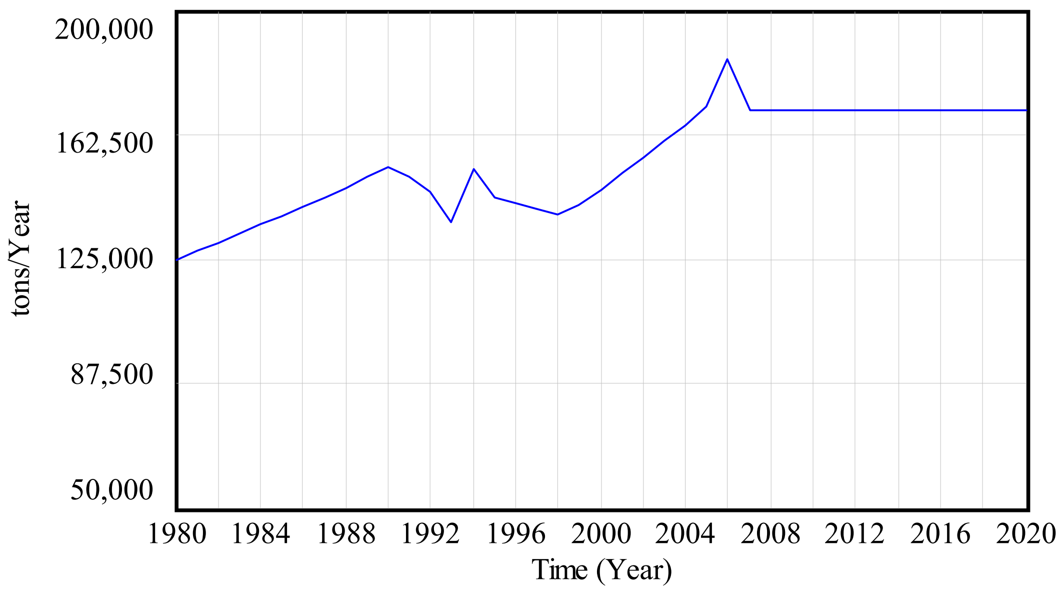

Figure 38: New EEE sales 1980-2020 (i=3)

In 2006, the average household equipage was 117.1% for refrigerators and 83% for freezing appliances (Destatis 2007). For the material flow simulation, the average use span of respective appliances was assumed to change over time between 11.4 and 15 years and the average storage span was assumed to be 5 years. A storage ratio of 0.2, a second hand import ratio of 0.01, and a second hand export ratio of 0.02 lead to Figure 39 that displays EEE in use simulations results for the period 1995-2020 and Figure 40 that displays disposed EEE I+II simulation results for the period 1995-2020. (own analysis, partly based on Huber 2006, Destatis 2007, Brüning 2007, Sander et al. 2006, SWICO 2006, EAR 2005, Lee et al. 2004)

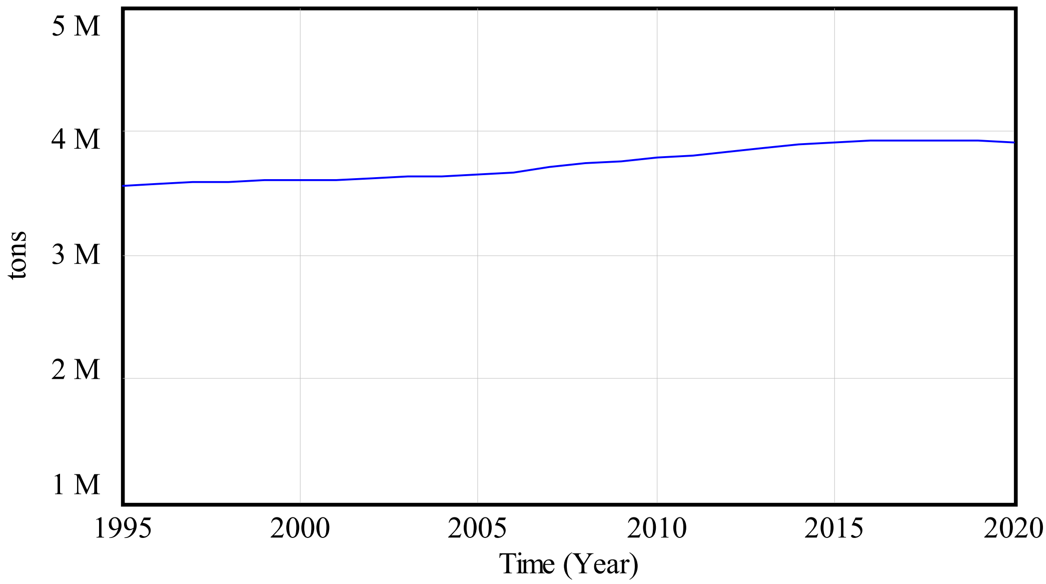

Figure 39: EEE in use for the period 1995-2020

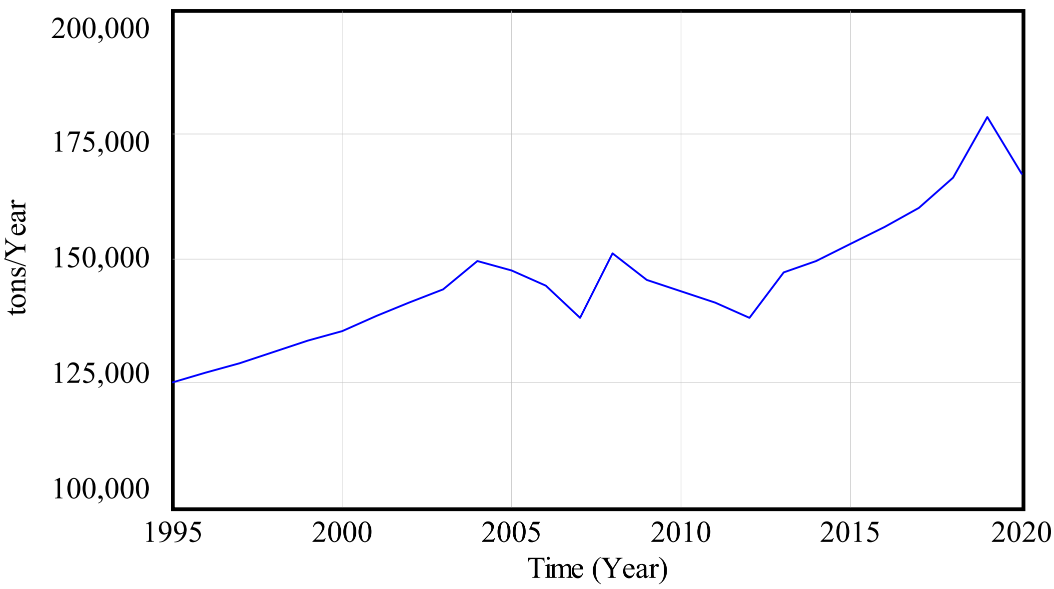

Figure 40: Disposed EEE I+II for the period 1995-2020

Almost no WEEE from segment 3 ends up being in the household waste or being illegally dumped. Dedicated collection therefore almost equals disposed EEE I+II. Repair and reuse are believed to account for 5% of the end-of-life fates and 0,01% are believed to be landfilled or incinerated accidentally. No data is available on the illegal export ratio and estimates of the variables and parameters underlying a recycler's decision-making on the illegal export ratio in

Germany yield a span of 0-20%. Figure 41 displays tonnages entering material recycling operations as a cohort of scenarios for differing illegal export ratio parameters.

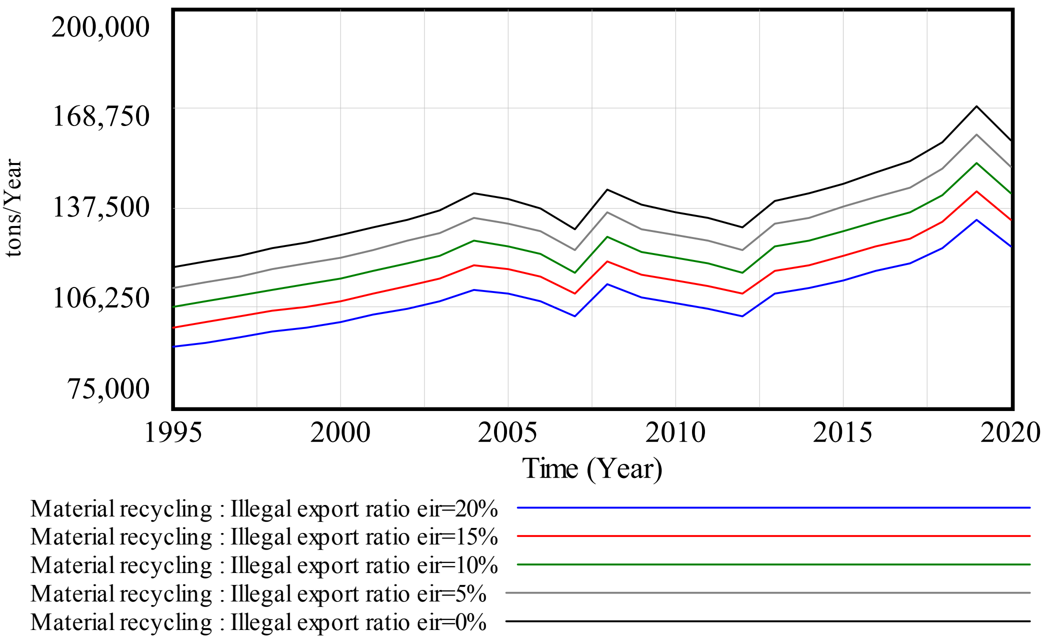

Figure 41: WEEE treated in material recycling operations

The German market is characterized by comparably strong competition if compared with the situation in other European countries. Approximately 20 recycling facilities are currently operated that can potentially process WEEE from segment i=3 in an environmentally sound way. Their recycling economics are approximated with a defined archetype plant featuring a maximum facility capacity per year of 320'000 units if run in 3 shifts. The level of competition is captured with the achievable *WACC*. The default value for *WACC* for the simulations of the German market is 6%. The average detour factor/road infrastructure coefficient v is assumed to be 1.3 and the default value for the transport utilization factor equals 2. The characteristics of the German economy are summarized in Table 20.

A more detailed description of segment-specific processing characteristics is necessary to understand further results of the recycling system simulation in segment i=3. Therefore, relevant variables and processing parameters for the recycling of cooling and freezing appliances are illustrated in more detail in the next section.

Depiction	Symbol	Unit	Value
Size of economy	ES	[capita]	82‘300‘000
Areal spread	A	[km^2]	357‘000
Amount of recycling facilities	RF_i	[units]	10-20 (default=20)
Labor costs	lr	[€/t]	15.0
Detour factor	k	[-]	1.3
Industry-specific costs of capital	WACC	[%]	6-25 (default=6)
Second hand export ratio	$sher_{i,t}$	[-]	0.02
Second hand import ratio	$shir_{i,t}$	[-]	0.01
Storage ratio	$sr_{i,t}$	[-]	0.2
Average use span	$aus_{i,t}$	[years]	11.4-15
Average storage span	$ass_{i,t}$	[years]	5
Illegal dumping ratio	$idr_{i,t}$	[-]	0.0005
Household bin ratio	$hwr_{i,t}$	[-]	0
Landfill and incineration ratio	$lar_{i,t}$	[-]	0.0001
Rem., repair, and reuse ratio	$rrr_{i,t}$	[-]	0.05

Table 20: Characteristics of the German economy

6.2.2 Processing Characteristics of Cooling and Freezing Appliances

Cooling and freezing appliances require specialized recycling equipment because the appliances contain CFCs with enormous global warming potential and fluids that have to be removed. Both insulation material and cooling circuit can contain environmentally harmful gases that have to be recaptured and destroyed according to the WEEE directive (COM 2003). The standard treatment procedure for cooling and freezing appliances is as follows:

First, remaining food wastes and glass/plastic shelves are taken out. The power cable is cut off and the appliances are checked for hazardous items such as mercury switches and polychlorinated biphenols containing capacitors that are manually removed. Second, the appliance is brought to a sucking station (step 1), usually on a roller conveyor. The recovery of CFCs is done in two steps, step 1 dealing with the liquids in the cooling circuit and step 2 recovering CFCs from the insulation. In step 1, the cooling circuit is spot-drilled with a special pincer and the refrigerant and oil are completely sucked off. In Europe, a distinction between three appliance classes is made for the requirements in step 1. The most important group are appliances that contain CFC, HFC, or HCFCs in the cooling circuit (mostly R12, R22, R134a, R404a, R502). The current share of returning appliances $sa_{i,t,g}$ of that type (i=3, g=1) is about 84% (representative for Germany in 2006, Hug 2006) and will decrease in the future due to the phase-out of CFCs. As an average value, the sucking station should recover 115g of these substances per appliance (90% of expected maximum). In addition, oil is seperately recovered from the cooling circuit with an average weight of 240g. In the second group are the HC appliances (R290, R600a) which currently have a share of 12% (i=3, g=2) of the returning volumes. HC is less problematic from an environmental point of view; nevertheless, those liquids are removed as well and an average weight of 60g refrigerant and 240g oil can be expected. The third group (4% share of returning volumes, i=3, g=3) represents the ammonia appliances (NH3) which are easily identified by the thickness of the pipes of the cooling circuit. These liquids are sucked off and stored in an extra tank. The recovered amounts of cooling refrigerants can be measured and monitored with a normal pair of scales. Step 1 is summarized in Table 21.

Step 1				
Classification	**Types**	**Global warming potential [GWP]**	**Share of returning volumes [%]**	**Expected mass to be recovered per unit [g]**
Group 1 (CFC, HCFC, HFC)	R12	10600	84	115
	R22	1700		
	R134a	1300		
	R404a	3750		
	R502	5590		
Group 2 (HC)	R290	3	12	60
	R600a	3		
Group 3 (NH3)	R717	0	4	-

Table 21: Recovery of CFCs and other gases in step 1 (IPCC 2001, Hug 2006, own analysis)

After removal of the liquids in the cooling circuit, the compressors are cut off with a hydraulic cutter. The appliances (now called "cabinets") are then further processed and reduced to small pieces (step 2). Various technologies are applied in step 2 for both size reduction and recovery of CFCs and cyclopentane from the insulation where such gases were used as a blowing agent. Most solutions use an enclosed shredder combined with nitrogen neutralization (inertisation, due to explosion potential of VOCs/HCs) for size reduction and a cryocondensation unit in order to recover the CFCs and cyclopentane from the gas stream that is sucked off from the shredder chamber. Additional equipment is necessary to decrease the residual CFC content in the shredded polyurethane pieces to the standards mandated in some countries (<0.2% residual CFC content).

Again, a distinction into three classes is apposite for step 2. The biggest share of appliances processed (about 81%, g=4) has insulation that contains CFCs (R11 and a minimal, negligible part of R12). The second largest group features insulation blown with cyclopentane (about 18%, g=5). These appliances are often processed as cabinets in a car shredder. A small remaining share of (mostly very old) appliances is insulated with glass wool or other "non-polyurethane" (1%, g=6) and needs separate manual preparation. As an average value, the insulation foam of the appliances from the first group should enable a recycler to recover 283g R11 per unit (90% value) and about 140g of cyclopentane per unit. Since some appliances were already damaged before processing or lost CFCs from the insulation for various reasons, the 283g are usually not attained in daily business. The recovery from step 2 can be monitored with raw gas measuring instruments on the input and output side and, in the case where cryocondensation is applied, with the weight of the tanks where the gases are collected. Step 2 is summarized in Table 22.

Step 2				
Classification	**Types**	**Global warming potential [GWP]**	**Share of returning volumes [%]**	**Expected mass to be recovered per unit [g]**
Group 1 (Polyurethane insulation with R11 + minimal share R12)	R11	4600	81	283
	R12	10600		
Group 2 (Polyurethane insulation with cyclopentane)	C5H10	11	18	140
Group 3 (glass wool and other non-PUR)	-	0	1	-

Table 22: Recovery of CFCs and other gases in step 2 (IPCC 2001, Hug 2006, own analysis)

After shredding, the material from the appliances is sorted into ferrous metals, non-ferrous metals, a polystyrene-dominated plastics fraction and a polyurethane fraction. Standard sorting equipment (eddy current, magnetic separator) is sufficient for these purposes. Ferrous metals are directly obtainable in a high separation quality (>99%) and are sold to steel mills. The non-ferrous fraction usually goes to secondary recyclers for further separation of metals or directly to foundries that can use the material mix in the manufacturing of aluminum parts.

The plastics fraction is further processed by plastics recyclers that focus on the polystyrene in order to reapply it in new plastic components. The polyurethane is usually reapplied as oil binder, used for methanol synthesis, or burned for energy recovery. Cables go to a cable recycler in order to recover the contained copper. The compressors are in principle reusable or can be further processed by secondary recyclers in order to recover the materials (ferrous metals, copper). Figure 42 provides an overview of secondary material flows and shows anchorage points for MRCs.

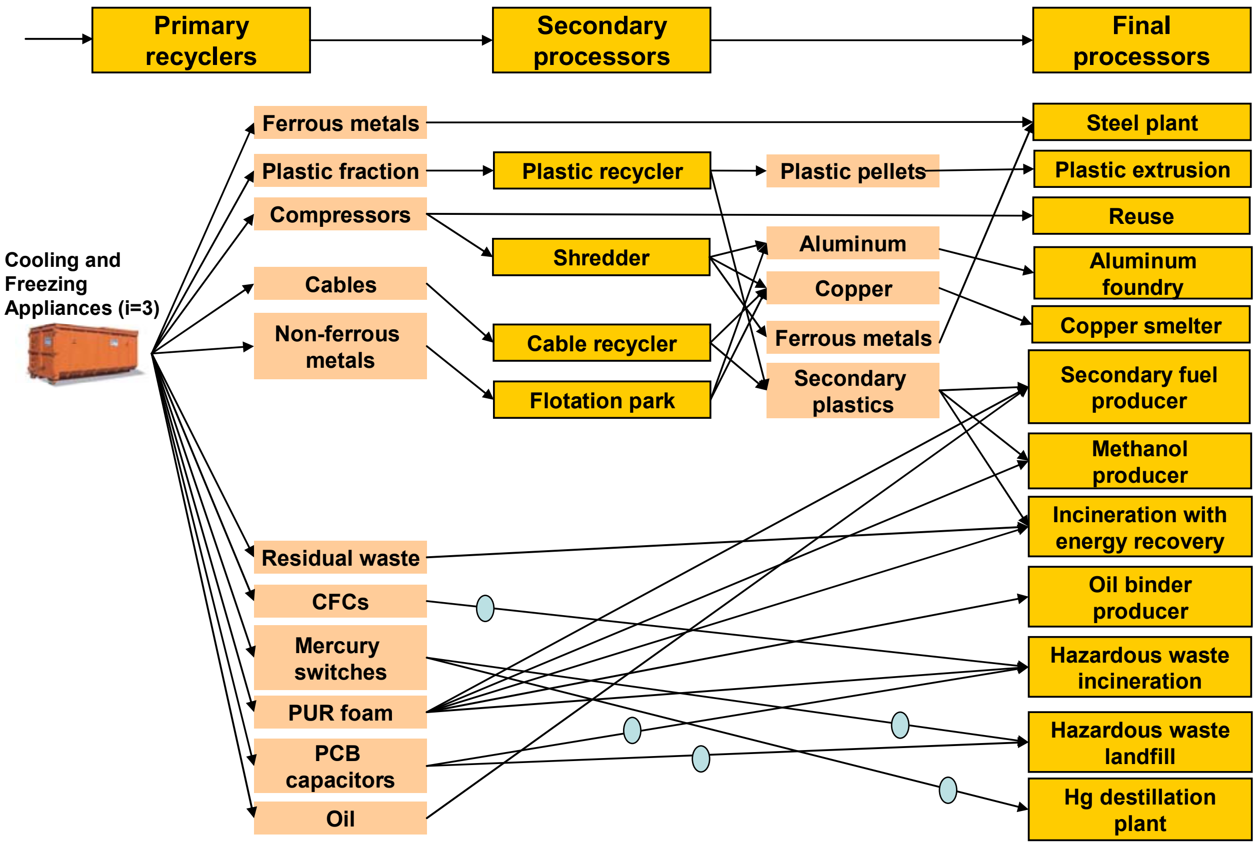

Figure 42: Secondary material flows from the recycling of cooling and freezing appliances

Furthermore, a quantitative overview of output fractions is necessary to determine recycling rates, depollution quality, and scoring factors under the MRC approach. Table 23 shows a representative split of output fractions for segment i=3, presuming a depollution quality level dql=0.9 and CFC appliances shares as indicated in Tables 21 and 22.

Code	Fraction description	Share
19 12 02/ 01-2	Ferrous metals (>99% pure)	47.000%
19 12 03/03-1	Aluminum/Copper	5.400%
16 02 16/ 12	Compressor	21.320%
16 02 16/ 10	Cables and wires	0.380%
19 02 07*/01	Oil	0.500%
19 12 04/ 02-1	Plastics (approx. 80 % polystyrene)	12.540%
19 12 04/ 05-1b	PUR foam (insulation)	9.760%
20 03 01	Non recyclable items (e.g. food residues)	1.380%
19 12 12	Glass wool and "non-PUR"	0.170%
14 06 03*/ 02	Cyclopentane (step 2)	0.050%
14 06 03*/ 02	R290, R600a (step 1)	0.010%
14 06 03*/ 01	NH3 (step 1)	-
19 12 05	Glass (from shelves)	0.700%
19 12 11*	Condensation water	0.330%
16 02 15*/01-2	Mercury components	0.011%
16 09 02*/ 02	PCB suspect capacitors	0.005%
14 06 01*	R11+R12 (step 2)	0.546%
14 06 01*	R12, R22, R134a, R502 (step 1)	0.228%
	Total	100.330%

Table 23: Representative split of output fractions in segment i=3

Recycling rates mandated by the C&C regulation approach (*rec*=75%) are easily attained for segment i=3 (Bohr et al. 2007b). The scoring factors sf_k for each fraction under the MRC market are determined according to equation (18b). If $\sum_{k'} fms_{i,k} \cdot sf_k$ is normalized to equal 1, incentive-compatible sf_k can be determined according to (26):

$$\Leftrightarrow sf_k = \frac{fms_{i,k} \cdot fp_k + adc_{i,k}}{\sum_{k'} fms_{i,k} \cdot fp_k + \sum_{k'} adc_{i,k}} \cdot \frac{1}{fms_{i,k}} \quad (26)$$

Table 24 amends Table 23 and shows sf_k for an MRC market approach and the contributional share of each fraction to the generation of MRCs.

Code fraction k	Fraction description	Share $fms_{i,k}$	Scoring factor sf_k	MRC contribution
1	Ferrous metals (>99% pure)	47.000%	0.00	0.00000
2	Aluminum/Copper	5.400%	0.00	0.00000
3	Compressor	21.320%	0.00	0.00000
4	Cables and wires	0.380%	0.00	0.00000
5	Oil	0.500%	0.00	0.00000
6	Plastics (approx. 80 % polystyrene)	12.540%	0.00	0.00000
7	PUR foam (insulation)	9.760%	0.00	0.00000
8	Non recyclable items (e.g. food residues)	1.380%	0.00	0.00000
9	Glass wool and "non-PUR"	0.170%	0.00	0.00000
10	Cyclopentane (step 2)	0.050%	0.00	0.00000
11	R290, R600a (step 1)	0.010%	0.00	0.00000
12	NH3 (step 1)	-	0.00	0.00000
13	Glass (from shelves)	0.700%	0.00	0.00000
14	Condensation water	0.330%	0.00	0.00000
15	Mercury components	0.011%	1198.39	0.13182
16	PCB suspect capacitors	0.005%	1315.68	0.06578
17	R11+R12 (step 2)	0.546%	120.95	0.66013
18	R12, R22, R134a, R502 (step 1)	0.228%	62.69	0.14293
	Total	100.330%		1.00

Table 24: Output fractions, scoring factors, and MRC contributions in segment i=3

The determination of depollution quality *dql* is subject to the decision-making of a recycler within the system model. This shall be analyzed in a dedicated section for the recovery of CFCs from step 1 and 2.

6.2.3 Decision-Making on Depollution Quality Levels

In order to provide a quantitative basis for decision-making of an archetype recycler in the German economy, it is necessary to analyze material disposal costs, depollution costs, and expected monetarized economic disadvantages from non-compliance.

Material disposal costs depend on fraction disposal prices fp_k and output mass shares $fms_{i,k}$ of each disposal fraction. The latter can be calculated with the expected maximal mass of recoverable hazardous materials of fraction k $ehmr_{i,k}$ and the depollution quality level $dql_{i,k}$ according to equation (27a). Average fraction disposal prices and the expected maximal mass of recoverable hazardous materials of fractions have been determined in the industry survey. Equation (27b) shows the respective values for CFCs from step 1 and 2.

$$mdc_i = \sum_k fms_{i,k} \cdot fp_k = \sum_k ehmr_{i,k} \cdot dql_{i,k} \cdot fp_k \qquad (27a)$$

$$mdc_{i=3} = \sum_{k=17,18} ehmr_{i=3,k} \cdot dql_{i=3,k} \cdot fp_k = 0.00859 \cdot dql_{i=3,k} \cdot 1800 \qquad (27b)$$

Depollution quality level and depollution costs are mainly influenced by the accuracy of screening for hazardous items, the maintenance level of equipment, processing care within the plant, shredder chamber sealing, and gas stream volumes sucked out off the shredder chamber. The basic functional relation between depollution costs and depollution quality level and its parameters are given in (28a). The depollution costs per ton input for the archetype plant for CFC recovery from step 1 and 2 have been estimated and gauged in the industry survey (own analysis) (28b).

$$adc_{i,k}(dql_{i,k}) = a_{i,k} \cdot b_{i,k}^{\;c_{i,k} \cdot dql_{i,k}} \qquad (28a)$$

$$adc_{i=3,k=17,18}(dql_{i=3,k}) = 0.05 \cdot 8.9^{3.35 \cdot dql_{i=3,k}} \qquad (28b)$$

Expected monetarized economic disadvantages from non-compliance are almost impossible to survey from practice. The exponential function revealed in (29a) serves as a default estimate for the relation between depollution quality level *dql* and *edfn*. The following rationale underlies the gauging of *edfn* according to equation (29b) for the system simulations underlying the German market: Current expert observations indicate that the depollution quality level as defined in this thesis lies slightly below 75% and in single cases around 50% in terms of CFC recovery in segment i=3 in the German market (ARD 2007). Therefore, an average value of *dql*=0.65 was estimated for the CFC depollution quality level in Germany's electronics recycling system under the current C&C regulation approach. This value has been used to gauge equation (29b) since the model's recycler profit maximization calculus should yield a maximal decision-relevant profit *DRP* for *dql*=0.65.

$$edfn_i(dql_{i,k}) = pncn \cdot d_i^{\;e_i \cdot dql} - f_i \qquad (29a)$$

$$edfn_{i=3}(dql_{i=3,k=17,18}) = 200 \cdot e^{-4 \cdot dql} - 3.6631 \qquad (29b)$$

Equation (29b) is not clearly defined with the matching condition of the optimal *dql* and the observed *dql* in current practice. Other parameters could be used in (29b) as well; however, (29b) is a reasonable proxy to describe and model a recycler's monetary incentive structures within the electronics recycling system. Figure 43 shows decision-relevant costs and the optimal *dql* under a C&C regulation approach.

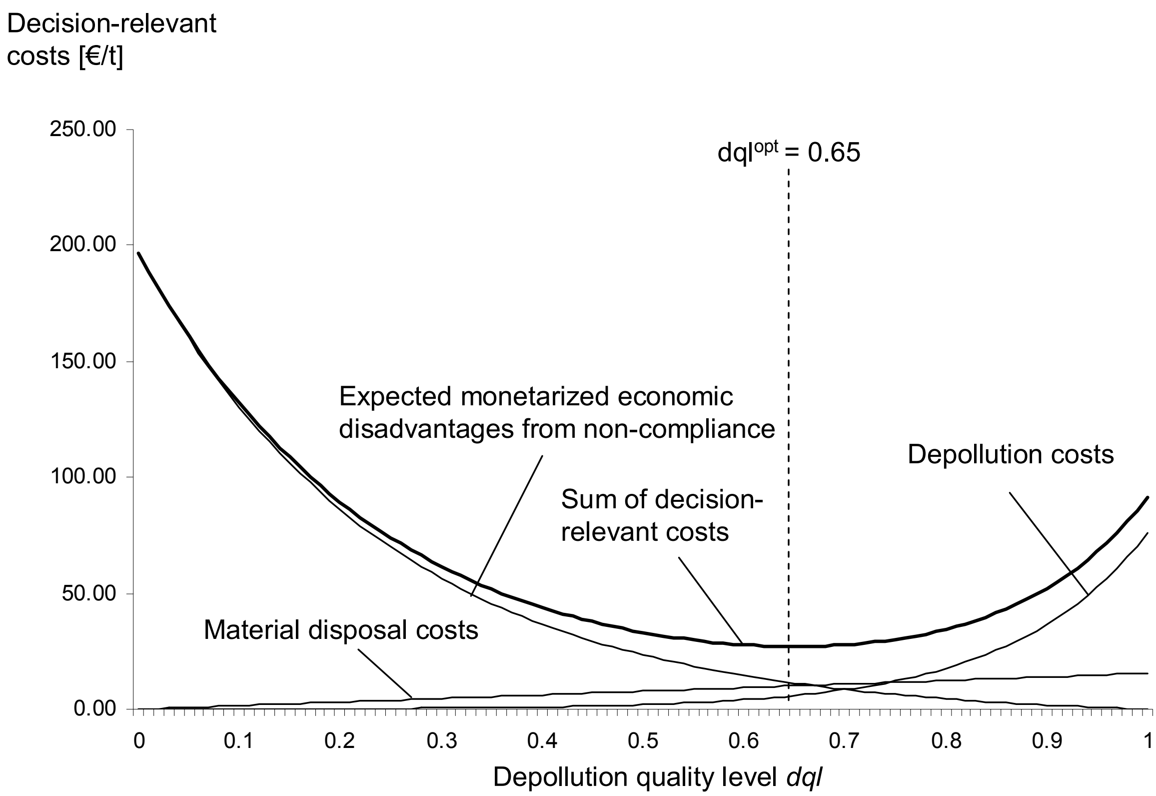

Figure 43: Decision-relevant costs under C&C regulation

Figure 44 shows decision-relevant profits faced by an archetype recycler in the German economy under a C&C regulation approach. The figure features fixed input remunerations of 126.76 €/t which result from the economic model for a *WACC* of 6%, a depollution quality level of 0.65, and average secondary material prices as surveyed in 2006.

The respective numbers under an MRC approach are interdependent since the depollution quality level affects the overall costs in the economy and indirectly the equilibrium material recovery certificate price $mrcp_{i,t}$ in the economic model. At the same time, $mrcp_{i,t}$ is used for the calculations of the MRC revenues that affect the recycler's decision-making on *dql*. Nested intervals can be used to solve for *dql*. For a defined setup of parameters, the equilibrium value for the simulation of the German market is *dql*=0.91 which gives rise to an $mrcp_{i,t}$ =324.99 €/t. Figure 45 illustrates decision-relevant profits for both the C&C approach and the MRC approach in the German market.

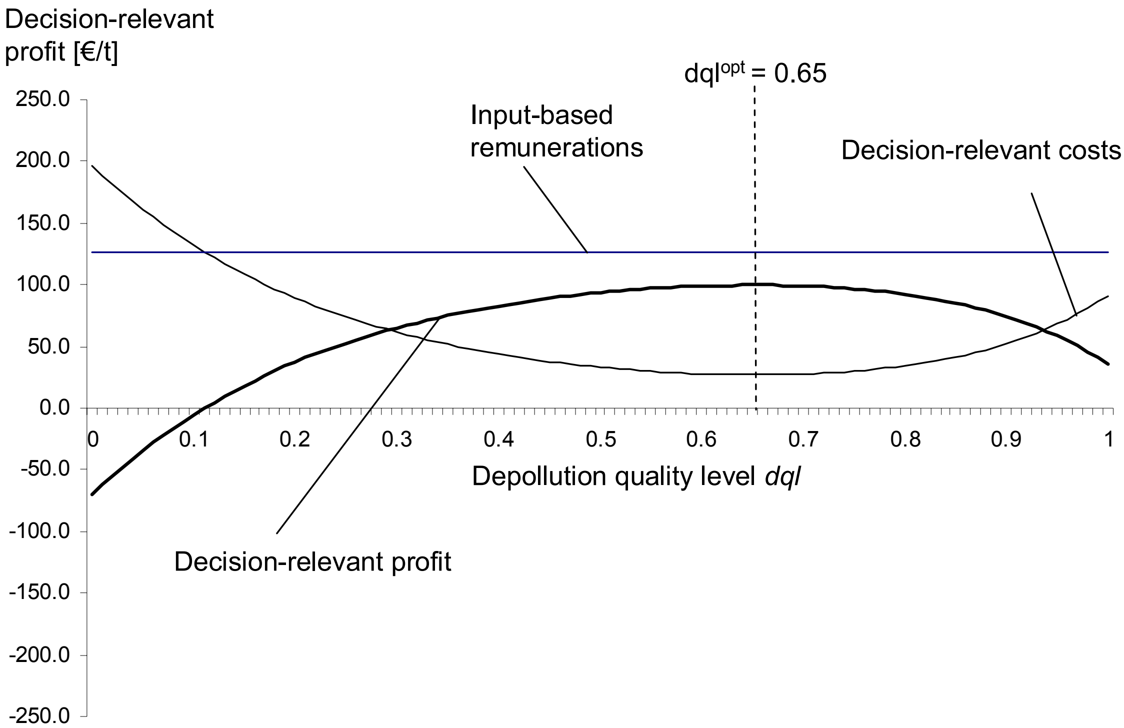

Figure 44: Decision-relevant profit under C&C regulation

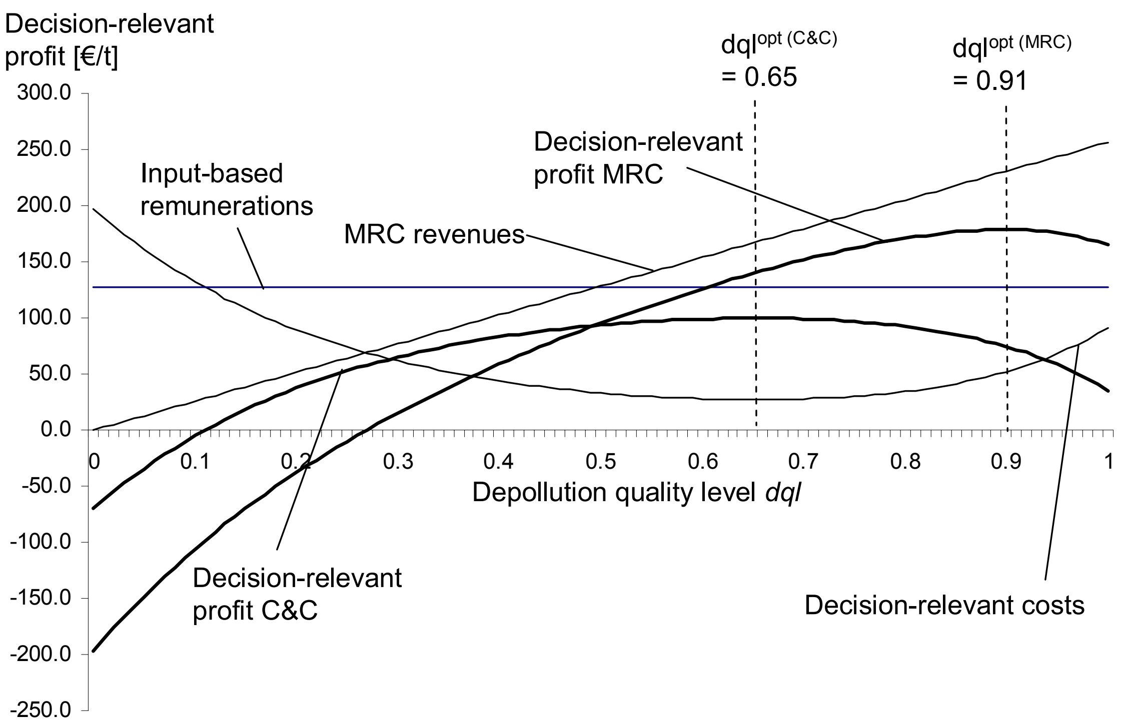

Figure 45: Decision-relevant profits of C&C and MRC regulation

In order to develop a grasp for the impact of an MRC market and its higher depollution quality levels on the environment, it is apposite to analyze economy-wide material flows. The recovery of CFCs from step 1 and 2 is used as an example and illustrated in the next section.

6.2.4 Economy-Wide Recovery of CFCs and Global Warming

The results from section 6.2.3 allow simulating the economy-wide recovery of CFCs from segment i=3. This requires further information on the development of the share of CFC appliances in return streams over time $sa_{i,t,g}$. A rough estimate for this development has been elaborated for the SENS system in an internal SENS document (Hug 2006). A prognosis of $sa_{i,t,g}$ is difficult because reliable data on the phase-out of CFCs in the production of appliances is not available. For the sake of simplicity, it was assumed that $sa_{i,t,g}$ decreases linearly from the 2006 levels to $sa_{i,t,g}$=0 in 2020. Equation (30) shows the calculation of economy-wide CFC recovery from segment i=3 based on this estimate.

$$ERHM_{k=17,18,i=3,t} = mr_{i=3,t} \cdot ehmr_{i=3,k} \cdot dql_{i=3,k=17,18}$$

$$= mr_{i=3,t} \cdot (sa_{i=3,t,g=1} \cdot 0.00253 + sa_{i=3,t,g=4} \cdot 0.00606) \cdot dql_{i=3,k=17,18} \quad (30)$$

Figure 46 shows the development of the economy-wide recovery of CFCs from step 1 for both C&C regulation (*dql*=0.65, *eir*=0.1) and MRC market regulation (*dql*=0.91, *eir*=0.05) over the period from 1995-2020.

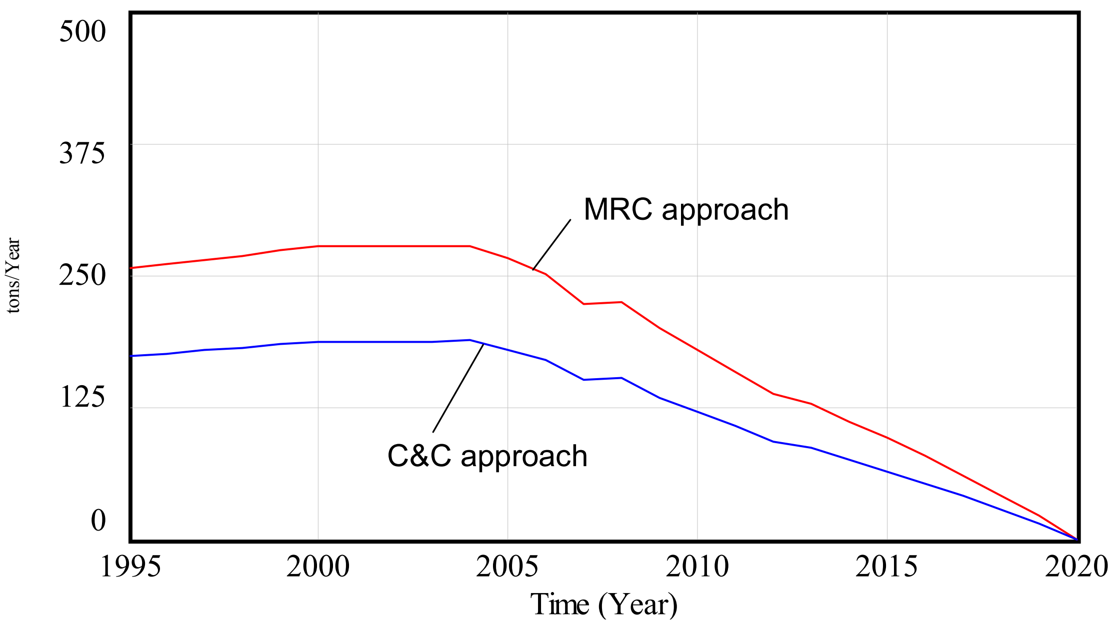

Figure 46: Economy-wide recovery of CFCs from step 1

Figure 47 shows the development of the economy wide recovery of CFCs from step 2 for both C&C regulation (dql=0.65, eir=0.1) and MRC market regulation (dql=0.91, eir=0.05) over the period from 1995-2020.

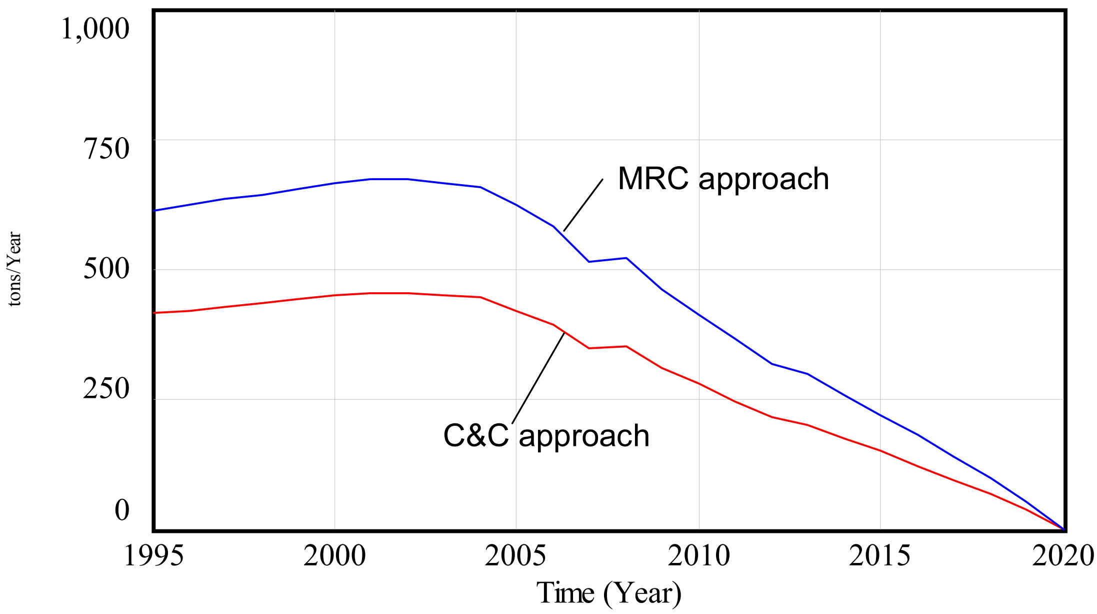

Figure 47: Economy-wide recovery of CFCs from step 2

In the light of global warming, it is also apposite to quantify the impact of both regulatory approaches on climate change, measured in CO^2 equivalents. Figure 48 illustrates the CO^2 equivalents recovered under both regulatory approaches.

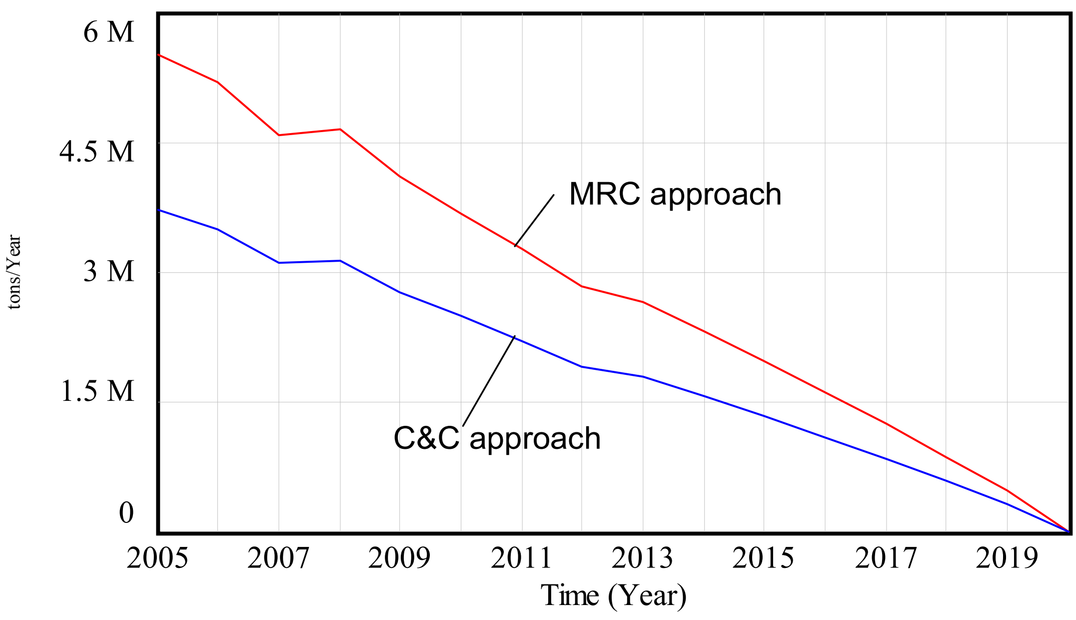

Figure 48: CO^2 equivalents recovered under both regulatory approaches

Figure 48 is based on an average GWP of 10.000 for CFCs recovered in step 1 (R12 is the prevailing cooling agent) and an average GWP of 4600 for CFCs recovered in step 2 (R11 is the prevailing blowing agent). Figure 49 shows the accumulated savings in CO^2 emission equivalents for the period 2005-2020 if an MRC approach was applied instead of a C&C approach in the German market.

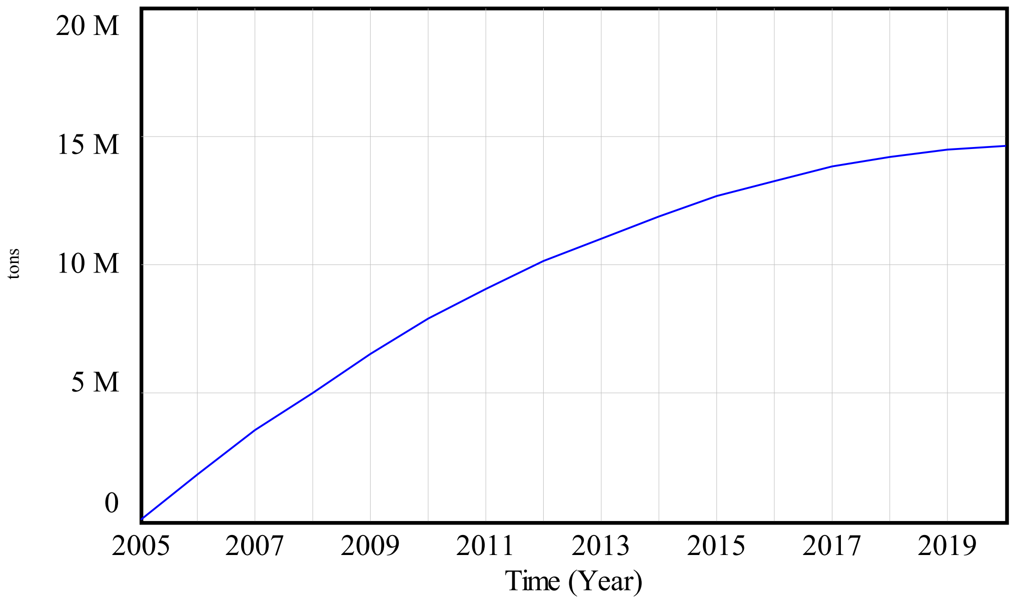

Figure 49: Accumulated savings in CO^2 emission equivalents

The graph reveals that the difference in CO^2 emission equivalents released to the atmosphere under the two different policy approaches is on average almost 1 million tons per year for the German economy. If one perceived the introduction of an MRC regulation approach as an investment, took into account a calculatory interest rate of 5%, and got paid the current price of CO2 equivalents under the European CO2 trading scheme for the EUA DEC 2008 (19.95€/t, Pointcarbon 2007) in each year, the present value of this "policy investment" would be roughly 198 Million €.

$$\text{Present Value of Policy Investment} = \sum_{t=1}^{15} (0.95)^t \cdot 972666 \cdot 19.95 \approx 198'000'000$$

6.2.5 Economics

An analysis of the results from the economic model shall be made for the material flows to be expected in 2008 according to the material flow model simulations. It is further assumed that the illegal export ratio under the C&C approach is *eir*=10% and under the MRC market *eir*=5%. This leads to 128'322 tons of WEEE entering material recycling operations in i=3 for the C&C approach and 135'871 tons under the MRC market.

6.2.5.1 Collection

Germany's collection infrastructure is very heterogeneous. Five collection options are modeled in the system simulations: Municipal garbage stations (j=1), collection events (j=2), retailer collection (j=3), direct returns at recycler sites (j=4), and bulky waste pickup (j=5). Estimates underlying the modeling of collection costs for the German market are based on the industry survey and a survey of administrative bodies from collection stations. Table 25 shows the class-specific collection cost shares sco_j, the collection point densities/frequencies cpd_j, fixed costs $fccp_j$ and variable costs per ton $vccp_j$ per collection point of option j, and collection costs per ton $cc_{j,t}$. Equations (31) and (32) show average collection costs per ton and economy-wide collection costs for both approaches (equal dedicated collection under both approaches).

2008	sco	cpd	fccp	vccp	cc
j=1	0.55	21	32526.25	0	68.79
j=2	0.05	300	15950	0	68.29
j=3	0.2	50	2630	79	117.71
j=4	0.05	5	1875	15	17.27
j=5	0.15	-	0	231.48	231.48

Table 25: Collection parameters and variables in the German economy

$$acc_t = \sum_j sco_j \cdot cc_{j,t} = 100.38 \text{ €/t} \qquad (31)$$

$$EACC_{i=3,t=2008} = dc_{i=3,t=2008} \cdot acc_t = 15'156'013 \text{ €} \qquad (32)$$

6.2.5.2 Transport

The average transport costs per ton and economy-wide transport costs strongly depend on the utilization factor uf and the amount of recycling facilities $RF_{i=3}$ that are active in the economy. Table 26 shows average transport distances between recycler and collection point $atd_{i=3}$, average transport distances per tour $atdt_{i=3}$, and average transport costs per ton $atc_{i=3}$ in segment i=3 for selected values of uf based on the load factor $lf_{i=3}$=2.9 and the default values for the economy-specific characteristics from Table 20.

2008	atd	atdt	atc
uf=1	80.03	80.03	37.44
uf=1.2	80.03	96.036	42.56
uf=1.4	80.03	112.042	47.65
uf=1.6	80.03	128.048	52.72
uf=1.8	80.03	144.054	57.76
uf=2	80.03	160.06	62.77

Table 26: Transport variables and parameters in the German economy

The economy-wide transport costs are shown in (33) for a default value of uf =2 that is deemed representative for the current C&C approach.

$$EATC_{i=3,t=2008} = dc_{i,t} \cdot atc_i = 9'477'425\,€ \qquad (33)$$

According to recyclers, a decentralized decision-making approach would allow to bring the utilization factor down to uf =1.2 in the German market. Respective savings in the economy-wide transport costs for segment i=3 are calculated in (34):

$$\text{Savings} = EATC_{i=3,t=2008}(uf = 2) - EATC_{i=3,t=2008}(uf = 1.2) = 3'051'328\,€ \quad (34)$$

Increased competition can change the amount of active recycling facilities in an economy. The impact of varying $RF_{i=3}$ on average transport distances per tour and average transport costs is portrayed in Figure 50 for uf =1.8 and a range $RF_{i=3}$=12 till $RF_{i=3}$=25.

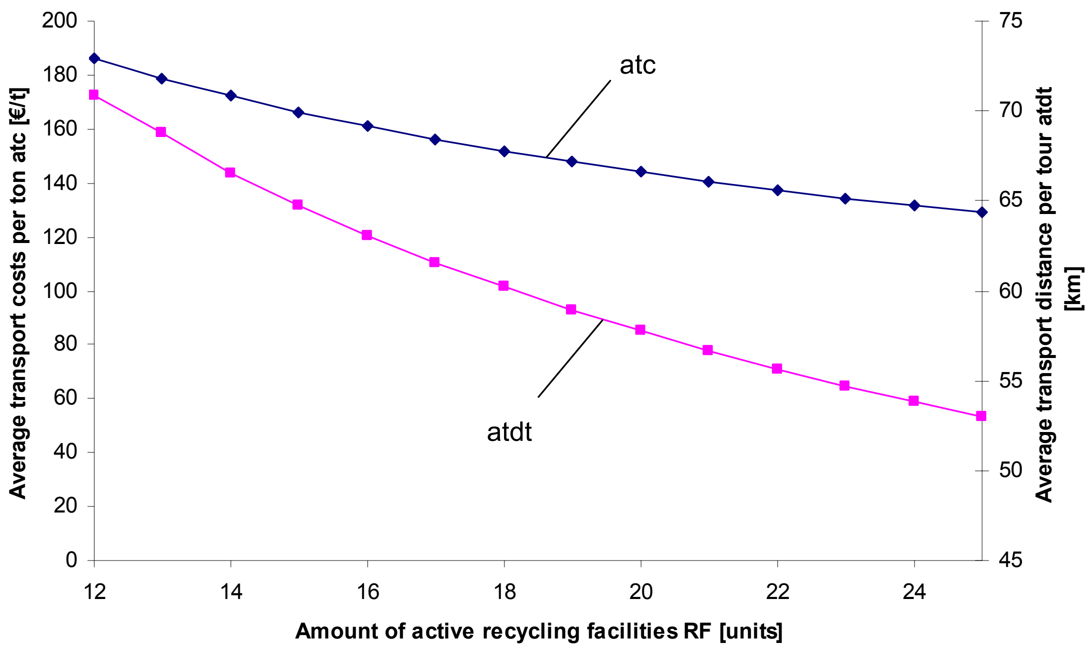

Figure 50: Sensitivity of transport costs to the amount of recycling facilities

6.2.5.3 Sorting

Sorting accounts for a negligibly small share of costs in segment i=3 in the activity framework defined in this thesis. The default value for the set h of sorting requirements underlying the German market simulation only requires sorting according to treatment purposes. Most related handling costs are captured under the operational costs from treatment. Average sorting costs per ton and economy-wide sorting costs are shown in equations (35) and (36a,b) for the C&C approach (a) and the MRC approach (b). The economy-wide sorting costs under the MRC approach are higher because more waste enters material recycling facilities and less waste gets illegally exported.

$$asc_{i=3,h} = \frac{15\text{ €/h}}{scc_{i,h}(spr_h) = 5t/h} = 3\text{ €/t} \quad (35)$$

$$EASC_{i=3,t=2008} = mr_{i=3,t=2008} \cdot asc_{i=3,h} = 384'971\text{ €} \qquad (36a)$$

$$EASC_{i=3,t=2008} = mr_{i=3,t=2008} \cdot asc_{i=3,h} = 407'618\text{ €} \qquad (36b)$$

6.2.5.4 Depollution

Depollution cost functions for fractions k=17,18 have already been shown in equation (28b). The remaining depollution costs related to PCB suspect capacitors and mercury switches are reflected in the sum of fraction-related depollution costs revealed in equations (37a,b). Average depollution costs per ton under the C&C approach are shown in (37a). Equation (37b) shows the respective value under the MRC market. Economy-wide depollution costs are calculated in equations (38a,b)

$$adc_{i=3}(dql_{i=3,k} = 0.65) = 14.61\text{ €/t} \qquad (37a)$$

$$adc_{i=3}(dql_{i=3,k} = 0.91) = 51.47\text{ €/t} \qquad (37b)$$

$$EADC_{i=3,t=2008}(dql_{i=3,k} = 0.65) = mr_{i=3,t=2008} \cdot adc_{i=3} = 1'984'530\text{ €} \qquad (38a)$$

$$EADC_{i=3,t=2008}(dql_{i=3,k} = 0.91) = mr_{i=3,t=2008} \cdot adc_{i=3} = 6'992'795\text{ €} \qquad (38b)$$

6.2.5.5 Treatment

The average recycling service remunerations or “treatment costs” strongly depend on the industry-specific weighted average costs of capital $WACC$ and the amount of recycling facilities $RF_{i=3}$ that are active in the economy. The latter directly impacts the capacity utilization in the economic model. Equation (39) shows average input-based recycling service remunerations per ton $arsr_{i=3,t=2008}$ under the C&C approach that include the coverage of depollution costs for a $WACC$ = 6% and $RF_{i=3}$=20. Economy-wide treatment costs for these parameters are calculated in equation (40).

$$arsr_{i=3,t=2008}(WACC = 6\%, \mathrm{RF}_{\mathrm{i=3}} = 20)$$

$$= asmr_{i=3} - fic_{i=3,t=2008} - daic_{i=3,t=2008} - ofc_{i=3,t=2008} - vco_{i=3} - adc_{i=3} - amdc_{i=3}$$

$$= 263.75\text{ €/t} - 47.36\text{ €/t} - 144.10\text{ €/t} - 83.67\text{ €/t} - 73.44\text{ €/t} - 14.61\text{ €/t} - 23.98\text{ €/t} = 123.41\text{ €/t} \quad (39)$$

$$EARSR_{i=3,t=2008}(WACC = 6\%, \mathrm{RF}_{\mathrm{i=3}} = 20) = mr_{i=3,t=2008} \cdot arsr_{i=3,t=2008} = 15'837'660\text{ €} \quad (40)$$

Figure 51 features a sensitivity analysis for the impact of varying $WACC$ on average input-based recycling service remunerations per ton and economy-wide treatment costs for a range of $WACC$ = 6% till $WACC$ = 20% and $RF_{i=3}$=20.

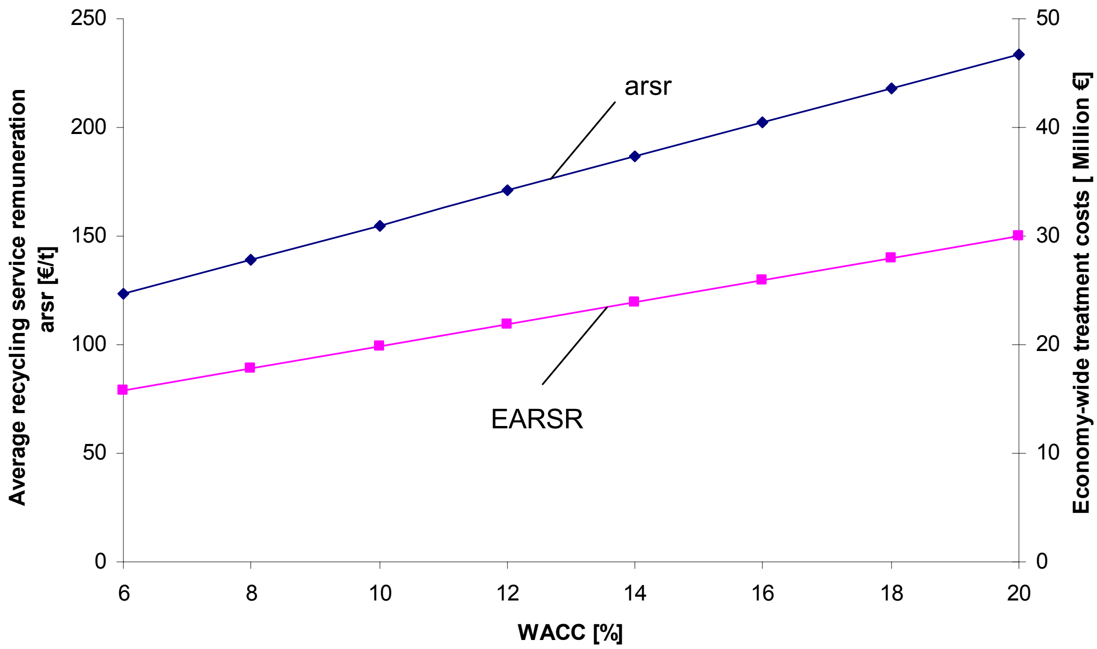

Figure 51: Sensitivity of treatment costs to industry-specific $WACC$

The output-based recycling service remunerations under an MRC market contribute with collection, transport, and sorting costs to the material recovery certificate price. In order to allow for a comparison to the input-based recycling service remunerations under the C&C approach, a separate analysis for MRCs that only cover treatment shall be made within this section. Table 27 reveals output mass shares of fractions with an anchorage point, scoring factors for each fraction, the MRC contribution of each fraction, and the amount of MRCs generated per ton input for a depollution quality level $dql_{i=3,k}$ =0.91.

2008	fms	sf	mrcs
k=15	0.00011	1198.39	0.132
k=16	0.00005	1315.68	0.066
k=17	0.00551	120.95	0.666
k=18	0.0023	62.69	0.144
Sum			1.008

Table 27: MRC supply parameters in the German economy

An $mr_{i=3,t=2008}$= 135’871 tons yields an economy-wide supply of material recovery certificates $EMRCS_{i,t}$ according to equation (41)

$$EMRCS_{i=3,t=2008} = mr_{i=3,t=2008} \cdot mrcs_{i=3,t=2008} = 136'959.63 \text{ t} \quad (41)$$

The MRC price per ton for an economy-wide depollution level $dql_{i=3,k} = 0.91$ and parameters from (41) is determined in (42) which shows the calculation of MRC prices in segment i=3 if only treatment costs (including depollution costs) are included in the pricing of MRCs.

$$mrcp_{i=3,t=2008} = \frac{EARSR_{i=3,t=2008}(dql_{i=3,k} = 0.91)}{EMRCS_{i=3,t=2008}} = 147.97 \text{ €/t} \quad (42)$$

The economy-wide MRC costs $EMRCP_{i,t}$ are given in equation (43).

$$EMRCP_{i,t} = mrcp_{i=3,t=2008} \cdot EMRCS_{i=3,t=2008} = 20'266'430 \text{ €} \quad (43)$$

6.2.5.6 Overall Costs

The activity-based costs presented in the previous sections constitute the major share of EPR-related costs for producers and importers in the German market. The economy-wide EPR costs $EEPRC_{i=3,t=2008}$ can therefore be approximated with the sum of the presented activity-based costs. Figure 52 shows the sum of activity-based costs and the contribution of each activity under the C&C approach ($dql_{i=3,k} = 0.65$, uf =1.8, $WACC$ =6%) for varying amounts of active recycling facilities in the economy in a range of $RF_{i=3}$=18 till $RF_{i=3}$=22.

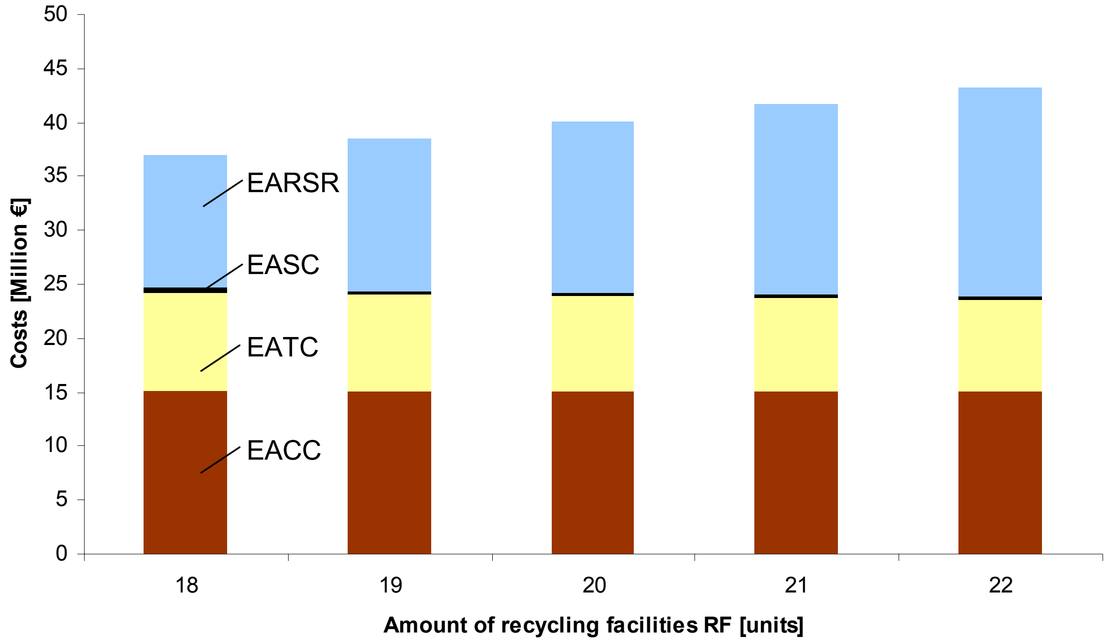

Figure 52: Economy-wide EPR costs for varying amounts of active recycling facilities

The economy-wide EPR costs for $RF_{i=3}$=20 are calculated in equation (44)

$$EEPRC_{i=3,t=2008} = EACC_{i=3,t=2008} + EATC_{i=3,t=2008} + EASC_{i=3,t=2008} + EARSR_{i=3,t=2008} = 40.10\,\mathrm{M}€ \quad (44)$$

An allocation of EPR-based costs on units sold in 2008 is made in equation (45). The estimated sales for the German market in 2008 are 3’618’753 units.

$$bsd_{i=3,t=2008} = \frac{EEPRC_{i=3,t=2008}}{\text{unit sales}} = 11.08\ €/\text{unit} \qquad (45)$$

A further specialty of the current German regulation approach is that collection costs are to be covered by municipalities. This reduces the unit-based costs of EPR considerably as shown in equation (46).

$$bsd_{i=3,t=2008} = \frac{EEPRC_{i=3,t=2008} - EACC_{i=3,t=2008}}{\text{unit sales}} = 6.89\ €/\text{unit} \qquad (46)$$

The refinancing mechanism of recyclers via material recovery certificates covers all steps of the value chain in the recommended default market setup with tradable material recovery certificates. This means that MRC prices $mrcp_{i=3,t}$ are in this case to be compared with the EPR costs per ton $eeprc_{i=3,t}$ under a C&C approach. Equation (47) shows how $mrcp_{i=3,t}$ can be calculated under the recommended setup and reveals the values for $uf = 1.2$, $WACC = 6\%$, and $RF_{i=3} = 20$.

$$mrcp_{i=3,t=2008} = \frac{acc_{i=3,t=2008} + atc_{i=3,t=2008} + asc_{i=3,t=2008} + arsr_{i=3,t=2008}}{mrcs_{i=3,t=2008}}$$

$$mrcp_{i=3,t=2008} = \frac{100.38\text{ €/t} + 42.56\text{ €/t} + 3\text{ €/t} + 149.16\text{ €/t}}{1.008} = 292.75\text{ €/t} \qquad (47)$$

Increased competition from MRCs could lead to a consolidation of active facilities while the industry-specific $WACC$ remains constant. Holding other parameters constant, Figure 53 shows the relation between the material recovery certificate price $mrcp_{i=3,t=2008}$ and the amount of active facilities in the economy ranging from $RF_{i=3} = 20$ till $RF_{i=3} = 12$.

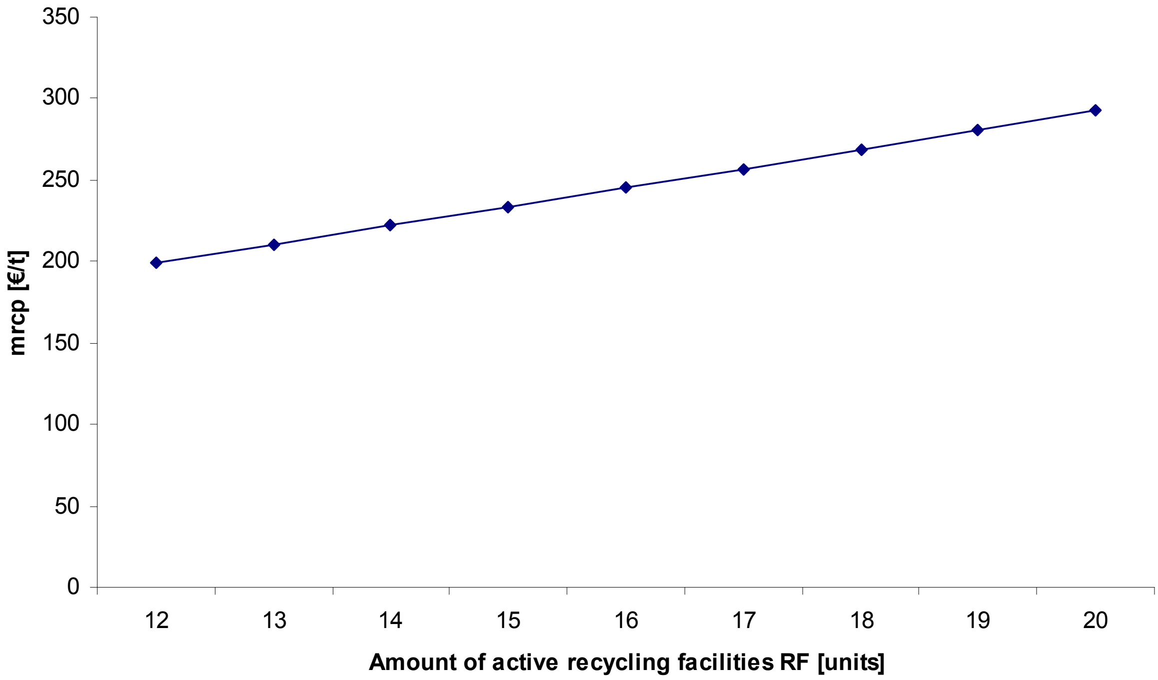

Figure 53: MRC price in relation to amount of active recycling facilities

The economy-wide MRC costs $EMRCP_{i,t}$ under the recommended default setup are given in equation (48) which draws from the economy-wide MRC supply $EMRCS_{i=3,t=2008}$ and (47).

$$EMRCP_{i,t} = mrcp_{i=3,t=2008} \cdot EMRCS_{i=3,t=2008} = 40'095'252\text{ €} \quad (48)$$

The demand side for MRCs is captured with the economy-wide demand $EMRCD_{i=3,t=2008}$ which depends on the steering lever $s_{i=3,t=2008}$ and the sales tonnage $ne_{i=3,t=2008}$. Equation (49) balances supply and demand on the MRC market for an $s_{i=3,t=2008}$=0.8056.

$$EMRCS_{i=3,t=2008} = 136'959.63 = 170'000 \cdot 0.8056 = ne_{i=3,t=2008} \cdot s_{i=3,t=2008} = EMRCD_{i=3,t=2008} \quad (49)$$

Figure 54 illustrates the MRC market setup in segment i=3 as simulated for the German economy.

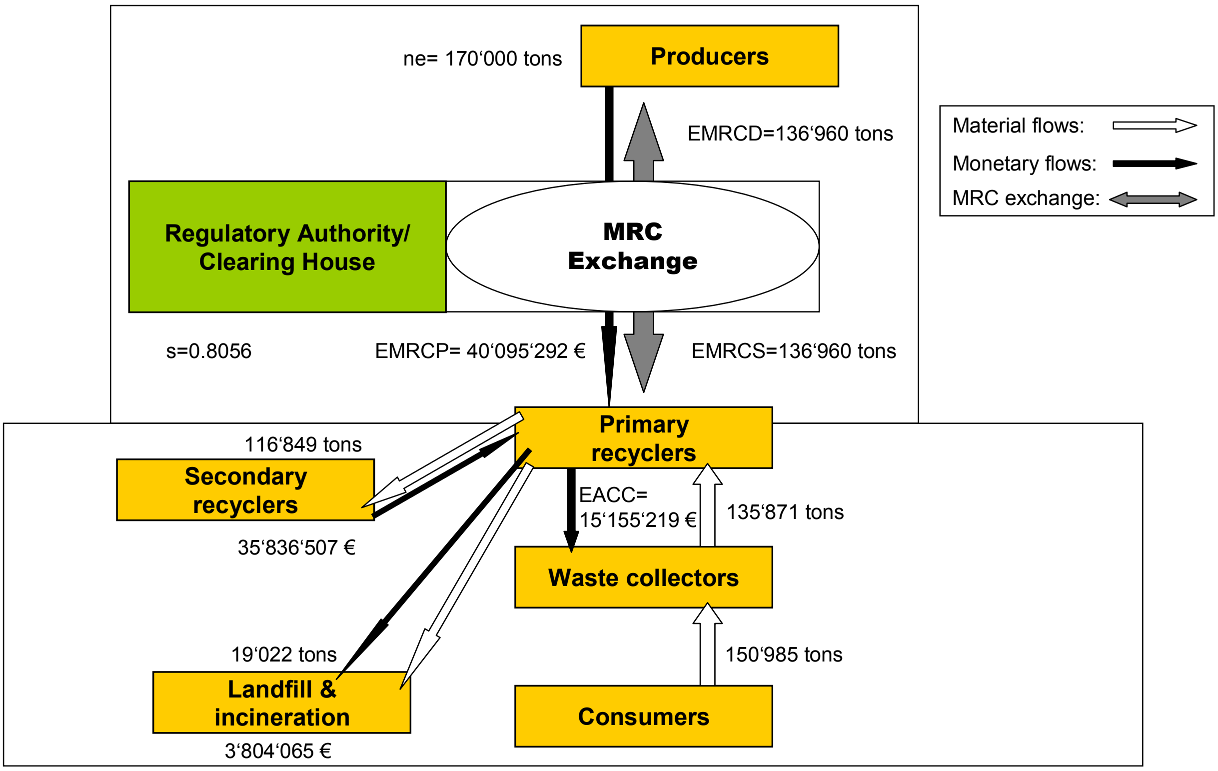

Figure 54: Key figures of an MRC market in the German economy in 2008

7 Conclusions

The design of regulatory policies has a significant impact on the ecological effectiveness and the cost-efficiency of electronics recycling systems. Both aspects need to be clarified and to be put in concrete measurable terms to allow for a meaningful analysis of regulatory policies and to drive their development. A further vital step in the design evolution process of regulatory approaches is an analysis of existing economic incentive structures in a system. Regulation should attempt to amend these structures in a way that relevant players face overall favorable incentives to achieve defined top targets.

Defining such top targets is not a simple, unbiased, and objective procedure. The policy assessment in this thesis relied on the performance in closing material loops *PCML* and the economy-wide recovery of hazardous materials *ERHM* in order to measure the ecological effectiveness of policy approaches. The cost-efficiency was reflected in the economy-wide EPR (compliance) costs *EEPRC* and the economy-wide MRC costs *EMRCP*. The figures are based on fundamentals of industrial ecology as well as expert feedback from the industry survey and are deemed suitable top targets in the context of electronics recycling systems.

Vigorous lobbying pressure from affected players and stakeholders often blurs the focus on such targets or the original environmental goals of a regulation. In the case of electronics recycling systems, complex reverse logistic networks, proprietary treatment technologies, non-transparent secondary material markets, and rapidly changing business environments further complicate the development and adjustment of regulatory policies. Nevertheless, an analysis of the current raw model policy and its implementation – the WEEE directive and corresponding national regulation – reveals several shortcomings that could be improved without compromising the interest of affected players and stakeholders. A mismatch of theoretical and practical system characteristics, a wrong or unclear goal focus, and no accounting for practical economic incentive structures characterizes most of Europe’s current national command & control approaches. A refocusing of the policy framework is urgent and imperative.

Using a more flexible approach which sets economic incentives to achieve defined top targets can substantially enhance the (cost-)efficiency and ecological effectiveness of regulatory systems (Costanza et al. 1997). By focusing on such targets instead of dictating solutions, the regulated community is given the opportunity to flexibly leverage their capabilities in order to meet the top goals (Richards et al. 1994). These considerations have lead to the MRC regulation approach proposed in this thesis. While the MRC market does not necessarily provide perfect solutions to all concerns raised in the context of electronics recycling, the MRC approach features intriguing properties compared to existing regulation practice in terms of system organization, recycling quality assurance, incentives for technological innovation, and EPR cost-efficiency. An MRC-based policy is therefore recommended to implement the concept of EPR. Section 7.1 summarizes the key results from policy analysis and evaluation and recapitulates the supporting rationale for the use of alternative MRC-based policy approaches. Section 7.2 contains major strategic implications for affected players and stakeholders under the recommended regulation approach.

7.1 Key Results from Policy Analysis

The main research results relevant for the design of regulatory policies are as follows:

- The WEEE directive does not appropriately address system-wide material flows. An inappropriate focus on the enforcement of recycling rates in conjunction with a low collection target per capita leads to considerable amounts of illegal exports with no subsequent sound treatment. However, extended producer responsibility should not end at the harbor. An MRC market allows for recycling service remunerations only in the case of treatment evidence and therefore turns illegal exports into a considerably less attractive end-of-life option. (see 4.2.2, 5.1.1.3)

- High recycling rates are increasingly fostered by economic incentives. However, a high depollution quality level and the material recycling of selected fractions is compromised by economic disincentives. Command & control approaches in association with low enforcement efforts lead to low industry-wide depollution quality levels and a low performance in the economy-wide recovery of hazardous materials. A refocusing of regulation policy is apposite. The depollution quality level should be directly connected to monetary flows in order to provide direct economic incentives to adhere to high depollution quality standards. The MRC approach achieves this connection and is capable to set focused incentives for the material recycling of problematic fractions. (see 4.2.1, 5.1.1.2, 6.2)

- Individual producer responsibility destroys economies of scale in the waste management sphere and does currently not create any environmental benefit. A cost-efficient electronics recycling system draws on large professional players with high-capacity plants and high capacity utilization rates. An MRC approach provides flexibility and avoids individual waste management responsibilities for producers in the waste management sphere. (see 4.2.3, 6.2.5.6)

- Individual producer guarantees are a complicated and powerless means to create incentives for environmentally favorable product design. Alternative means such as a voluntary demonstration of environmentally favorable product design characteristics and its remuneration via lowered MRC obligations hold a stronger potential to create design incentives and feature a higher likelihood to be politically enforceable. An MRC approach allows incorporating such feedback mechanisms. Furthermore, an MRC approach is based on a differing interpretation of producer responsibility and defines EPR obligations as a duty to feed a certain amount of materials back into the loop in accordance with the materials currently used in production. Producer guarantees are obsolete under an MRC market. (see 4.2.3, 4.2.5)
- A distinction of historical and new WEEE is pointless if no real individual producer responsibility is achieved. The latter is impossible or at least highly unlikely to be enforceable under current economic conditions. A collective responsibility is sufficient to tackle the tasks in the waste management sphere. (see 4.2.5)
- Decentralized decision-making on how to collect, transport, and treat WEEE is highly likely to boost the cost-efficiency in large economies. A centralized random take-back allocation as observed in Germany considerably compromises the cost-efficiency of an electronics recycling system. An MRC market allows for decentralized decision-making and ensures a competitive recycling market via well thought-out steering mechanisms and transparent obligations. (see 4.2.4, 5.1.2)
- The reporting and monitoring structure defined by the WEEE directive needs to be adjusted to practical treatment flows. The segmentation of an MRC market can account for such flows. (see 4.2.4)
- The allocation of burdens among producers should solely be based on the mass and characteristics of products currently put on the market. Tracking and analyzing the composition of WEEE over time does neither serve any environmental purpose nor the fairness in the allocation of burdens. An MRC market abstains from tracking or analyzing the composition of WEEE over time. (see 4.2.6, 5.1)

7.2 Strategic Implications for Affected Players and Stakeholders

The application of a market with tradable material recovery certificates or the use of MRCs as remuneration mechanisms is recommended to both policy-makers designing new legislatory approaches and policy-makers concerned with an amendment of existing legislation. Affected players and stakeholders should be aware of the following strategic implications for their role and responsibility under MRC-based policies:

- Environmental authorities do not completely pass on the responsibility to safeguard the achievement of policy goals to industry. They are required to establish a clearing house that steers and controls the MRC market. The clearing house can be based on a lean management structure and can outsource several service requirements. Its financing is possible via fees on MRCs traded. The efforts related to the establishment and running of a clearing house are justified by a high likelihood for improved environmental results and higher system-wide cost-efficiency.
- The role of producer responsibility organizations is partially shifted and supplemented. PROs act as certificate brokers in the MRC market, leveraging their market power and industry expertise in order to offer competitively priced compliance packages to producers and importers. Furthermore, they can establish and supervise international projects in a form similar to the CDM and JI mechanisms under the European emission trading scheme in order to generate additional MRCs that can be offered to producers. Their expertise can further be sought-after for centralized quality control by the clearing house. If a virtual product design feedback is established which allows to lower MRC obligations of producers via a voluntary demonstration of superior product properties, PROs are a natural candidate to bundle expertise in this regard and offer consulting services to manufacturers and importers.
- Manufacturers and importers are liberated from burdensome guarantees and waste management tasks they neither have expertise nor interest in. Physical responsibilities for producers in the waste management sphere as currently observed in the German system do not arise under an MRC market. Furthermore, producers should lobby for the integration of CFCs in emerging or existing CO^2 emission markets. This would allow shifting the burden of the treatment of historical CFC-containing wastes to the industrial sectors covered by the European CO^2 market. The application of CFC-based MRCs would be a first preparational step toward this integration.
- Municipalities are free to negotiate with regional recyclers and determine the way to collect and the way to transport WEEE in association with their recycling partner in a direct business relationship. Established service connections can be leveraged. Municipalities are paid by recyclers for their collection services. Municipalities

compete with other collectors in the collector market. Appropriate collection service levels lie in the responsibility of municipalities.

- Transport companies and recyclers are free to work the collector market and they differentiate their collection payment according to the attractiveness and the collection service level provided by the collector. This implies that they have an enlarged control over the value chain and decide on WEEE sourcing and treatment. Recyclers compete with their MRC offers on the MRC market. Since payments are based on critical and hazardous fractions, a market consolidation that favors recyclers with leading technology and high depollution quality standards is likely to occur.

Admittedly, the political enforceability of an MRC policy is a challenge because it would require major changes in the cycle economy and a reinterpretation of extended producer responsibility and related duties. However, superior environmental results and a high cost-efficiency should encourage affected players and stakeholders to adapt a policy that is certainly closer to “focusing more on the results than on the means of achieving them”.

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Annex I

Visited Institution	Date
R-Plus, Eppingen, Germany	02/13/2006
R-Plus, Lustadt, Germany	02/14/2006
Bral, Berlin, Germany	02/17/2006
IWF, TU Braunschweig, Germany	02/20/2006
Electrocycling, Goslar, Germany	02/21/2006
Stiftung Entsorgung Schweiz, Zurich, Switzerland	03/02/2006
Immark AG, Regensdorf, Switzerland	03/03/2006
Schiess Recycling, Niederuzwil, Switzerland	03/06/2006
Bühlmann Altgeräte Recycling, Münchenviler, Switzerland	03/07/2006
Sohlenthaler Recycling, Gossau, Switzerland	03/08/2006
Tonner, Business House Zerlegebetrieb, Berneck, Switzerland	03/09/2006
Kasser, Büro für Umweltchemie, TK-SENS, Zurich, Switzerland	03/15/2006
Elektro Einkaufsvereinigung, EEV, Bern, Switzerland	03/20/2006
Electrolux, Zurich, Switzerland	03/22/2006
SENS, Zurich, Switzerland	03/23/2006
SENS, Zurich, Switzerland	03/27/2006
Edi Entsorgung, Lyss, Switzerland	03/28/2006
BAFU, Bern, Switzerland and FES/OREP, Bern, Switzerland	03/30/2006
MEWA Recycling, Gechingen, Germany	03/31/2006
Migros AG, Zurich, Switzerland	04/04/2006
Miele AG, Spreitenbach, Switzerland	04/05/2006
Entsorgung & Recycling Zürich, Zurich, Switzerland	04/06/2006
SWICO, Zurich, Switzerland	04/18/2006
DRISA, Lausen, Switzerland	04/19/2006
Karl Kaufmann AG, Thörishaus, Switzerland	04/20/2006
RUAG AG, Altdorf, Switzerland	04/24/2006
Roos & Partner, Luzern, Switzerland	04/24/2006
Thommen/Ceren AG, Kaiseraugst, Switzerland	04/25/2006
SEG Recycling, Mettlach, Germany	06/19/2006
Umweltforum Haushalt UFH, Vienna, Austria	07/11/2006
Gabriel, Technisches Umweltbüro, Vienna, Austria	07/12/2006

Visited Institution	Date
MBA Polymers, Kematen, Austria	07/13/2006
Kerp Engineering, Vienna, Austria	07/14/2006
CCR Logistics, Munich, Germany	07/17/2006
Metall + Recycling, Bergkamen, Germany	07/18/2006
ERN Hamburg, Germany	07/19/2006
MVR Hamburg, Germany	07/19/2006
SVZ Schwarze Pumpe, Cottbus, Germany	07/21/2006
Indra Recycling, Hockenheim, Germany	07/24/2006
Pagenkopf Abfalltrennprozesse, Köpenick, Germany	08/03/2006
Federal Ministry for the Environment, Nature Conservation and Nuclear Safety, Berlin, Germany	08/04/2006
STENA Miljo, Aussenfjellet, Norway	08/15/2006
Elretur, Oslo, Norway	08/15/2006
WEEE Recycling AS, Trondheim, Norway	08/16/2006
Hasopor AS, Lighting Equipment Recycling, Meraker, Norway	08/17/2006
Elkretsen, Stockholm, Sweden	08/21/2006
LESNI/STENA Refrigerator Recycling, Halmstad, Sweden	08/22/2006
Elretur Denmark, Copenhagen	08/23/2006
Air Purification Engineering LESNI, Billund, Denmark	08/24/2006
H.J. Hansen Recycling, Vejle, Denmark	08/25/2006
Theo Steil GmbH, Trier, Germany	09/06/2006
NVMP, Eindhoven, The Netherlands	09/07/2006
SIMS Mirec, Eindhoven, The Netherlands	09/07/2006
GfM, Ennepetal, Germany	09/08/2006
Norddeutsche Affinerie, Lünen, Germany	10/23/2006
Remondis Electrocycling, Lünen, Germany	10/23/2006
Nehlsen AG, Entsorga, Köln, Germany	10/24/2006
TechRec, Dublin, Ireland	11/23/2006
WEEE Forum Conference Interviews, Dublin, Ireland	11/24/2006
Appliances Recycling Center of America ARCA Inc., Compton, CA, USA	04/03/2007
ARC International, City of Industry, CA, USA	04/03/2007
Jaco Environmental, Fullerton, CA, USA	04/04/2007
Adams Steel, Anaheim, CA, USA	04/05/2007
AER Worldwide, Fremont, CA, USA	04/12/2007