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SUMMARY

This dissertation considers Sustainable Materials Management (SMM) as one of the cornerstones of the future green economy. The basic question regarding SMM is how to shift the behaviour of society towards meeting its material needs without destabilising the natural system nor jeopardising its future, in other words: how to preserve natural capital and reduce the environmental impacts of material life cycles. Taking into consideration the difficulty of identifying appropriate SMM policies, this dissertation focuses on examining the socio-economic aspects of SMM. More specifically, it studies the roles that sustainability assessment and economic optimisation methodologies together with policy instruments can and should play in the shift or transition in the management of materials. Forming the first part of this dissertation, chapter 2 investigates the interrelationships between **sustainability assessment methodologies** like Life Cycle Analysis (LCA), Life Cycle Costing (LCC) and Cost-Benefit Analysis (CBA). Due to methodological disparity of these three tools, conflicting assessment results generate confusion for many policy and business decisions. By providing a framework that shows the interrelationships between the different assessment methodologies, the dissertation helps to interpret and integrate different assessment results. Building on this framework, the dissertation discusses key aspects like reported metrics, different scopes and data requirements for example, that need to be considered when designing and integrating full sustainability assessment methodologies. The added value of the second chapter is illustrated by using the results of a study on automotive glass recycling which was carried out in Flanders (Belgium).

Next to the first part of the dissertation that discusses differences, similarities and connections between LCA, LCC and CBA, the second part develops complementary **economic optimisation modelling** techniques. These models can support policy in developing and implementing sustainable practices with regard to scarce landfill and non-renewable resource reserve problems. In that regard, the dissertation proceeds from structuring an existing product and project related methodological assessment structure on micro level towards the

development of economic optimisation models on macro level to simulate the effects policy instruments can have in terms of supporting a transition towards sustainable practices with regard to scarce landfill and non-renewable resource reserves. This way, a complementary methodological structure is created that puts emphasis on different aspects of the ‘real world’.

In practice, SMM policies are distinctive because of the complexity of the decision making chain within which they have to operate and the mix of policy instruments that is required throughout a material’s life cycle to support the move towards a circular economy. Policy instruments like extraction, production or consumption taxes and waste taxes all have a distinctive effect and therefore their, sometimes counteracting, results should be considered carefully when developing SMM policies. This all indicates that it can be hard to identify policy that triggers the transition towards a resource-efficient, circular economy. What makes this even harder is the fact that appropriate methodologies combining different elements like waste accumulation, recycling and substitution in a unified framework are lacking. In this regard, economic optimisation modelling techniques can be of great value. The economic optimisation methodologies developed in the second part of the dissertation are based on the well-known Hotelling model, and non-conventional applications and extensions are added. Instead of looking at only pure resource depletion problems, chapters 3 and 4 adapt the standard Hotelling formulation and apply it to **landfilling and Enhanced Waste Management (EWM)** practices in order to investigate how remaining landfill capacities and waste can be used in the most sustainable way. Carrying out example simulations using Flemish data, chapter 3 studies the effects of levying landfill taxes on remaining landfill capacity use and paves a way for applications higher ranked in the waste hierarchy like recycling for example. As this chapter shows, within the used simulation structure levying a landfill tax has the effect that yearly landfilled volumes decrease considerably, thereby prolonging the time horizon until exhaustion of remaining landfill capacity is reached. Moreover, although discounted profit falls when landfill taxes are used, discounted total welfare increases considerably. In addition to the third chapter, chapter 4 goes one step further by including the so-called EWM concept that includes Waste to Material (WtM) and Waste to Energy (WtE) practices. In carrying out example simulations based on Flemish data, this fourth chapter includes all relevant EWM process flows and input parameters and thereby develops a dynamic optimisation model that originates from the one presented in chapter 3. Doing this, chapter 4 analyses how landfill taxation schemes can be adjusted and transformed into waste taxes in case EWM would be applied, thereby creating a combination that enables sustainable ways of processing future waste streams. As the example simulations show, the optimal tax that optimises welfare in the no EWM scenario is higher than the one in the EWM scenario. This difference is justified by the positive external effects that are generated by EWM practices. Based on the example simulation results, generally applicable insights are given such as the fact that it is not necessarily the case that the higher the tax, the more effective waste management improvements can be realised. By deriving such insights, the third and fourth chapter could provide relevant information about the added value of using landfill and waste taxation systems within circular economy contexts.

From a **non-renewable resource depletion** perspective, the fifth chapter of this dissertation extends the Hotelling type of modelling technique by including aspects such as recycling, substitution and waste accumulation that, to our knowledge, were never combined all together with such a Hotelling model before. Next to being generically designed in order to be applied to different kinds of non-renewable resources, the developed simulation model's added value comes forward especially when taking into account non-competitive market settings, interacting policy instruments and environmental externalities at different stages of a material's life cycle. A numerical example about sand extraction is included in the fifth chapter to show typical outcomes and results that can be generated. As the results of these example simulations are all in line with expectations based on theory and logical reasoning, and taking into account the generic design of the model, the provided optimisation model can be of great value as a tool for designing policies supporting the transition towards a more resource-efficient economy. In turn, this can boost economic performance while reducing resource use and negative externalities, and shift the economy towards a more resource-efficient path that should bring competitiveness and new sources of growth and jobs through cost savings from improved efficiency and better management of resources over their whole life cycle.

SAMENVATTING

In dit doctoraatsproefschrift wordt duurzaam materialenbeheer (SMM) als één van de bouwstenen van een toekomstige, groene economie gezien. De centrale vraag die zich binnen SMM stelt, is hoe de maatschappij ertoe aangezet kan worden om aan haar materiaalnoden te voldoen zonder dat daarbij natuurlijke systemen uit balans gebracht worden en zonder dat daarbij de toekomst van de maatschappij in gevaar komt. Met andere woorden: hoe kunnen we ervoor zorgen dat natuurlijk kapitaal gevrijwaard wordt, daarbij rekening houdend met de milieu impacts die ontstaan in de levenscycli van materialen? Dit doctoraatsproefschrift benadrukt de moeilijkheden die gepaard gaan met het identificeren van geschikt SMM beleid en concentreert zich op de socio-economische aspecten van SMM. Meer bepaald bestudeert dit doctoraatsproefschrift de rollen die duurzaamheid beoordelingsmethodes en economische optimalisatiemethodes samen met beleidsinstrumenten kunnen en zouden moeten spelen bij het bewerkstelligen van een transitie in materialenbeheer. Hoofdstuk 2 vormt het eerste deel van dit doctoraatsproefschrift en gaat de onderlinge relaties na tussen verschillende **duurzaamheid beoordelingsmethodes** zoals Levenscyclus Analyse (LCA), Levenscyclus Kostenanalyse (LCC) en Kosten-Baten Analyse (CBA). Methodologische dispariteit in deze drie methodes leidt tot conflicterende beoordelingsresultaten die er op hun beurt voor zorgen dat er verwarring ontstaat bij vele beleids- en bedrijfsbeslissingen. Door een gestructureerd kader aan te reiken dat de connecties en samenhang tussen de verschillende beoordelingsmethodes weergeeft, helpt dit doctoraatsproefschrift bij het interpreteren en integreren van verschillende beoordelingsresultaten. Voortbouwend op het verduidelijkend kader behandelt het doctoraatsproefschrift sleutelaspecten zoals bijvoorbeeld rapporteereenheden, verschillende toepassingsgebieden en datavereisten. Met deze sleutelaspecten dient rekening gehouden te worden bij het ontwerpen en integreren van methodes die alle duurzaamheidsaspecten omvatten. De toegevoegde waarde van het tweede hoofdstuk wordt geïllustreerd door gebruik te maken van de resultaten van een

gevalstudie over de behandeling van automobielglas dat zijn einde levensfase bereikt heeft.

Naast het eerste deel van het doctoraatsproefschrift dat de verschillen, gelijkenissen en connecties tussen LCA, LCC en CBA bestudeert, worden er in het tweede deel complementaire **economische optimalisatie- en modelleringstechnieken** ontwikkeld. Deze modellen kunnen beleidsondersteuning geven bij het ontwerpen en implementeren van duurzame praktijken op gebied van schaarse stortcapaciteit reserves en niet-hernieuwbare grondstofreserves. In dat opzicht gaat dit doctoraatsproefschrift over van het verduidelijken en structureren van een bestaande product en project gerelateerde methodologische beoordelingsstructuur op micro niveau, naar het ontwikkelen van economische optimalisatiemodellen op macro niveau. Met deze modellen kunnen de effecten gesimuleerd worden die beleidsinstrumenten hebben in het ondersteunen van een transitie naar een duurzaam gebruik van resterende stort- en niet-hernieuwbare grondstofreserves. Op die manier wordt er een complementaire methodologische structuur gecreëerd die nadruk legt op verschillende aspecten van de ‘echte wereld’.

In praktijk zijn SMM gerelateerde beleidsmaatregelen vaak distinctief door de complexiteit van het besluitvormingsproces waarbinnen het beleid moet functioneren en de mix van beleidsinstrumenten die doorheen een materialencyclus vereist is om de transitie naar een circulaire economie te ondersteunen. Beleidsinstrumenten zoals extractie-, productie- of consumptieheffingen en afvalheffingen hebben allen een onderscheidend effect en daardoor zouden hun, soms tegenstrijdige, resultaten zorgvuldig overwogen moeten worden bij het ontwikkelen van SMM beleid. Dit geeft aan dat het vaak moeilijk is om een optimaal beleid te identificeren dat de transitie naar een grondstoffen-efficiënte, circulaire economie mogelijk maakt. Wat dit zelfs nog moeilijker maakt, is het gebrek aan geschikte methodes die verschillende elementen zoals afvalaccumulatie, recyclage en substitutie combineren in één universeel kader. In dat opzicht kunnen economische optimalisatietechnieken van groot belang zijn. De economische optimalisatiemethodes die ontwikkeld worden in het tweede deel van dit doctoraatsproefschrift zijn gebaseerd op het welbekende Hotelling model. Hieraan worden verscheidene niet-conventionele toepassingen en extensies toegevoegd. In plaats van enkel te kijken naar pure grondstofuitputting problemen, passen hoofdstukken 3 en 4 de standaard Hotelling formulering aan en worden de ontwikkelde modellen toegepast op **stort- en gevorderde afvalbeheerpraktijken (EWM)**. Dit om te onderzoeken hoe resterende stortcapaciteiten en afval gebruikt kunnen worden op de meest duurzame manier. Door simulaties uit te voeren aan de hand van Vlaamse gegevens gaat hoofdstuk 3 het effect na van het heffen van stortheffingen op het gebruik van resterende stortcapaciteit. Hierdoor wordt er een weg geëffend voor toepassingen die hoger gerangschikt staan in de afvalhiërarchie, zoals recyclage bijvoorbeeld. Zoals hoofdstuk 3 binnen de gecreëerde simulatiestructuur aantoont, heeft een invoering van stortheffingen het effect dat afvalvolumes die jaarlijks gestort worden, drastisch verminderen. Hierdoor wordt een volledige uitputting van resterende stortcapaciteit uitgesteld. Hoewel het verdisconteerde winstcijfer daalt wanneer stortheffingen gebruikt worden, neemt de totale verdisconteerde welvaart aanzienlijk toe. Aansluitend op het derde hoofdstuk, gaat hoofdstuk 4 een stap verder door het EWM concept aan het model toe te voegen. Dit concept omvat zogenaamde Afval naar Materiaal (WtM) en Afval naar Energie (WtE) praktijken. Bij het uitvoeren van de simulaties die gebaseerd zijn op Vlaamse data, voegt hoofdstuk 4 alle relevante EWM processtromen en input parameters toe en wordt er zo een dynamisch optimalisatiemodel ontwikkeld dat voortkomt uit het model dat gepresenteerd wordt in hoofdstuk 3. Door het model op deze manier verder te ontwikkelen, analyseert hoofdstuk 4 hoe stortheffingen aangepast kunnen worden en getransformeerd kunnen worden naar zogenaamde afvalheffingen in het geval dat EWM praktijken toepast zouden worden. Hierdoor wordt er een combinatie gecreëerd die het duurzaam omgaan met toekomstige afvalstromen mogelijk maakt. Zoals de voorbeeldsimulaties in het vierde hoofdstuk aantonen, ligt de optimale, welvaart maximerende stortheffing in het scenario zonder uitoefening van EWM praktijken hoger dan in het scenario waarin deze wel uitgeoefend worden. Dit verschil in heffingen wordt aangestuurd door de positieve externe effecten die door EWM gegenereerd worden. Zich baserend op de simulatieresultaten, verschaft hoofdstuk 4 algemeen toepasbare inzichten zoals bijvoorbeeld het feit dat het niet noodzakelijk het geval is dat hoe hoger de stortheffing is, hoe effectiever verbeteringen in afvalbeheer gerealiseerd kunnen worden. Door dergelijk inzichten aan te reiken, zouden het derde en vierde hoofdstuk relevante informatie kunnen verschaffen over de meerwaarde van het gebruik van stort- en afvalheffing systemen binnen de context van de circulaire economie.

Het vijfde hoofdstuk van dit doctoraatsproefschrift handelt vanuit het perspectief van **niet-hernieuwbare grondstofuitputting** en breidt de op Hotelling gebaseerde optimalisatietechniek uit met aspecten zoals recyclage, substitutie en afvalaccumulatie. Dergelijke aspecten werden, voor zover onze kennis reikt, nog niet eerder op een gecombineerde wijze in een Hotelling model verwerkt. Naast het feit dat het ontwikkelde model generiek ontworpen werd opdat het toegepast kan worden op verschillende niet-hernieuwbare grondstoffen, komt de toegevoegde waarde in het bijzonder naar voren wanneer gelet wordt op functies zoals het modelleren van niet-competitieve marktscenario's, interagerende beleidsinstrumenten en milieu-externaliteiten die kunnen ontstaan in verschillende fases van de materialencyclus. Het vijfde hoofdstuk bevat een numeriek voorbeeld over de ontginning van zand. Aan de hand van deze gevalstudie wordt er getoond wat typische conclusies en resultaten zijn die gegenereerd kunnen worden met behulp van het ontwikkelde model. Aangezien de resultaten van de voorbeeldsimulaties in lijn liggen met wat verwacht kan worden op basis van theoretische en logische redenering, en lettende op het generiek ontwerp van het model, kan besloten worden dat het ontwikkelde model als een betrouwbaar instrument gebruikt kan worden. In praktijk kan het model toegevoegde waarde leveren bij het ontwikkelen van beleid dat de transitie naar een grondstoffen-efficiënte economie ondersteunt. Deze grondstoffen-efficiënte oriëntering zou meer competitiviteit met zich kunnen meebrengen, samen met nieuwe bronnen van groei en werkgelegenheid die gecreëerd kunnen worden door een efficiënter grondstoffengebruik en een beter beheer van deze grondstoffen gedurende hun hele levenscyclus.

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LIST OF ABBREVIATIONS

CBA	Cost-Benefit Analysis
CF	Cash Flow
CGE	Computable General Equilibrium
CSR	Corporate Social Responsibility
eCBA	Environmental Cost-Benefit Analysis
EIA	Environmental Impact Assessment
eLCA	Environmental Life Cycle Analysis
eLCC	Environmental Life Cyce Costing
ELFM	Enhanced Landfill Mining
ELV	End-of-Life Vehicles
EWM	Enhanced Waste Management
fCBA	Financial Cost-Benefit Analysis
feLCC	Full Environmental Life Cycle Costing
fLCC	Financial Life Cycle Costing
LCA	Life Cycle Analysis
LCC	Life Cycle Costing
LCSA	Life Cycle Sustainability Assessment
MC	Marginal Cost
MCLT	Marginal Cost of Landfill Taxation
MCP	Mixed Complementarity Program
MCPF	Marginal Cost of Public Funds
NIMBY	Not In My Back Yard
NPV	Net Present Value
OVAM	Openbare Vlaamse Afvalstoffen Maatschappij
PDF	Private Discount Factor
PDR	Private Discount Rate
sCBA	Social Cost-Benefit Analysis
SDF	Social Discount Factor
SDR	Social Discount Rate
sLCA	Social Life Cycle Analysis
sLCC	Societal Life Cycle Costing

SMM	Sustainable Materials Management
WEEE	Waste from Electrical and Electronic Equipment
WtE	Waste to Energy
WtM	Waste to Material

CHAPTER 1.

Introduction

Considering Sustainable Materials Management (SMM) as one of the cornerstones of the future green economy, this dissertation is inspired by the need to shift the behaviour of society towards meeting its material needs without destabilising the natural system nor jeopardising its future. While highlighting the difficulty of identifying appropriate SMM policies, this dissertation focuses on examining the socio-economic aspects of SMM. More specifically, it studies the roles that sustainability assessment and economic optimisation methodologies together with policy instruments can and should play in the shift or transition in the management of materials. In a first part, the dissertation focuses on sustainability assessment methodologies and discusses differences, similarities and connections between LCA, LCC and CBA. By also developing a structured framework that shows the interrelationships between different sub-methodologies, the knowledge provided in the first part of this dissertation should support decision making in those cases where methodological disparity leads to conflicting assessment results, which in turn generate confusion for many policy and business decisions. In a second part, the dissertation develops economic optimisation models that can be used as a means to support policy in developing and implementing sustainable practices with regard to scarce landfill and non-renewable resource reserve problems. In this chapter, an introduction to the SMM concept is given in section 1.1. Section 1.2 and 1.3 function as an introduction to the two building blocks on which this dissertation is based, being the sustainability assessment part and the part about economic optimisation modelling of scarce reserve problems respectively. Section 1.4 presents the research design by handling on the research questions that are tackled in this dissertation.

1.1. Sustainable Materials Management (SMM)

Over a long period in history, human behaviour had only little impact on the environment from a global perspective. Since the industrial revolution however, things changed considerably as the energy system became more and more based on fossil fuels and yet more materials were introduced to increase output and efficiency. Together with growing populations, and an increasingly intensive and extensive use of natural resources, global material extraction more than doubled in the past 30 years, from around 36 billion tonnes in 1980 to almost 85 billion tonnes in 2013. Between 1980 and 2002, only a modest growth in material extraction took place. From 2003 onwards however, global material extraction increased significantly, mainly driven by the rise of emerging economies like China and India and led by an enormous increase in the extraction of industrial and construction minerals (by more than 240%) (Materialflows, 2016).

With the size of the world economy expected to double and world population to increase by one-third by 2030, establishing a resource efficient economy is a major environmental, developmental and macroeconomic challenge today, especially since there are strong links between environmental degradation and social welfare reduction. In this context, Sustainable Materials Management (SMM) is an approach which aims to promote the sustainable use of materials from the point of resource extraction through to material disposal. Increased material sustainability should not only aim to reduce negative environmental impacts, but also account for economic efficiency and social aspects like social equity and ethical accountability. In December 2005, OECD agreed on a working definition for SMM by stating: "Sustainable Materials Management is an approach to promote sustainable materials use, integrating actions targeted at reducing negative environmental impacts and preserving natural capital throughout the life cycle of materials, taking into account economic efficiency and social equity." In this definition, the term 'materials' relates to all substances extracted or derived from natural resources, which may be either inorganic or organic substances. A further important aspect of the definition is the fact that the whole life cycle of a material should be included. In other words, all activities related to materials such as extraction, production, consumption, reuse, recovery and disposal should be included when investigating or developing sustainable material related practices. As a result, by reducing the amount of resources human economic activities require as well as diminishing the associated environmental and social impacts and improving resource security and resource scarcity while supporting reuse and recycling, SMM practices could support the transition towards a resource-efficient Europe. Moreover, in addition to making an important contribution to environmental protection and resource conservation, SMM can help to improve competitiveness and create growth and jobs.

The need for policy intervention in the field of SMM arises mostly because unregulated resource and waste markets would lead to socially-undesirable outcomes in the presence of scarcity and externality issues. These issues underline the strong link between environmental degradation and social welfare reduction. After all, natural resources are taken out of their natural deposits, which already has an impact on the environment where extraction takes place, get processed and used in the production of consumption goods and are disposed when they become waste in their end-of-life stage. Consequently, as remaining natural resource reserves continue to get scarcer, their availability for the economic and social system is limited. This would even worsen the rivalrous character of common pool resources like virgin sand for example. Next to the scarcity issue, another market failure exists in the form of external effects of production and consumption on the environment and, perhaps less commonly but equally important, their social and ethical consequences. Relevant examples of externalities are pollution and groundwater contamination, and other external effects associated with landfill disposal and virgin resource extraction that might also play an important role in getting a so called social license to operate. In practice, market prices rarely reflect external costs and scarcity issues that exist in the absence of proper government regulation (Eyckmans and Dubois, 2014; Söderholm and Tilton, 2012; Smith, 2014). Current resource prices are increasingly volatile and do not provide sufficient incentives for an efficient use of resources. Moreover, dependency on resources, as well as long-term impacts on the economy, the environment and society are often not properly reflected in economic decisions (Wilts et al., 2014). For this reason, there is a need for policy to counteract market system failures and ensure that external effects and scarcity issues are taken into account by the relevant decision makers. Implementing policy instruments to support a more sustainable material use is thereby justified. Moreover, this would be in accordance with the calls for 'true pricing' by accurately reflecting external costs and scarcities, and the green tax shift debate. At present however, the shift from labour towards environmental taxation is not tackled at all in a lot of European Member States, even though environmental taxes can be a step towards reflecting the full external and social costs of resource extraction, utilisation and end-of-life practices (Bringezu, 2002; Wilts et al., 2014). SMM policies are distinctive because of the complexity of the decision making chain within which they have to operate. Typically, a mix of policy instruments will be required as single instruments would lack the systematic regard to the impact on the overall structure incentives for consumers, producers and the waste management and recycling industries.

Being part of the bigger picture of sustainable development, SMM forms only one of several existing terminologies used for an approach to promote sustainable material use. By promoting this, SMM is closely linked to the flagship initiative on resource-efficiency in the EU 2020 strategy, which aims to create a framework for policies to support the shift towards a resource-efficient and low-carbon economy (European Commission, 2014). As resource-efficiency implies that natural resources, raw materials, products and also waste are used as efficiently and as environmentally responsible as possible, the link with waste management and waste valorisation is obvious. Speaking about waste and resources, another relevant concept that deserves adequate attention is the circular economy package. Being submitted by the European Commission to the European Parliament in December 2015, this new concept aims to foster a circular economy that turns end-of-life goods into resources for others, closes loops in industrial ecosystems and minimises waste. To facilitate the move towards this kind of economy, the package consists of an EU action plan that establishes an action programme to increase recycling, reduce landfilling and address obstacles in terms of an improvement of waste management. By setting targets such as a landfill target to reduce landfilling to a maximum of 10% of municipal waste by 2030 and by relying on economic incentives to support recovery and recycling schemes, the proposed actions should contribute to closing the loop of product lifecycles through greater recycling and reuse, and bring benefits for both the environment and the economy (European Commission, 2015).

Within the encompassing SMM context, this dissertation highlights the difficulty of identifying appropriate SMM policies and focuses on examining the socio-economic aspects of SMM. More specifically, it studies the role that sustainability assessment methodologies should play in the selection of sustainable alternatives. Furthermore, the dissertation develops economic optimisation techniques that can be used to examine the effects of policy instruments like extraction, production or consumption taxes and waste taxes within scarce landfill and non-renewable resource reserve problems. In the end, the knowledge provided in this dissertation should act as support for policy in triggering a resource-efficient, circular economy. The next sections give an introduction to the two main building blocks of this dissertation, being the sustainability assessment part and the part about economic optimisation modelling.

1.2. Sustainability assessment

Since sustainability gains importance for decision making in policy and business cycles, interest in sustainability assessment tools is growing. An important aspect of the application of sustainability is the emphasis on multidimensionality by including economic, environmental and social dimensions. On the one hand, it is essential to analyse these dimensions in a weighted manner. On the other hand, an integrated view, translated into well-defined methods and procedures for weighting economic, social and environmental aspects is necessary for technology development, policy development, political opinion formation and well-considered private and public action (Van Passel, 2007). In recent years, the amount of methodologies relevant to SMM that can be used to assess material use in terms of its potential impacts on the environment, benefits to society and value for the economy, has proliferated. Although this variety of assessment methodologies allows analysis of all aspects of sustainability, it also induces confusion and disagreement. Methodological differences and different weights for environmental, economic and societal priorities lead to conflicting assessment results for many important policy and business issues. Furthermore, while broadening the methodologies to conduct sustainability assessments, fragmented developments by a variety of research disciplines led to vague distinctions and as a result, synergies between different assessment methodologies became difficult to identify.

In an OECD study on methodologies relevant to the OECD approach on SMM, sustainability assessment methodologies like Life Cycle Analysis (LCA), Life Cycle Costing (LCC) and Cost-Benefit Analysis (CBA) are being classified as problem-oriented approaches. Such approaches focus on specific problems within a system and provide detailed results to assist in making decisions and implementing policy measures. Complementary to these problem-oriented approaches, systems-oriented approaches like economic optimisation and equilibrium modelling techniques have a broader scope and provide a context to assist in forming strategies and setting priorities. In other words, in the context of SMM it can be valuable to combine systems- with problem-oriented approaches (OECD, 2008; Bouman et al., 2000). The first part of this dissertation discusses the differences, similarities and connections between LCA, LCC, and CBA and their most relevant sub-methodologies. By also developing a structured framework that shows the interrelationships between different sub-methodologies, the provided knowledge should support complementary use of the different methodologies. Next to illustrating the policy problems resulting from the proliferation of assessment tools, key aspects in developing full sustainability assessment methods and connections between existing methodologies are discussed.

1.3. Economic optimisation: inducing SMM

As mentioned earlier, SMM policies are distinctive because of the complexity of the decision making chain within which they have to operate and the mix of policy instruments that is required throughout a material's life cycle to support the move towards a circular economy. Policy instruments like extraction, production or consumption taxes and waste taxes all have a distinctive effect and therefore their, sometimes counteracting, results should be considered carefully when developing SMM policies. This all indicates that it can be hard to identify policy that triggers the transition towards a resource-efficient, circular economy. What makes this even harder is the fact that appropriate methodologies combining different elements like waste accumulation, recycling and substitution in a unified framework are lacking. In this regard, economic optimisation modelling techniques can be of great value. Numerical optimisation problems are known to serve a couple of objectives. Firstly, they are able to quantify the net effect of counteracting forces which theoretical models usually cannot sign unambiguously. Secondly, they serve as a bridge between theoretical general models and actual analyses of real-world policy design problems (Conrad, 1999; Epple and Londregan, 1993; Flakowski, 2004). Although such optimisation problems are actually simplified roadmaps of reality, they are able to describe the outcome of a market by depicting the behavioural relations that underlie this outcome. Furthermore, they can be used to track the impact of various possible policy instruments down to their effects on consumption and production (Bouman et al., 2000). This way, these optimisation techniques can provide generally applicable and policy relevant insights about how to develop and bring into practice more sustainable material practices. Furthermore, these kind of models could ideally provide more theoretical background to the concept of the circular economy, which would otherwise risk to remain an all-purpose word without precise meaning (Hoogmartens et al., 2016).

This dissertation applies economic optimisation modelling techniques as a means to simulate scarce landfill and non-renewable resource reserve practices and the effects different policy instruments can have within a material’s life cycle. Using GAMS software, Mixed Complementarity Program (MCP) formats were adopted to create detailed, robust and flexible modelling structures. Although the economic optimisation methodologies developed in this dissertation are based on the well-known Hotelling model, non-conventional applications and extensions will be shown like an application of the model to landfilling and EWM practices, and a development of a generic non-renewable resource model that includes aspects such as recycling and substitution that, to our knowledge, were never combined all together with such a Hotelling model before. This way, this dissertation provides a tool for designing policies supporting the transition towards a more resource-efficient, circular economy which can boost economic performance while reducing the scarcity of landfill and non-renewable resource reserves.

1.4. Research design

Having stressed the important roles that sustainability assessment and economic optimisation methodologies together with policy instruments can play in SMM contexts, this dissertation aims at gaining insight into the following main research question:

How can sustainability assessment and economic optimisation methodologies be designed and applied to support the transition towards Sustainable Materials Management?

In order to operationalise this main research question, it has been subdivided into several subquestions which will be answered throughout this dissertation. Forming the first part of the dissertation, chapter 2 focuses on differences, similarities and connections between LCA, LCC and CBA and develops a framework that shows the interrelationships between different assessment methodologies. In the second part, consisting of chapters 3, 4 and 5, the dissertation develops economic optimisation modelling techniques as a means to support policy in developing and implementing sustainable practices with regard to scarce landfill and non-renewable resource reserve problems. In that regard, the dissertation proceeds from structuring existing micro level product and project related sustainability assessment tools towards more macro level types of models with objective functions in the form of profit and welfare maximisation.

By answering the different subquestions below, an improved complementary methodological structure is elaborated that consists of problem-oriented approaches like LCA, LCC and CBA, and systems-oriented approaches like economic optimisation and equilibrium modelling techniques. A graphical overview of the structure and contents of this dissertation is provided at the end of this section in Figure 1. As this figure shows, the different chapters go deeper into the different methodological approaches, thereby creating a complementary methodological structure that provides different kinds of problem solving options. Below, the different subquestions are outlined and it is indicated in which chapter they are elaborated upon.

Giving shape to the first part of this dissertation, chapter 2 aims at gaining insight into the following research question:

What are similarities, differences and connections between LCA, LCC and CBA sub-methodologies?

Within widely applied methodologies such as LCA, LCC and CBA, further sub-methodologies were created in order to deal with the three pillars of sustainability, namely environment, economy and society (Gasparatos and Scolobig, 2012; Heijungs et al., 2010; Kloepffer, 2008; Pope et al., 2004). Although this proliferation of assessment tools allows analysis of all aspects of sustainability, it also creates confusion as methodological differences and different weights for environmental, economic and societal priorities lead to conflicting assessment results for many important policy and business issues. Moreover, although different research disciplines tried to broaden the different methodologies to conduct sustainability assessments, their fragmented developments led to more blurred distinctions and as a consequence, synergies between methodologies became even harder to identify. In this regard, the second chapter of this dissertation provides added value by examining the structural differences, similarities and connections between LCA, LCC and CBA as micro level product and project related assessment tools. Based on a descriptive review that presents their most relevant sub-methodologies in the way they are generally used in research, a structured framework then shows the interrelationships between different sub-methodologies, thereby providing knowledge that should support complementary use of the different methodologies. Moreover, chapter 2 discusses key aspects that should be taken into account when developing full sustainability assessment methodologies. Finally, the second chapter then illustrates how policy problems resulting from the proliferation of assessment methodologies can be dealt with by using the results of a study on automotive glass recycling which was carried out in Flanders (Belgium).

Being complementary to the first part of this dissertation, the second part consisting of chapters 3, 4 and 5 develops macro level economic optimisation models with objective functions in the form of profit and welfare maximisation. Justifiable within the framework of an illegal waste disposal directive, the second part assumes that there is no free disposal of waste in terms of illegal dumping or street litter and that there is no piling up of waste with consumers. Furthermore, in chapters 3 and 4 it is assumed that profit-maximising landfill and waste management operators are working in a perfectly competitive market. Although this perfect competition is more an ideal market structure rather than an actual market reality, it provides a simplified model to understand a complex economic system and it can estimate economic variables approximately. Moreover, it provides a strong base for welfare maximisation and helps in developing norms for performance applicable to an industry. Also in chapter 5 it is assumed that recyclers and producers of the substitute material are operating in a competitive market setting. As regards the representative mining companies that are included in chapter 5 however, both competitive as well as monopolistic scenarios are simulated as it is often difficult to maintain the assumption of competitive market behaviour given the high level of market concentration in the mining sector. With respect to the waste volumes that are generated in chapter 5, it is assumed that there is no market for waste and that recyclers can therefore obtain waste for free. All produced waste volumes are given to the recycling sector, which in turn perfectly complies by accepting the whole stream. Obviously, these assumptions hold interesting opportunities in terms of future research. Each of them however, would add considerable complexity to the developed optimisation models. Only by using consistent numerical simulation modelling frameworks such as the ones presented throughout this dissertation, it will be possible to investigate more complicated but realistic scenarios.

Dealing with the scarcity of remaining landfill reserves and sustainable ways of managing waste, chapter 3 and 4 focus on the landfilling and waste management related aspects of the circular economy concept. In this regard, chapter 3 aims at providing knowledge with respect to the research question:

Do landfill taxes support a more sustainable way of managing remaining landfill capacities and waste?

In the past, landfills were emerging at an increasing pace in order to deal with growing waste generation. Starting in the 1970s however, public attitude towards waste started to change as people realised the negative environmental externalities caused by landfilling and the valuable space it occupies (Strasser, 1999; Walsh, 2002; Van Passel et al., 2013). Together with the emergence of what is nowadays called the Not In My Back Yard (NIMBY) syndrome, this led to the awareness that volumes of landfilled waste have to decrease and be earmarked for applications higher ranked in the waste hierarchy like recycling for example. Although this syndrome has not been taken into consideration in policy decisions consciously, its occurrence actually played a constraining role in the failure to issue new permits for landfills. Together with the fact that vacant land is scarce and therefore expensive, this moratorium on new landfill zones made remaining landfill reserves scarce (Levinson, 1999; Van der Zee et al., 2004). Economic theory predicts that scarce goods increase in price such that consumption gets restrained. This effect however, is also the aim of legally introduced landfill taxes. Although the arguments for high and harmonised landfill taxes in Europe seem strong, debates on tax levels are heated because not all economic scholars are convinced that high landfill taxes are justified. The question that then arises, is how to reconcile the economic perspective with policy discourses. In this respect, the third chapter of this dissertation provides added value by applying a Hotelling-based optimisation methodology in a non-conventional way by focusing on scarce landfill reserves. By using constrained objective functions in the form of landfill company profit maximisation, the effects of landfill taxes on corresponding landfill and price paths are examined with the aim of identifying the best allocation of remaining landfill capacity over time. Furthermore, by applying welfare maximisation algorithms, optimal landfill taxation levels from a societal perspective can be identified.

Building on chapter 3, chapter 4 goes one step further by including the so-called EWM concept that includes WtM and WtE practices. Doing this, the fourth chapter aims at gaining insight into the following research question:

How to combine landfill taxes and EWM in order to enable sustainable ways of processing future waste streams?

In practice, it may be difficult to find a balance between imposing landfill taxes and defining an appropriate taxation level on the one hand and applying EWM practices on the other. After all, as landfill taxes have the effect of mitigating the scarcity issue of landfill capacity by reducing landfilled waste volumes, less material is made available for valorisation and the application of EWM practices also becomes less essential from a capacity point of view, thereby possibly eroding the application of EWM. Similar reasoning can be applied the other way around. As EWM practices substantially reduce the volumes of permanently landfilled waste, remaining free capacity will become practically inexhaustible. This has the effect that landfill taxes are made redundant from a depletion point of view. Only their use in terms of internalising external effects as a Pigovian tax remains in that case partially valid. In this regard, the fourth chapter of this dissertation provides added value by developing economic modelling techniques that can be used to analyse how landfill taxation schemes can be adjusted in case EWM would be applied, thereby creating a combination that enables sustainable ways of processing future waste streams. To this end, chapter 4 includes relevant EWM process flows and input parameters that are based on Flemish data, and thereby develops a dynamic optimisation model that originates from the one presented in chapter 3. Based on constrained objective functions in the form of EWM operator profit maximisation, it is examined how landfill taxes actually become waste taxes when levied at the gate of an EWM operator that permanently landfills only a minor part of all incoming waste streams. Next to this, chapter 4 uses the same type of welfare maximisation algorithms as in the previous chapter to analyse which tax level is optimal from a societal point of view. By applying dynamic optimisation techniques to the EWM concept, the fourth chapter provides a valuable contribution to the relatively new and unexplored area of applying such techniques to the waste management field.

While chapters 3 and 4 focus on the scarcity of remaining landfill reserves, chapter 5 starts from the same kind of optimisation model but adjusts and extends it substantially in order to apply similar techniques in the field of scarce non-renewable resource reserve problems. In contrast to chapters 3 and 4, chapter 5 does not impose a constraint on remaining landfill capacity, but imposes a restriction on the remaining non-renewable resource reserve volume. In this regard, chapter 5 aims at gaining insight into the research question:

How to optimally extract scarce non-renewable resources over time, taking into account recycling and substitutes?

Non-renewable resources are created by long-term geological processes. Consequently, their rate of formation is sufficiently slow – in timescales relevant to humans – that they are labelled as non-renewable (Perman et al., 2011). This, together with the intensive use of resources that enabled European wealth and wellbeing to grow, and strict demarcations of mining areas, causes remaining reserves to be limited and scarce (European Commission, 2011a). The European Union therefore identified resource-efficiency as one of seven flagship projects to pursue in its Europe 2020 strategy and looks for ways to answer the key question about how to identify the optimal extraction path over time for any particular non-renewable resource reserve. In practice, there is no straightforward answer to this question because non-renewable resources are quite heterogeneous and it is often unclear what policies should be undertaken in order to facilitate the transition towards a resource-efficient economy. This challenge is exacerbated by the lack of appropriate methodologies that combine elements such as waste accumulation, recycling and substitution in a unified framework. After all, although several theoretical models on resource extraction and recycling were developed in the past, together with some numerical simulation models for phosphorous extraction and recycling, none of these combined the different elements in a unified framework. In this regard, chapter 5 adds value by developing a comprehensive generic optimisation model that can be used to simulate non-renewable resource practices and effects of different policy instruments within the material flow of a particular resource. By including recycling, substitution and waste accumulation, the developed model extends the seminal cake-eating Hotelling-type model that dominates the resource economics literature. In addition to being generically designed, the added value of the model emerges especially when taking into account non-competitive market settings, interacting policy instruments and environmental externalities. Furthermore, a variety of constrained objective functions are included that allow utility, profit and first-best welfare maximisation. Including all this, chapter 5 provides a tool for designing policies that support the transition towards a more resource-efficient economy.

A final chapter holds the general conclusions and provides knowledge with respect to the different research questions cited above. The dissertation then finishes by providing policy recommendations and making suggestions for future research.

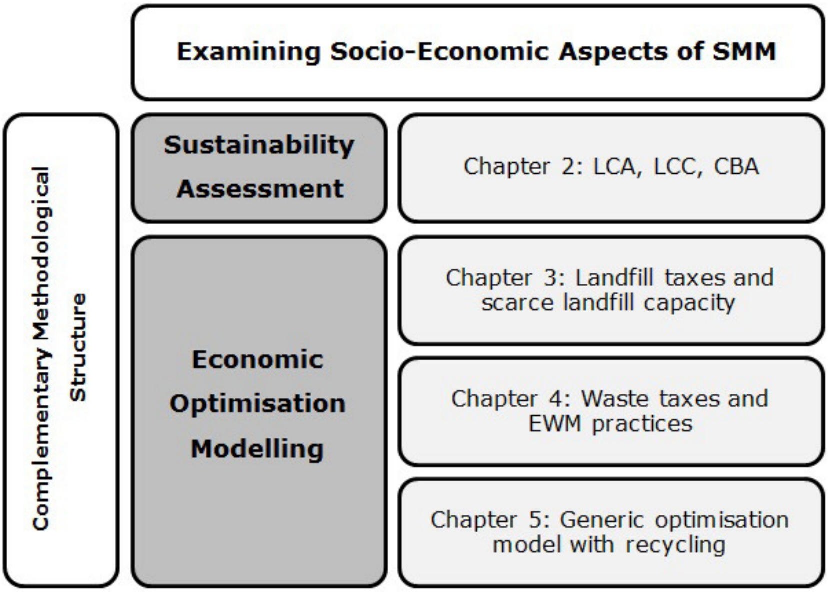

Figure 1: Structure and contents of this dissertation

CHAPTER 2.

Bridging the gap between LCA, LCC and CBA as sustainability assessment tools[1]

[1] This chapter is based on the publication: Hoogmartens, R., Van Passel, S., Van Acker, K., Dubois, M. (2014). Bridging the gap between LCA, LCC and CBA as sustainability assessment tools. Environ. Impact Assess. 48, 27-33.

Abstract

Increasing interest in sustainability has led to the development of sustainability assessment tools such as Life Cycle Analysis (LCA), Life Cycle Costing (LCC) and Cost-Benefit Analysis (CBA). Due to methodological disparity of these three tools, conflicting assessment results generate confusion for many policy and business decisions. In order to interpret and integrate assessment results, this chapter discusses the differences, similarities and connections between LCA, LCC and CBA. Next to developing a structured framework that shows the interrelationships between different sub-methodologies, the chapter further focuses on key aspects to adapt any of the methodologies to full sustainability assessments. Aspects dealt with in the review are for example the reported metrics, the scope, data requirements, discounting, product- or project-related and approaches with respect to scarcity and labor requirements. In addition to these key aspects, the review shows that important connections exist: (i) the three tools can cope with social inequality, (ii) processes such as valuation techniques for LCC and CBA are common, (iii) both LCA and CBA have a connection with Environmental Impact Assessment (EIA) and (iv) LCA can be used in parallel with LCC. Furthermore, the most integrated sustainability approach combines elements of LCA and LCC to achieve the Life Cycle Sustainability Assessment (LCSA). The key aspects and the connections referred to in the review are illustrated by using the results of a study on automotive glass recycling which was carried out in Flanders (Belgium).

2.1. Introduction

Since sustainability gains importance for decision making in policy and business cycles, interest in sustainability assessment tools is growing. In the context of SMM, sustainability assessment methodologies like LCA, LCC and CBA are being classified as problem-oriented approaches. Such approaches focus on specific problems within a system and provide detailed results to assist in making decisions and implementing policy measures. Complementary to these problem-oriented approaches, systems-oriented approaches like economic optimisation and equilibrium modelling techniques have a broader scope and provide a context to assist in forming strategies and setting priorities (OECD, 2008; Bouman et al., 2000). In order to deal with the three pillars of sustainability, namely environment, economy and society, further sub-methodologies have been developed (Gasparatos and Scolobig, 2012; Heijungs et al., 2010; Kloepffer, 2008; Pope et al., 2004). Figure 2 illustrates that within widely applied tools such as Life Cycle Analysis (LCA), Life Cycle Costing (LCC) and Cost-Benefit Analysis (CBA), further sub-methodologies may be distinguished that focus on different aspects of sustainability: environmental LCA (eLCA), social LCA (sLCA), financial LCC (fLCC), environmental LCC (eLCC), full environmental LCC (feLCC), societal LCC (sLCC), financial CBA (fCBA), environmental CBA (eCBA) and social CBA (sCBA). Other methodologies such as Life Cycle Sustainability Assessment (LCSA) combine elements of different tools. Clearly, Figure 2 depicts the borders between the sub-methodologies in a simplified way. Indeed, sub-methodologies can contain more nuances and can be less easy to delineate as partially interrelated methodologies exist.

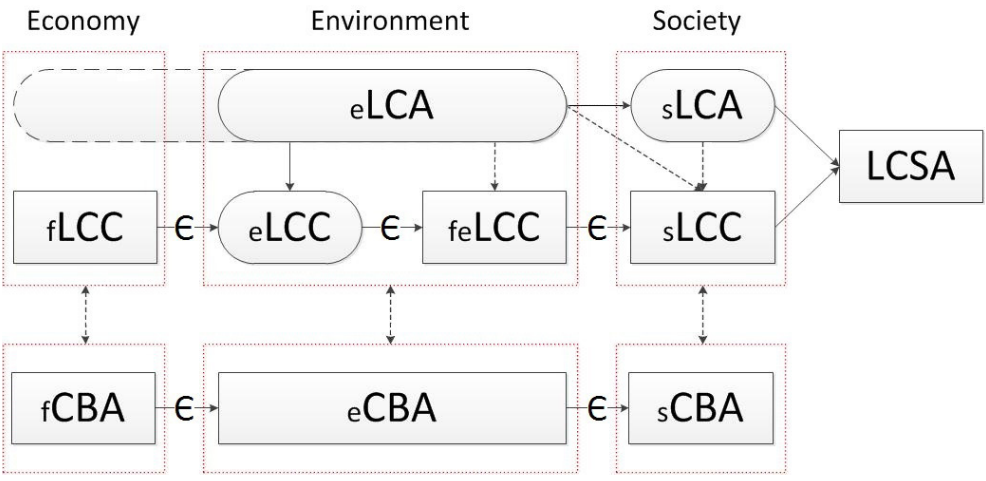

Figure 2: Integrated sustainability assessment framework
(Full arrows indicate that information from one methodology is needed to perform another methodology. Dashed arrows indicate that although a methodology can provide useful input to another one, both remain stand-alone methods. When '€' is included as a symbol, this indicates that a methodology is part of another one. eLCC for example builds further on fLCC.)

Although the proliferation of assessment tools allows analysis of all aspects of sustainability, it also induces confusion. Indeed, methodological differences and different weights for environmental, economic and societal priorities lead to conflicting assessment results for many important policy and business issues. Furthermore, while broadening the different methodologies to conduct sustainability assessments, fragmented developments by a variety of research disciplines led to blurred distinctions and as a consequence, synergies between tools become harder to identify. Carrying out a non-normative, descriptive review, this chapter presents the different sub-methodologies in the way they are generally used in research and discusses the methodological differences between LCA, LCC, CBA and their most relevant sub-methodologies. Based on the structural differences, a framework is developed that supports complementary use of the different methodologies.

Sections two, three and four handle the key characteristics and sub-methodologies of respectively LCA, LCC and CBA. Section five discusses key aspects in developing full sustainability assessment methods and connections between existing methods. Section six illustrates the methods and the framework by using the results of a study on automotive glass recycling which was carried out in Flanders (Belgium). The article concludes with a short discussion and an overview of the most important findings.

2.2. Life Cycle Analysis (LCA)

Although LCA-studies were already carried out in the 1960s, it was only in 1990 that SETAC initiated the standardisation process that led to the well-known ISO 14040+44 series (Bouman et al., 2000; Guinée et al., 2002; Tillman, 2000; UNEP, 2011). According to these standards, an LCA consists of four steps. In the first step not only the goal and scope of the study are defined but also the functional unit and system boundaries are selected. This functional unit relates to the product function rather than a particular physical quantity and is typically time-bound. In case of windows, the product function is to provide light and insulation and the functional unit therefore would be a square meter of glass with a certain life expectancy and insulation performance, rather than the weight of glass. In the second step, a life cycle inventory is made up which identifies and quantifies inputs and outputs at every stage of the life cycle. In the third step, the impact assessment phase, these inputs and outputs are categorised in different midpoint (climate change, land use, …) and endpoint (human health, resource depletion, …) impact categories (Finkbeiner et al., 2010; Jolliet et al., 2004; UNEP, 2011; Weidema et al., 2008). The aggregation and weighting of different environmental and social impact categories in order to compare different products is controversial. Indeed, weighting requires a subjective judgment on the priority of different impact categories (Benoit and Rousseaux, 2003; Daniel et al., 2004). In addition, weighting requires quantification (Rorarius, 2007; Ross and Evans, 2002), and as a result, the significance of impact categories that are hard to quantify, such as biodiversity and human health, will fade away in the assessment (Kloepffer, 2008). In the last step, the interpretation step, the results of the LCA should be interpreted. Furthermore, a sensitivity analysis checks the consistency of the results (UNEP, 2011).

In order to deal with different pillars of sustainability, two LCA types have developed that can be applied separately or in combination: eLCA and sLCA (Hauschild et al., 2008; Ramirez and Petti, 2011; UNEP, 2009). Although Figure 2 suggests that both sub-methodologies are well-delineated, overlaps may emerge. Below, both tools are discussed as stand-alone methodologies as they are generally used as such in research.

An environmental LCA (eLCA) is the conventional type of LCA that assesses environmental impacts such as material, energy and waste flows of a product from cradle to grave (Hunt et al., 1992). eLCA differs from assessment tools that focus on one environmental aspect such as 'Carbon Footprint' because its comprehensive environmental scope covers greenhouse gases, water emissions, ecosystems quality, natural resources and human health (Weidema et al., 2008).

Since firms are increasingly held responsible for causing social impacts, they are looking for tools such as social LCA (sLCA) that permit to make informed decisions about social impacts throughout the full life cycle of products

(Hauschild et al., 2008; Jorgensen et al., 2008; UNEP, 2009). sLCA is recently developed based on the already well-grounded eLCA. Being a rather immature tool, literature proposes different methodological alternatives focused mainly on differences in step 2 and step 3 of the LCA process (Dreyer et al., 2006; Hauschild et al., 2008; Hunkeler, 2006; Jorgensen et al., 2008; Weidema, 2006). sLCA is different than 'Corporate Social Responsibility' (CSR), because sLCA measures impacts along the full life cycle of a product while CSR is a management tool that typically focuses on the production phase. Moreover, sLCA uses information gathered at company, plant and process level while CSR only uses management information at company level (Ramirez and Petti, 2011).

Common properties of eLCA and sLCA are the four-steps procedure and the large data requirement. Differences include the approach with respect to people and social impacts. eLCA only considers damages on people if they are a consequence of impacts on the environment. On the other hand, sLCA embraces a broader understanding of human life and considers all social impacts and damages to people. Another aspect that is also dependent on the scope of the analysis, being cradle-to-gate or crade-to-cradle, is the inclusion of positive effects. Although an eLCA can contain positive impacts like avoided burdens and CO_2 uptake in the growth of plants for example, it often includes mainly negative impacts. sLCA however, usually addresses both positive and negative impacts. Furthermore, in comparison to eLCA, location and site-specific information is usually more vital in sLCA. The social impact of employment for example differs from region to region. Similarly, data should be inventoried by stakeholder category because social impacts often depend on the living conditions of these stakeholders (Dreyer et al., 2006; Hunkeler, 2006; UNEP, 2009).

2.3. Life Cycle Costing (LCC)

In the US, as early as 1933 life cycle costs were included in operating and maintenance costs when the General Accounting Office bought tractors. Although these life cycle costs were included, the term LCC was not yet used and LCC was actually not yet born as a methodology. It was only in the 1960s that an LCC as such was first used by the US Department of Defense for the acquisition of high-cost military equipment such as planes and tanks (Sherif and Kolarik, 1981). In Europe, the methodology has been used since the '70s to make policy and business decisions (UNEP, 2011). Typical LCC assessments compare durable products with a purchase price that only makes up a small part of the life cycle cost. Other costs over the lifetime of the product are discounted to current values (Asiedu and Gu, 1998; Gluch and Baumann, 2004; Kloepffer, 2008). Although discounting is a generally accepted practice, the applied discount rate is often controversial. In business circles high discount rates are applied such that current financial flows have a higher weight. In contrast, from a societal or environmental point of view, low discount rates are preferred to avoid that current activities impose large costs on future generations (Azar and Sterner, 1996; Rabl, 1996; Sáez and Requena, 2007; Weitzman, 1994).

In order to deal with financial, environmental and social concerns, four LCC types have been introduced: fLCC, eLCC, feLCC and sLCC (Finkbeiner et al., 2010). Although in literature combinations of these four LCC types exist, most researchers use these LCC types as stand-alone methodologies.

Conventional LCC assessments that only focus on private investments from one actor (a firm or consumer) are categorised as financial LCC (fLCC) (Ness et al., 2007; Rorarius, 2007). Generally, only costs borne by that actor matter, and environmental costs or external end-of-life costs are omitted. Consequently, an fLCC does not always consider the complete life cycle as only the economic lifetime matters (Hunkeler et al., 2008; Norris, 2001). Financial streams included in fLCC comprise for example investment costs, R&D costs and sales revenues (presented as negative costs) that are borne by the actor in question. Although the focus usually lays on these private costs, sometimes user costs can be included. For example, if companies are developing new products, they may take their customers' cost of ownership into account. It is common practice to discount the cash flows that occur within the economic lifetime. In an fLCC assessment, a quasi-dynamic approach is used. Quasi-dynamic models mainly contain variables that remain constant over time. For the variables that do vary over time, discounting is applied. A fully dynamic model would determine the key variables in an endogenous way, i.e. the variables change throughout time depending on the outcome of the model (Hunkeler et al., 2008).

An Environmental LCC (eLCC) builds upon data of fLCC and extends it to all costs borne by different actors present in the full life cycle of a product (Carlsson Reich, 2005). Consequently, eLCC considers the full life cycle of a product (Kloepffer, 2008). The focus remains, however, on real cash flows that are internalised or expected to be internalised. There is no conversion from environmental emissions to monetary measures. Examples of costs included in an eLCC are waste disposal costs, CO2 taxes that are expected to be implemented and global warming adaptation costs that may have to be paid by different actors within different stages of the full product life cycle. In contrast with fLCC, eLCC uses a steady state cost model in which all variables are kept constant over time. Technologies for example do not change. In addition, discounting is not applied (Hunkeler et al., 2008; Kloepffer, 2008).

The full environmental LCC (feLCC) is not a commonly accepted concept in the world of sustainability assessment tools but has been introduced to show in an explicit way that eLCC is in no way an equivalent of eLCA or eCBA. feLCC extends eLCC with monetised, non-internalised environmental costs that can be identified by an environmental assessment method such as eLCA. However, the transition to convert environmental impact figures to monetised measures is not always straightforward.

In Societal LCC (sLCC) all costs borne by anyone in society, whether today or in the future, and associated with the life cycle of a product are taken into account. Impacts such as public health and human well-being have to be quantified and translated into monetised measures, which is often a challenge in carrying out this type of assessment. As the analysis is carried out from a societal perspective, transfer payments like subsidies and taxes should be subtracted from the cost data as they have no net cost effect (Massarutto et al., 2011; Rus, 2010). sLCC applies a quasi-dynamic approach with discounting. Considering that sLCC is typically used from a societal perspective, low discount rates are mostly preferred (Rabl, 1996; Rambaud and Torrecillas, 2005).

2.4. Cost-Benefit Analysis (CBA)

CBA was already carried out in the early 20th century to assess the attractiveness of projects (Brent, 2009). In order to obtain a net present value (NPV), discount rate (*x*) applies to cash flows (CF) across *T* years as indicated by equation 1 (Kloepffer, 2008; Ness et al., 2007):

$$NPV = \sum_{t=1}^{T} \frac{CF_t}{(1+x)^{t-1}} \tag{1}$$

If the NPV is positive, the project achieves the imposed profitability requirements. Financial, environmental and social concerns have led to three CBA types: fCBA, eCBA and sCBA. Although one CBA can include all three concerns, typically only one or two aspects are taken into account (Rorarius, 2007). Below, the different CBA types are presented as they are commonly used in research. In literature however, a wider array of proposed (sub-) methodologies exists, including combinations of the three presented types.

Financial CBA (fCBA) is a tool for private profitability assessment. Only discounted cash flows of one actor are taken into account (Pearce et al., 2006; Rus, 2010). Only in a perfect market, fCBA would be sufficient to assess sustainability. Indeed, since market prices rarely reflect environmental and social values, eCBA and sCBA are needed for a more comprehensive view on sustainability.

In the late 1960s and early 1970s, environmental concerns led to the development of environmental CBA (eCBA) (Hunkeler et al., 2008; Weidema, 2006). Central to eCBA is the concept of external costs caused by environmental impacts. Expressing the damage in monetary values is often quite a challenge. Furthermore, the monetised impact is external to the producer because he does not bear that cost. Examples of external costs are pollution impacts, ecosystem losses and property damages to neighbors (Molinos-Senante et al., 2010; Pearce et al., 2006). As eCBA contains financial aspects, information of fCBA is integrated. The NPV will, however, only be positive if the financial gains do not induce excessive environmental losses.

Social CBA (sCBA) evaluates a project from the viewpoint of society as a whole. Although money is used as a common unit in which social and environmental costs and benefits can be expressed, the focus is on welfare. By using money as unit of account, welfare can be measured and compared across projects. Included benefits can be recreational benefits, reduction of air pollution, reduction of noise level and job creation for target groups that are weak on the labor market. Examples of costs are construction and maintenance costs, health impact costs, environmental impact costs and safety deteriorations. Monetisation of such benefits and costs is a challenge, especially when taking into account that beneficiaries and losers are individuals with different incomes, education and health. Likewise, non-tangible effects that are difficult to quantify are often not included which can lead to biased results. Although redistribution effects may require a social weighting to assess the welfare effect of projects, the Kaldor-Hicks potential compensation criterion remains central to sCBA: a project is positive for society as long as (after weighting) the overall NPV is positive (Bossert, 1996; Demange and Laroque, 1999; Krutilla, 2005). As sCBA comprises financial and environmental aspects, information of fCBA and eCBA is integrated.

2.5. Interactions between assessment tools

This section looks at key aspects that need to be considered when the sub-methodologies reviewed in the previous sections, are extended to integrated sustainability assessments. Again, it should be stressed that the concise overview might omit nuances of the different sub-methodologies. The connections that are of key importance to integrate these methodologies, are also discussed. Table 1 summarises these key elements. By focusing on the key aspects, underlying differences between the methodologies are disclosed.

Table 1: Key aspects and connections between sustainability assessment tools.

Key aspects	
LCA	metric, data requirements
LCC	scope, time frame, discounting
CBA	focus point, timespan, references, labor requirements, scarcity
Connections	
sLCA, sLCC, sCBA	distribution of impacts in society
CBA, sLCC	valuation methods and input
CBA, LCA	environmental impact assessment
LCA, fLCC	parallel use
eLCC, eLCA, sLCA	complementary

2.5.1. LCA versus LCC

Considering that both eLCA and sLCA lack an economic dimension (UNEP, 2009), an economic pillar should be added to achieve a full sustainability assessment method. This economic pillar can be found in the form of an eLCC. Combining these three tools, an LCSA method is composed. However, when combining the LCA tools with an eLCC, some key aspects have to be taken into account. First, the used metric is different. That is, an LCC expresses all units in monetary terms whereas LCA denominates flows by physical quantities such as mass, energy and volume. A second key aspect concerns the data requirements. In LCA, all environmental impacts of upstream processes have to be gathered to calculate the total environmental impacts of a particular product. In contrast, for LCC assessments, the price of a given process input can serve as a measure for the aggregated upstream costs, so detailed costs of upstream activities need not to be known (Hunkeler et al. 2008). As regards the scope, LCA assesses environmental impacts from a broad perspective, as does eLCC. Furthermore, since eLCC addresses the full life cycle of a product, it has a similar time frame

(Kloepffer, 2008). Finally, LCA and eLCC apply the same approach with respect to discounting as they both use a steady state approach that assumes that timing of an impact does not influence its weight.

As shown in Figure 2, an sLCC covers all aspects of sustainability including the economic, environmental and societal pillar. However, differences between the sub-methodologies constituting the building blocks of the full sustainable sLCC, have to be heeded. A first key aspect concerns the scope. A conventional fLCC only focuses on the perspective of one economic actor. When extending the fLCC to an sLCC, impacts from a broad societal perspective have to be integrated. Related with this aspect, is the difference in time frame. fLCC does not always consider the complete life cycle as only the economic lifetime matters (Hunkeler et al., 2008; Norris, 2001). When extending the fLCC to an sLCC, all phases in the life cycle of the product should be included. A last key aspect is the use of discounting. In contrast to eLCC, both fLCC and sLCC use a quasi-dynamic approach in which discounting is applied such that current monetary flows matter more than future monetary flows. This different approach with respect to timing makes it hard to compare results of the different tools (Hunkeler et al., 2008; Ness et al., 2007). As regards the used metric and the data requirements, all LCC types use monetary expressions and have similar data requirements.

2.5.2. LCA and LCC versus CBA

As are LCA and LCC, Figure 2 indicates that a CBA is able to carry out full sustainability assessments. However, there are key aspects that have to be taken into account.

The first dissimilarity between LCA and LCC on the one hand and CBA on the other concerns the focus point. LCA and LCC can be catalogued as product related assessments while CBA mostly focuses on projects or policies (Ness et al., 2007; Rorarius, 2007). In some cases however, the borderline between a product and a project can be vague. For example, if the study of a road focuses on the negative externalities that are caused by exhaust gases, the road is taken to be a product. However, when investigating whether a road should be constructed or not, the construction of the road can be seen as a project and a CBA would be appropriate. Future methodological developments can cause the borderline to become even more vague. A second key aspect is the timespan. LCA and LCC focus on the (full or economic) life cycles of the assessed products while CBA, focusing on the lifetime of a particular project, makes the lifetime of used products secondary. Although LCA will often make the capital equipment used to produce the products secondary, sometimes there is a broader issue of system boundaries. In fact, research indicates that capital goods cannot always be excluded (Frischknecht et al., 2007). For newsprint paper for example, Kasah (2013) shows that for most eLCA categories, the use phase of a paper machine is the most important life cycle stage. A third key aspect relates to the kind of reference. LCA and LCC are comparative assessment tools that compare different products having the same functional abilities. Being typically used for project evaluations however, CBA calculates an NPV value that always has an economic meaning compared to a zero reference. A fourth key aspect shows itself in the perspective on labor requirements. In sLCA and sLCC, job creation is usually regarded as a benefit. For example, an sLCA about recycling options for Waste from Electrical and Electronic Equipment (WEEE) that was carried out in China, states that in terms of employment creation, informal waste collection systems are better than formal ones because they require more manual labor (UNEP, 2011). An sCBA however, considers labor as a cost. If jobs are created in labor markets with high unemployment rates, job creation can have positive external effect. If, in contrast, new jobs are created for already highly demanded skilled workers, negative external effects can arise in the economy (Bartik, 2012; Masur and Posner, 2012). In addition, it is important to compare a project with the alternative use of the resources. So, when the capital invested in a project generates a number of new jobs, one has to ask what the alternative use of the same capital could have generated elsewhere in the economy (Berck and Hoffmann, 2002; Harberger, 2008). A last key aspect is the way in which scarcity is dealt with. LCA sees scarcity as a societal problem and counts recycling or use reduction of a scarce material as a benefit. As CBA focuses on external effects, scarcity is not necessarily a problem. Indeed, since scarcity leads to high prices, economic markets typically internalise scarcity. In this way, prices represent the full cost to society if the private discount rates are comparable to the social optimal level. However, markets typically use higher discount rates than those that are used from a societal perspective (Pearce, 1998; Pearce et al., 2006; Powell et al., 1998).

2.5.3. Connections between LCA, LCC and CBA

In order to integrate methodologies to full sustainability assessments, connections between them are of key importance. The connections discussed in this section are depicted by the lines drawn in Figure 2.

Although sLCA, sLCC and sCBA differ in methodology they can all deal with the unequal distribution of impacts in society. For example, sLCA would attribute a high weight to an income increase in low-income regions by weighting impacts with the inverse of the income level relative to the societal average. As explained earlier sCBA can also give more weight to positive or negative impacts on a low-income group (Jorgensen et al., 2008; Weidema, 2006).

CBA and sLCC have inspired each other mutually. sLCC has not only taken over important techniques of sCBA, such as valuation methods like hedonic pricing or contingent valuation, but it also regularly serves as input in CBA (Hunkeler et al., 2008; UNEP, 2011).

An LCA assessment is often called the physical counterpart of the environmental impact analysis that is required for eCBA and sCBA. Both LCA and CBA however have a connection with Environmental Impact Assessment (EIA). CBA goes a step further than EIA by putting money values on the environmental impacts. As regards LCA, EIA can be complementary to LCA since it provides further and more detailed information about the analysed object. On the one hand, LCA can go further than EIA by looking not only at the impacts that directly arise from a product, but also at the impacts of the whole life cycle (Pearce et al., 2006; Tukker, 2000). At the other hand, as LCA permits the inclusion of these upstream and downstream activities, this information can be crucial for strategic EIAs where environmental comparisons of different alternatives have to be made (Pennington et al., 2004; Manuilova et al., 2009).

As shown by the lines in Figure 2, eLCA has common features with all LCC types, except fLCC. As double counting is not a problem, LCA and fLCC can be used in parallel. This is shown by the dashed eLCA extension in Figure 2. Although the information resulting from both assessments is complementary, the results may point to different actions. Indeed, an environmental measure may be positive for the environment, but negative from a purely financial point of view. Two methodological differences may complicate the interpretation of the results. Firstly, the perspective between both methodologies is different (private point of view versus life cycle point of view) and secondly, fLCC uses a quasi-dynamic approach with discounting whereas eLCA uses a steady state methodology where the weight of the impacts in time remains constant.

An important aim of a sustainability assessment is to include the three pillars of sustainability. Our analysis indicates that eLCC, eLCA and sLCA do not only address the three pillars but are also complementary in methodology and avoid double counting. eLCC and eLCA do not only define system boundaries, time span and functional units in a similar way, they also share the steady state approach without discounting of impacts. This is important, as eLCC is primarily set up as an assessment method that is carried out in combination with eLCA. As an eLCC only includes real money flows, the risk for double counting with environmental impacts included in eLCA is minimised (Hunkeler et al., 2008). By applying the same system boundaries, time span, functional unit and steady state cost model as eLCA and eLCC, also an sLCA is compatible. The relatively new and comprehensive tool that summarises the results of these three sub-methodologies is called LCSA (Hunkeler, 2006; Klopffer and Ciroth, 2011; Swarr et al., 2011). Combining the results, an LCSA can provide more comprehensive insights to invest limited societal resources in an optimal way.

2.6. Case study: treatment of end-of-life automotive glass

The treatment of end-of-life automotive glass has recently sparked interest in business and policy circles. The European directive for End-of-Life Vehicles (ELV) (2000/53/EC) imposes high weight-based recycling rates for ELV by 2015. Since ELV glass is about 3 % of the weight of a car, policy makers in different European Member States are studying potential recycling of this material stream. In order to recycle ELV glass, it has to be dismantled before the remaining ELV is shredded to extract the metals. A study by Farel et al. (2013) on the costs and benefits of ELV glass recycling found that, in certain scenarios, a recycling network could become financially beneficial and could be self-sustaining. A study by Badino et al. (1997) found that the Italian glass recycling system is positive from an environmental point of view. In Belgium, the Public Waste Agency of Flanders (OVAM) has ordered a study (OVAM, 2013a,b) to assess which policy approach regarding glass recycling would be most sustainable. This case study will be used as an example in this paragraph as it includes an eLCA as well as an fCBA part.

In the study, two different recycling routes for automotive glass are compared (OVAM, 2013a,b). The first recycling route involves the depollution and shredding of ELV without prior glass dismantling. Subsequent use of post shredder treatment extracts the glass from the landfilled fraction such that it can be valorised in low-value construction applications. The second recycling route dismantles glass before ELV shredding such that the glass can be recycled. To examine environmental and economic aspects, eLCA and fCBA were carried out. These two assessment methodologies are shown in Figure 3, together with sub-methodologies that can be carried out in order to adopt a full sustainability perspective. For each sub-methodology a non-exhaustive list of data elements is shown that refers to the glass recycling case.

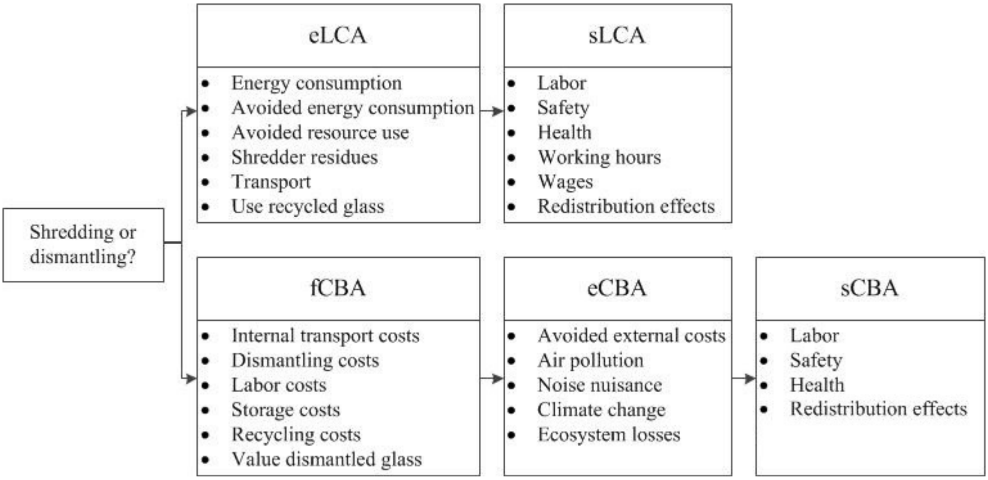

Figure 3: Flow diagram glass recycling assessment

For eLCA a gate-to-grave study is performed to compare the scenarios. The functional unit is one kg of glass from a depolluted ELV. In the study, no comparison is made between different products, rather, a comparison is made between treatment types of one product. As shown in Figure 3, to make up the life cycle inventory, detailed information is needed about the shredder, the post-shredder treatment and the glass recycling routes: energy consumption, avoided consumption of energy and resources from recycling, composition and volume of shredder residues and transport. For the glass recycling step, inventory data are gathered for each possible application of recycled glass: use in float glass for new automotive windows, use in foam glass or glass wool for insulation material or use in fiberglass. The eLCA indicates that the shredding route has minor environmental impacts and benefits depending on the impact category, while dismantling in all categories has a much larger environmental benefit. Concerning climate change for example, recycling one ton of ELV windows by shredding yields a net emission impact of 10kg CO_2 equivalents, while the dismantling route would yield a net reduction of 280kg CO_2 equivalents. Furthermore, when looking at fossil depletion, recycling one ton of ELV windows by shredding yields a net emission impact of 3,4kg oil equivalents, while the dismantling route yields a net reduction of 46kg oil equivalents (OVAM, 2013a).

A detailed overview of all environmental impacts for both recycling routes can be found in Appendix A.

The fCBA studies the costs of the two recycling routes. In line with the short time horizon of both recycling routes, discounting is not applied. Table 2 shows the fCBA results for both recycling routes. As can be seen, the total marginal cost for the glass dismantling route is estimated at €213/ton of glass. The highest cost, being the dismantling cost of €178/ton respresents 75% of the total cost. This cost consists of €57 moving the ELV and placing the protection layer costs, €109 dismantling time costs, €5 work tool costs and €7 work space costs. Depending on the glass quality and the supply and demand, the glass recycling step generates a net benefit of €10/ton glass. With respect to the total marginal cost of the shredding route, this is estimated at only €49/ton of glass. Marginal shredder costs only include maintenance costs due to abrasion and extra staff costs. The transport costs are low as well, as the distances are relatively low. As regards the glass recycling step, in the shredding route automotive glass ends up in the mineral fraction after PST. This fraction is then recycled as lower-grade building material, making the glassrecycling less valuable than in the dismantling route (OVAM, 2013b).

Table 2: Overview marginal glass recycling costs (€/ton glass), (OVAM, 2013b).

Shredding route		**Dismantling route**	
Shredder	4	Dismantling	178
Transport to shredder	3	Storage + transport to glass recycler	45
PST	34	Glassrecycling (net benefit)	-10
Transport to recycling	7	Total	213
Glassrecycling	2		
Total	49		

The results of the eLCA and fCBA point to other policy recommendations. Indeed, the eLCA indicates that policy makers should stimulate dismantling because it is good for the environment while the fCBA indicates that the marginal costs of the dismantling route are much higher than those of the shredding route. Although both assessments have been done in a rigorous way, the conflicting results lead to ambiguity and confusion.

Even though conflicts between priorities of the three sustainable pillars are not easy to overcome, the review suggests actions that can contribute to a more comprehensive understanding of the problem. First, as shown in Figure 3, the fCBA can be extended to an eCBA by integrating monetised environmental impacts that are related to the treatment of ELV glass: avoided external costs of primary material extraction, air pollution, noise nuisance, climate change and ecosystem losses. An eCBA would deal with both the environmental and financial aspect of ELV glass in one coherent methodology. Second, as dismantling is typically executed by low-skilled labor, the job creation may create positive externalities. Conversely, safety is an often-used argument against dismantling, especially when it is done manually as glass splinters can cause injuries. With respect to the scarcity issue, dismantling could be preferred over shredding, as in that case the recycled glass could be reused in the production of higher-grade products, thereby reducing the need for virgin material and fossil fuels for example. Consequently, an sCBA could include aspects such as shown in Figure 3 and could assess the environmental, social and financial aspects of ELV glass in one coherent methodology. Third, considering that the fCBA already contains an inventory of costs and benefits per kg of automotive glass, the assessment can be easily transposed to the LCC methodology. The inventory of environmental impacts that has been drawn up for the LCA analysis could be used to extend the fLCC to an feLCC. If social and labor aspects are also integrated, an sLCC allows assessing the environmental, social and financial aspects of ELV glass in one coherent methodology. Fourth, as discussed, the existing fCBA can be transposed to the LCC methodology, more specifically eLCC. In addition, the existing eLCA can be extended to an sLCA if social aspects such as labor, safety and health are included. As indicated by the review, eLCC, eLCA and sLCA complement each other. The combination of the three tools leads to a comprehensive LCSA that takes all three pillars of sustainability into account.

2.7. Discussion and conclusions

Although the proliferation of sustainability assessment tools makes more information available on important aspects of sustainability, it also creates ambiguity and confusion. As illustrated by the results of the Flemish study on automotive glass recycling, assessment via different methodologies may lead to conflicting policy recommendations. In order to interpret and integrate results of different methodologies, a better understanding is needed of the relations between the existing assessment tools. Furthermore, the key aspects discussed in this chapter should be taken into account when future efforts are made to broaden LCA, LCC and CBA to full sustainability assessments. Based on a review of LCA, LCC and CBA methodologies, this chapter presents a framework that shows the interrelationships between these existing assessment tools.

Key aspects that need to be considered when LCA is adapted to a full sustainability assessment are: (i) the metric to report results and (ii) the data that are required. To broaden LCC, key aspects are: (i) the scope of the assessment, (ii) the time period taken into account and (iii) the use or absence of discounting to deal with long time horizons. As are LCA and LCC, a CBA is able to carry out full sustainability assessments. However, there are some key aspects that have to be taken into account: (i) the focus point, (ii) the timespan, (iii) the need for a reference scenario, (iv) the way to deal with labor requirements and (v) the approach towards scarcity as a policy concern. In addition to these aspects, there are connections between the sustainability assessment tools that are of key importance. The logic applied to integrate the social pillar in the sustainability tools (sLCA, sLCC, sCBA) is comparable. CBA and sLCC share key techniques such as valuation methods to monetise external impacts. Both LCA and CBA have a connection with Environmental Impact

Assessment (EIA). Furthermore, LCA and fLCC can be applied in a parallel way. Although their results cannot be integrated they provide information on different aspects of sustainability without double counting. Finally, the methodologies of eLCC and eLCA/sLCA are sufficiently similar such that they can be used in a complementary way. The summation of the results of these three tools leads to the comprehensive LCSA assessment that includes the three pillars of sustainability.

By discussing the key aspects and the relations between the different tools, this chapter contributes to the understanding of conflicting assessment results and to the harmonisation of assessment methodologies. Moreover, the key aspects should be taken into account when further efforts are made to broaden existing assessment tools to full sustainable ones. Further research and permanent vigilance is needed to develop more comprehensive tools that integrate the three pillars of sustainability. Methods like 'Ecocosts 2012', 'Ecovalue08' and 'Stepwise 2006', that allow valuing outputs of a life cycle inventory by converting them to eco-costs, show that integration is not an insurmountable problem. However, such methods should be further developed before one can speak of a fully integrated assessment methodology.

CHAPTER 3.

Identifying the effects of landfill taxes on scarce landfill capacity. A simulation using dynamic optimisation modelling and Flemish (Belgian) data[2]

[2] This section is based on Hoogmartens, R., Dubois, M., Van Passel, S. (2014). Identifying the interaction between landfill taxes and NIMBY. A simulation for Flanders (Belgium) using a dynamic optimisation model. This article was published as a book chapter in 'Legal Aspects of Sustainable Development: Horizontal and Sectorial Policy Issues' and presented at amongst others the ISDRC 2014 Conference: International Sustainable Development Research Conference, Trondheim - Norway, 18/06/2014 - 20/06/2014.

Abstract

In the past, landfills were emerging at an increasing pace in order to deal with growing waste generation. The negative externalities that are caused by these landfills however, together with concerns about spatial scarcity led to the awareness that volumes of landfilled waste had to decrease. As a result, restrictions on remaining landfill capacities emerged which causes remaining capacity to be regarded as non-renewable and scarce. In this chapter, a dynamic optimisation model is constructed to assess the evolution of landfill volumes and landfill prices in time. Carrying out example simulations using Flemish (Belgian) data, landfill paths and price paths were constructed for two different scenarios. In the first scenario, landfill taxes are taken up in the model, whereas these taxes were omitted from the model in scenario two. As the simulation results show, when landfill taxes are levied, it takes 42 years for landfill exhaustion to occur. When no landfill taxes are being used, this period would be shortened to only 20 years. Therefore, in our simulations a landfill tax has the effect that yearly landfilled volumes decrease considerably. In addition, when landfill taxes are used, discounted total welfare increases significantly.

3.1. Introduction

In the 1960s, as a result of mass production and growing consumption that led to a steep incline in waste generation, landfills were emerging everywhere. Starting in the 1970s, public attitude towards waste started to change as people realised the negative environmental externalities caused by landfilling and the valuable space it occupies (Strasser, 1999; Walsh, 2002; Van Passel et al., 2013). A well-known and often used instrument to internalise external effects such as noise, odor, groundwater pollution and air emissions, is a landfill tax.

Comparing Flanders with other European countries, we find a wide variety of diverging taxation rates with Flanders belonging to those regions that apply the highest rates. As Europe is moving towards an open market for waste management, calls are made to harmonise waste policies across borders in order to prevent, inter alia, a so called ‘race to the bottom’ (Dubois, 2013).

Considering that several front runners in waste management, including Flanders, have high landfill taxes, the arguments for high and harmonised landfill taxes in Europe seem strong. In addition, high landfill taxes directly target the lowest level of the Waste Hierarchy by raising the cost of landfilling such that other waste treatment methods become more attractive (Calcott and Walls, 2005; Dinan, 1993; IVM, 2005; Watkins et al., 2012; Bio IS, 2012). Although there is some evidence on the effectiveness of a landfill tax to reduce landfilling (Monier et al., 2011; OECD, 2012; Oosterhuis et al., 2009), not all economic scholars are convinced that high landfill taxes are justified (Dijkgraaf and Vollebergh, 2004; Dubois, 2014; Eshet and Shechter, 2005; Kinnaman, 2006). These scholars argue that external costs of modern sanitary landfills with methane extraction are rather low (€5-€30/ton). A Pigovian tax would therefore be positive, but typically lower than current landfill tax rates in Belgium or in the UK (CEWEP, 2012). The question then arises how to reconcile the economic perspective with policy discourses.

A historic element caused by spatial scarcity and negative externality concerns is the 'Not In My BackYard' syndrome (NIMBY). Although this syndrome has not been taken into consideration in policy decisions consciously, its occurrence actually played a constraining role in the failure to issue new permits for landfills. Together with the fact that vacant land is scarce and therefore expensive, this moratorium on new landfill zones made remaining landfill reserves a scarce good in densely populated regions in Western Europe (Levinson, 1999; Van der Zee et al., 2004). Economic theory predicts that scarce non-renewable goods will increase in price such that consumption will be restrained. Remarkably, this effect is also the aim of a landfill tax, so attention has to be paid to the effects landfill taxes have on scarce remaining landfill capacities.

In view of the above, this chapter handles following research questions:

- What are optimal landfill and price paths, and when will landfill capacity be exhausted?

- Knowing that remaining landfill capacity is scarce, what are the effects of landfill taxation?
- Can a landfill tax above the landfill externalities be justified?

To gain insight into these research questions, a dynamic economic model was developed. As the model should identify the best allocation of landfill volume over time, the model was set up as a dynamic optimisation problem. Focusing on the simulation of economic models, this chapter does not address non-economic factors other than the potential impacts of applying landfill taxes. When supporting decision-making in practice, also non-economic and societal factors should normally be included. To illustrate the theoretical approach, Flemish data were used to perform example simulations.

In the second section, all elements of the dynamic optimisation model are discussed and a theoretical model is built up. Based on these theoretical underpinnings, illustrative example simulations are carried out in the third section. The article concludes with a discussion and an overview of the most important findings.

3.2. Dynamic optimisation model

Numerical optimisation problems are known to serve at least two functions. First, they make theory and methods less abstract and more meaningful and secondly, they can serve as a bridge from theory and general models to actual analyses of real-world allocation problems (Conrad, 1999). Figuring out optimal landfill and price paths, and effects of influencing variables, requires a dynamic optimisation model to be elaborated. Below, all elements the optimisation model is based on, are discussed.

3.2.1. Landfill demand

To map the aggregate landfill volume demanded to market price, a linear inverse demand function is used. Using this basic formulation for the demand function allows a straightforward determination of the choke-off price and makes it possible to present a detailed stepwise overview of the model setup. This way, it should become clear how the model works and what the meanings are behind the model-based derivatives. Drawing on the basic formulation, the next chapters will develop a more elaborated version. In this chapter, we write $p_t = D(Q_t)$, where p_t is the price per ton landfilled in period t given that an aggregate landfill volume of Q_t is supplied to the market. We will assume that price decreases with increases in Q_t (so $D'(Q_t) < 0$). In our model, the inverse demand function is given by:

$$p_t = D(Q_t) = a - bQ_t \tag{2}$$

with: p_t = price in year t (€/ton)

Q_t = volume landfilled in year t (million ton)

a = choke-off price, intercept on price axis (€/ton)

b = slope of the inverse demand function

An important characteristic of the linear demand curve is the implied maximum choke-off price at the intercept $p_t = a$ when Q_t is equal to zero. Such an upper bound may result from the existence of a substitute, available at constant marginal cost MC = a. In scheduling landfill volumes, each competitive firm is assumed to know about this backstop substitute and to know what price will reach the intercept when the full remaining landfill volume has been exhausted. As substitutes for Flemish landfilling, one can think of an increase in waste export or an increase in the waste recycling rate. In the model, we assume exhaustion occurs in t = T. At that time, remaining landfill volume falls to zero ($Q_T = 0$). The date of exhaustion, T, is unknown and must be determined along with the competitive landfilling and price paths.

3.2.2. Competitive landfill companies

In the model, it is assumed that there exists a competitive landfill industry facing a linear inverse demand curve for aggregate landfill volume, Q_t. The landfill companies are maximising their profits, so they will try to offer landfill volume so as to:

$$\text{Maximise}_{Q_t}\, \pi = \sum_{t=0}^{T} \beta^t * \pi_t = \sum_{t=0}^{T} \beta^t * [p_t - c - l]Q_t \tag{3}$$

$$\text{s.t.} \sum_{t=0}^{T} Q_t = S_0$$

with: π_t = profit in year t (€)

c = landfill cost (€/ton)

l = landfill tax (€/ton)

S_0 = Remaining landfill capacity in year 0 (million ton)

β = 1/(1 + δ) and δ is the discount rate

The Lagrangian for this problem may be written as:

$$\text{Maximise}_{Q_t} L = \sum_{t=0}^{T} \beta^t * [p_t - c - l]Q_t - \lambda\left[\sum_{t=0}^{T} Q_t = S_0\right] \quad (4)$$

The first-order-conditions require:

$\frac{\partial L}{\partial Q_t} = \beta^t * [p_t - c - l] - \lambda = 0$ or:

$$\beta^t * [p_t - c - l] = \lambda \quad (5)$$

Considering that we work with an inverse linear demand, it is straightforward to determine the choke-off price a. The price in period T (the last period of landfilling) will be equal to this choke-off price. So:

$$p_T = a \quad (6)$$

Equality (6) can then be inserted into (5) to determine λ, the shadow price of volume restriction. λ can be regarded as an economic measure of scarcity which is different from standard measures based on physical abundance. From an economic point of view, scarcity should reflect the marginal value net of the marginal costs associated with landfilling. Filling in (6) into (5) gives us:

$$\beta^T * [a - c - l] = \lambda \quad (7)$$

In order to assess the yearly volumes landfilled, we insert (2) into (5) and we get:

$$\beta^t * [a - bQ_t - c - l] = \lambda. \quad (8)$$

If we rewrite (8), we get:

$$Q_t = \frac{a - c - l - \lambda\beta^{-t}}{b} \quad (9)$$

Again, rewriting (9) results in:

$$Q_t = \frac{[1-\beta^{T-t}] * [a-c-l]}{b} \qquad \text{if } \lambda > 0 \qquad (10)$$

With:

$$S_0 = \sum_{t=0}^{T-1} \frac{[1-\beta^{T-t}] * [a-c-l]}{b} \qquad (11)$$

Making use of equation (11), we can calculate a value for T, so we can estimate how long it takes before all remaining landfill volume will be exhausted.

3.2.3. Welfare calculation

From a societal point of view, there are two more major aspects that are related to landfill taxation and landfilling in general. First of all, landfilling has some negative externalities, like for example noise, odor, groundwater pollution and air emissions. As these externalities have an impact on society, they carry a cost with them, which is called an externality cost. In the model, the unit externality of landfilling is presented by parameter e, whose value is strictly larger than zero. The second aspect relates to the Marginal Cost of Public Funds (MCPF). Government revenues are typically expensive for society because of tax dodging and administration (Barrios, 2013; Glomm et al., 2008; Schob, 1997). In contrast, landfill taxes are way easier to monitor, especially in developed regions such as Flanders. Indeed, there are only a few landfill sites and landfill monitoring would happen regardless of the fact whether taxes apply or not. The Marginal Cost of Landfill Taxes (MCLT) is therefore low. This gives:

$$\chi = \text{MCPF} - \text{MCLT} > 0 \qquad (12)$$

As χ represents the relative benefit of using landfill taxes instead of other, more expensive taxes like for example a labour tax, this supports the double dividend hypothesis and fits in with the green tax shift debate (Groth and Schou, 2007; Schob, 1997). Taking into account all this, equation 13 shows the total welfare function used in this chapter. In this equation, parameter B represents the net residual effect of landfill volume after year T, which can include externalities from the landfill after it has been closed for example. In the illustrative example below, this parameter is set equal to zero.

$$W = \sum_{t=0}^{T} \beta^t * [\int D(Q_t) - c - e + \chi l] Q_t + B \qquad (13)$$

In the next section, all of the foregoing formulas will be used in an illustrative simulation. With these formula, values are defined for T, p_t, Q_t, π_t and W. All other variables are defined exogenously.

3.3. Illustrative example simulations

Carrying out example simulations to illustrate the theoretical approach, data from the Flemish part of Belgium were used. Making use of equation (11) and using exogenously determined values for parameters a, c, l, b, e, δ, χ and S_0, a value can endogenously be determined for parameter T. The exogenous values that were used, are presented in Table 3.

Table 3: Exogenous parameters

Parameter	Value	Parameter	Value
a	100	e	50
c	15	δ	0.05
l	60	χ	0.2
b	50	S_0	12

Using the choke-off price and taking into account observed landfill volumes and prices from previous years (Briffaerts et al., 2011), the linear inverse demand function was estimated to be:

$$p_t = 100 - 50\, Q_t \qquad (14)$$

Based on a study carried out by OVAM (2013c), the remaining landfill volume in period 0 was taken to be 12 million tons, so the value of parameter S_0 was set to 12. As to the value of parameter χ, a cautious, rather conservative estimate was made taking into account the significant influence of the parameter on welfare calculations, the fact that the double dividend hypothesis is often only marginally true and results of previous research (Goulder, 1994; Carraro et al., 1996; Barrios et al., 2013; Danthurebandara et al., 2015a). Below, the simulation results are given for two scenarios, one with and one without using landfill taxes.

3.3.1. Scenario 1: simulation with landfill taxes

By assigning parameter l a positive value, a landfill tax is directly taken up in the simulation exercise. By solving the dynamic maximisation problem and using equation (11) to define a value for parameter T, results like presented in Table 4 can be obtained.

Table 4: Simulation with landfill taxes

t	Q_t	p_t	π_t (discounted)	W_t (discounted)	λ
0	0.43	78.33	1,442,349.28	15,675,116.13	3.33
1	0.43	78.49	1,431,276.22	14,848,151.02	3.49
2	0.43	78.67	1,419,649.51	14,060,024.74	3.67
3	0.42	78.85	1,407,441.46	13,308,860.79	3.85
4	0.42	79.04	1,394,623.00	12,592,870.68	4.04
5	0.42	79.25	1,381,163.63	11,910,349.67	4.25
…					
38	0.08	96.25	249,674.83	530,221.44	21.25
39	0.05	97.31	178,968.04	366,227.93	22.31
40	0.03	98.43	104,725.92	206,592.46	23.43
41	0.01	99.60	26,771.69	50,934.97	24.60
42	0	100	0	0	25
$\sum_{t=0}^{T}$	12		39,931,451.13	239,109,115.1	

As can be deduced from Table 4, the value of T satisfying equation (11) is T = 41.33. In a discrete-time problem such as this, where T must be an integer, we round T up to 42. This means that it takes 42 years for exhaustion to occur. The bottom row in Table 4 shows the total landfilled volume, the discounted total profit of the landfill companies and the discounted total welfare. Remember that λ can be regarded as an economic measure of scarcity. In the above model, λ is the value of marginally loosening the constraint, that is, increasing the landfill capacity. Given the results in Table 4, one could numerically plot the time paths for landfill volume and landfill price. These paths are respectively shown in Figure 4 and Figure 5 and show how landfill volumes and landfill prices change in time.

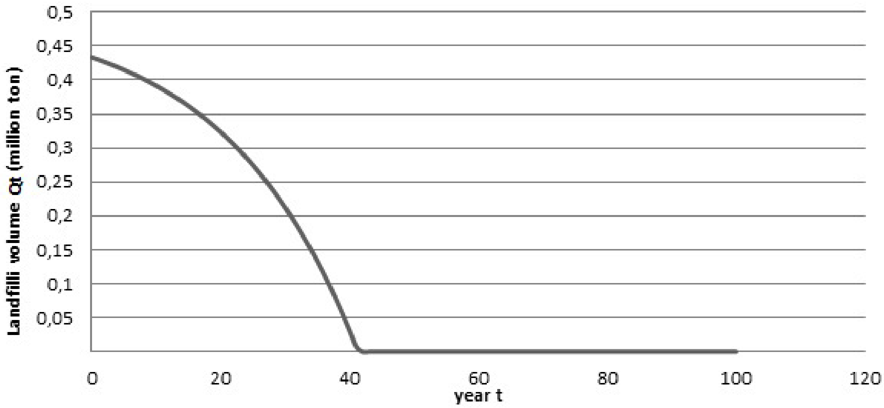

Figure 4: Landfill path when using landfill taxes

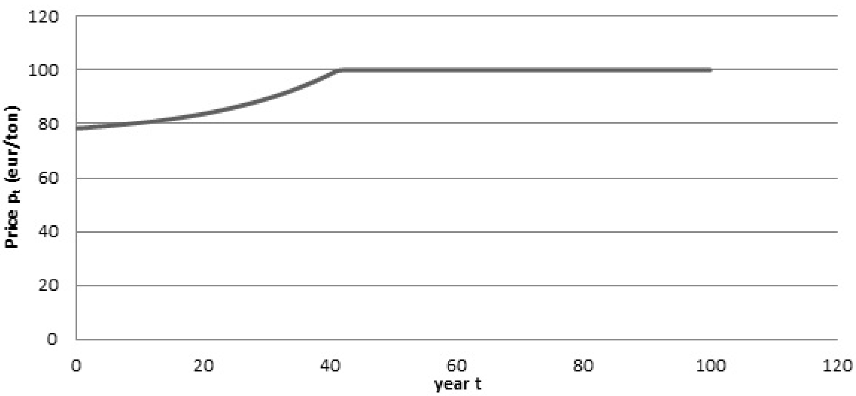

Figure 5: Price path when using landfill taxes

3.3.2. Scenario 2: simulation without landfill taxes

By setting the value of parameter l equal to zero, landfill taxes are left out of the model. When we then solve the dynamic optimisation problem and use equation

(11) to define a value for parameter T, results like presented in Table 5 are obtained.

Table 5: Simulation without landfill taxes

t	Q_t	p_t	π_t (discounted)	W_t (discounted)	λ
0	1.03	48.27	34,419,977.41	9,450,015.63	33.27
1	1.00	49.93	33,313,294.35	9,503,816.96	34.93
2	0.97	51.68	32,151,277.15	9,501,039.65	36.68
3	0.93	53.51	30,931,159.08	9,441,677.08	38.51
4	0.89	55.44	29,650,035.11	9,325,587.93	40.44
5	0.85	57.46	28,304,854.94	9,152,495.77	42.46
…					
16	0.25	87.62	8,238,632.16	3,268,421.10	72.62
17	0.18	91.25	5,822,881.85	2,338,685.08	76.25
18	0.10	95.06	3,286,344.01	1,335,310.58	80.06
19	0.02	99.06	622,979.29	255,908.62	84.06
20	0	100	0	0	85
$\sum_{t=0}^{T}$	12		399,202,148.3	132,563,771.7	

Looking at Table 5, we see that the value of T satisfying equation (11) has decreased from T = 41.33 in the case of using landfill taxes, to T = 19.23. As before, we round T up to 20. This means that it takes 20 years for exhaustion to occur. When we compare Table 4 with Table 5, we see that a landfill tax substantially reduces the volumes that are landfilled each year. For example, when a landfill tax would be applied, the total volume landfilled in the first six years would add up to only 2,547,316 tons, whereas this total volume landfilled would add up to 5,674,436 tons when no landfill tax is levied. When landfill taxes are used, prices start at a much higher level than when no landfill taxes would be applied, but they move less rapidly to the choke-off price level. As expected, the landfill companies' discounted total profits are much higher when no landfill taxes have to be paid. However, when we look at the simulated discounted total welfare figures, we see that the use of landfill taxes is preferable within our simulation structure. After all, the simulated total welfare figure is much higher in the scenario where landfill taxes are applied. This difference is mainly related to the parameter χ, which takes into account that the MCPF is larger than the MCLT. The parameter λ can still be regarded as the value of marginally increasing the available landfill capacity. In this scenario where no landfill taxes are applied, the value of this scarcity indicator is higher than in the case where landfill taxes were used. This is quite logical, taking into account that the remaining landfill capacity is depleted at a higher rate when no landfill taxes are being used, making the remaining stock more scarce and valuable.

Given the results in Table 5, the time paths for landfill volume and landfill price can numerically be plotted. These paths are respectively shown in Figure 6 and Figure 7 and show how landfill volumes and landfill prices change in time, when no landfill taxes are applied. Based on these paths, and by comparing them to Figures 4 and 5, the same conclusions can be drawn as were discussed in the description of Table 5.

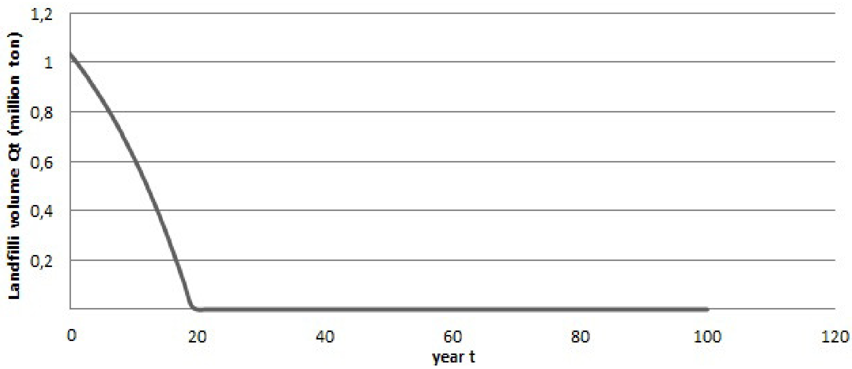

Figure 6: Landfill path without using landfill taxes

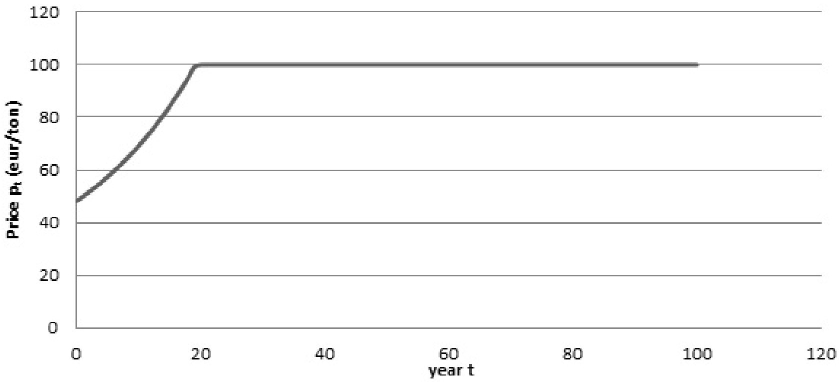

Figure 7: Price path without using landfill taxes

Using the dynamic optimisation model, it can further be investiged whether a landfill tax above the landfill externalities would be justified. After all, one could expect, looking at the MCPF, that a landfill tax above the landfill externalities can be justified. In our simulation model, this effect is not as large as expected. Only when minor tax increases are implemented, discounted total welfare increases. However, a turning point will be achieved from where discounted total welfare starts to decrease. Eventually, discounted total welfare will be lower than the value initially started from. In this regard, the objective of maximising the value of equation (13) results in an optimal landfill tax of 55€/ton. Compared with the results of the scenario in which the landfill tax is equal to 60€/ton, the optimal landfill tax also results in a higher profit figure (60,219,542 vs. 39,931,451) and shortens the period until landfill capacity exhaustion with five years (T=37 vs. T=42). Would the discount rate be increased to 0.10, the optimal landfill tax that maximises welfare would be equal to 52€/ton. Due to the higher discount rate, remaining landfill capacity is depleted at a faster rate than before, causing parameter T to be equal to 28 years. Correspondingly, profit and welfare figures are lower than before, being equal to €28,969,538.43 and €170,679,310.70 respectively.

3.4. Discussion and conclusions

As well-known and often used instrument to internalise external effects of landfilling, landfill taxes can be also used to increase the cost of landfilling such that other waste treatment methods become more attractive. Although there is evidence that a landfill tax can be effective in reducing landfilling, not all economic scholars are convinced that high landfill taxes are justified (Dijkgraaf and Vollebergh, 2004; Eshet and Shechter, 2005; Kinnaman, 2006). A historic element caused by spatial scarcity and negative externality concerns is the 'Not In My BackYard' syndrome (NIMBY). Although this syndrome has not been taken into consideration in policy decisions consciously, its occurrence actually played a constraining role in the failure to issue new permits for landfills. Together with the fact that vacant land is scarce and therefore expensive, this moratorium on new landfill zones made remaining landfill reserves a scarce good in densely populated regions in Western Europe (Levinson, 1999; Van der Zee et al., 2004) Economic theory predicts that scarce non-renewable goods will increase in price such that consumption will be restrained. Remarkably, this effect is also the aim of a landfill tax, so attention has to be paid to the effects landfill taxes have on scarce remaining landfill capacities. By gradually elaborating a dynamic optimisation model, optimal landfill and price paths could be defined by running the algorithm which includes maximising the profits of the landfill companies taking into account that the sum of all volumes that are landfilled yearly should equalise the initial remaining landfill volume. Flemish data were used to give a simulation example, and to provide knowledge with respect to the three research questions that were stated at the beginning of this chapter. Using these data, landfill paths and price paths were defined for both a scenario where landfill taxes are being used and a scenario without landfill taxes being applied. These paths were presented above in Figures 4, 5, 6 and 7. Starting with a remaining landfill capacity of 12 million tons in year zero, the scarcity of landfill capacity was taken into account. By solving the dynamic optimisation problem for different scenarios where the value of the landfill tax was strictly positive or equal to zero, different landfill paths and price paths were constructed. As could be seen, in our simulations the introduction of a landfill tax has the effect that yearly landfilled volumes decrease considerably. When a landfill tax of 60€/ton is used, it takes 42 years for full exhaustion to occur, whereas this period would be shortened to 20 years would no landfill taxes be levied. Although discounted total profit falls when landfill taxes are used, discounted total welfare increases considerably (from €132,563,772 to €239,109,115). This difference is mainly achieved by the value of the MCLT being smaller than the value of the MCPF. With regard to the optimal landfill tax level from a welfare maximisation point of view, it was shown that this tax level is equal to 55€/ton. Compared with the results of the scenario in which the landfill tax was equal to 60€/ton, this optimal tax level also results in a higher profit figure and shortens the period until landfill capacity exhaustion with five years. With respect to the total level of externalities, only the discounted values are smaller when landfill taxes are being used, as eventually the same amount of waste is landfilled which is equal to the initial remaining landfill capacity. To conclude, we can say that within our simulation structure, the added value of a landfill tax -from a broad societal point of view and knowing that remaining landfill capacity is scarce- is considerable in terms of welfare gain. In practice however, care should still be taken not to jeopardise the competitiveness of the concerned industry.

CHAPTER 4.

Waste taxes and Enhanced Waste Management: combining valuable practices with respect to future waste streams[3]

[3] This section is based on the publication: Hoogmartens, R., Eyckmans, J., Van Passel, S. (2016). Landfill taxes and Enhanced Waste Management: combining valuable practices with respect to future waste streams. Waste Manage. 55, 345-354.

Abstract

Both landfill taxes and Enhanced Waste Management (EWM) practices can mitigate the scarcity issue of landfill capacity by respectively reducing landfilled waste volumes and valorising future waste streams. However, high landfill taxes might erode the application of EWM, even though EWM creates value by valorising waste. In this regard, this chapter provides added value by developing economic modelling techniques that can be used to analyse how landfill taxation schemes can be adjusted and transformed into waste taxes in case EWM would be applied, thereby creating a combination that enables sustainable ways of processing future waste streams. As the example simulations show, the optimal tax that optimises welfare in the no EWM scenario is higher than the one in the EWM scenario (€93 against €50/ton). This difference is justified by the positive external effects that are generated by EWM practices. Within our simulation structure, and as the current Flemish landfill tax is slightly lower than these optimal levels, the choice that could be made is to further increase taxation levels or show commitment to EWM practices. Based on the Flemish example simulation results, insights are offered that can be generally applicable. A first insight points to the fact that it is not necessarily the case that the higher the tax, the more effective waste management improvements can be realised. Another insight is about keeping an eye on the extent to which the capacity scarcity issue is reflected into much higher prices and profits. By these insights, this chapter could provide relevant information about optimising waste management systems within circular economy contexts.

4.1. Introduction

As already mentioned in chapter 3, in the 1970s public attitude towards waste started to change as people became more sensitive to the negative environmental externalities caused by landfilling and the valuable space it occupies (Cainelli et al., 2015; Strasser, 1999; Van Passel et al., 2013; Walsh, 2002). Together with the fact that vacant land is scarce and therefore expensive, the constraining role of the NIMBY syndrome made remaining landfill reserves a scarce good in densely populated regions in Western Europe (Levinson, 1999; Van der Zee et al., 2004).

Flanders (Belgium) has a rich history of complicated landfill and incineration tax systems which have a double purpose. First, they want to reduce the amount of waste that is landfilled and incinerated. Secondly, they want to make environmentally friendly handling of waste and recycling of materials more attractive (Bartelings et al., 2005). The Flemish landfill tax was introduced in 1990 at a standard rate of almost €10/ton albeit with some differentiation in function of the type of waste. For combustible waste for example, the category with the highest tax rates, the nominal tax level rose from €15/ton to €50/ton between 1993 and 1997. During the following 9 years, this tax increased only moderately and in 2007 it was raised from €64/ton to €75/ton (Bartelings et al., 2005; Weissenbach, 2007). From July 2015 onwards, all environmental taxes were multiplied by a factor of 1.5. According to the permitted types of waste streams, Flemish landfills belong to one or more of three different categories of landfills. In contrast to chapter 3, this chapter focuses on category two landfills that contain inorganic non-hazardous industrial waste, household waste and industrial waste that is comparable to household waste. The reason for this is that hazardous and inert waste streams are small and less suitable for valorisation. For category two landfills, taking into account their waste composition, the increase in tax level results in an average landfill tax rate of €42/ton (including municipal surcharges). This figure was calculated based on a report of the Flemish public waste agency (OVAM, 2015b).

Being part of the bigger picture of sustainable development, Sustainable Materials Management (SMM) forms only one of several existing terminologies used for an approach to promote sustainable material production, material use and end-of-life material management. By promoting this, SMM is closely linked to the flagship initiative on resource-efficiency in the EU 2020 strategy, which aims to create a framework for policies to support the shift towards a resource-efficient and low-carbon economy. As resource-efficiency implies that natural resources, raw materials, products and also waste are used as efficiently and as environmentally responsible as possible, the link with waste management and waste valorisation is obvious. Speaking about waste and resources, another relevant concept that deserves adequate attention is the circular economy package. To facilitate the move towards a more circular economy, this package establishes a long-term vision to increase recycling, reduce landfilling and address obstacles in terms of improvement of waste management. Two concepts that can be imbedded into the framework of a transition towards a more mature SMM and a resource-efficient Europe, are Enhanced Landfill Mining (ELFM) and Enhanced Waste Management (EWM) (Jones et al., 2010; Wante, 2010). EWM consists of two pillars, of which the first one is built around the idea that future landfills become temporary storage places or future mines for those materials that cannot be directly recycled with existing technologies or show a clear potential to be recycled in a more effective way in the near future. In this approach towards future waste, eventually the entire waste management system should be optimised, with EWM practices that include not only more, but also better recycling. The second pillar is actually nothing more than the ELFM concept itself. With regard to this second ELFM pillar, it was defined as "the safe conditioning, excavation and integrated valorisation of (historic and/or future) landfilled waste streams as both materials (Waste-to-Material, WtM) and energy (Waste-to-Energy, WtE), using innovative transformation technologies and respecting the most stringent social and ecological criteria" (Danthurebandara et al., 2015a; Hermann et al., 2014; Jones et al., 2013). In Europe, the first steps towards the development of these concepts were taken when excavation and recovery of landfilled materials emerged as a promising strategy to solve the increasing shortage of landfill capacity. At the same time, benefits such as the revenues from recovered materials and reclaimed land could be obtained and the growing need for remediation of old landfills and removal of deposits hampering urban development increased interest in landfill mining as well (European Commission, 2011; Krook et al., 2012; Krook and Baas, 2013). In 2008, a trans disciplinary consortium of experts was established in Flanders in order to develop a general ELFM approach and to integrate landfilling in a radically more sustainable waste management practice called EWM. The fact that the ELFM and EWM concepts have only been under development since 2008 underlines their innovative nature and results in an academic literature review that is growing but rather limited. In 2013, a Flemish study showed that technology, regulation and markets have a clear impact on the economic potential of landfill mining and that this potential is positive for Flanders (Van Passel et al., 2013).

In the current chapter, as we focus on future incoming waste streams, the focus lies on the first pillar of EWM. Therefore, the remainder of this chapter will speak of EWM when referring to waste management. Based on foregoing descriptions, it can be seen that it may be difficult to find a balance between imposing landfill taxes and defining the taxation level on one hand and applying EWM practices on the other. After all, as higher landfill taxes have the effect of mitigating the scarcity issue of landfill capacity by reducing landfilled waste volumes, less material is made available for valorisation and the application of EWM practices also becomes less essential from a capacity point of view, thereby possibly eroding the application of EWM. Similar reasoning can also be applied the other way around. As EWM practices substantially reduce the volumes of permanently landfilled waste, remaining free capacity will be practically inexhaustible. This has the effect that landfill taxes are made redundant from a depletion postponing point of view. Only their use in terms of internalising external effects as a Pigovian tax remains in that case partially valid. In this regard, the remainder of this chapter provides added value by developing economic modelling techniques that can be used to analyse how landfill taxation schemes can be adjusted and transformed into waste taxes in case EWM would be applied, thereby creating a combination that enables sustainable ways of processing future waste streams. To this end, chapter 4 includes relevant EWM process flows and input parameters, and thereby develops a dynamic optimisation model that originates from the one presented in chapter 3. Technological data from a Flemish case study are being used to run example simulations (Danthurebandara et al., 2015a; Danthurebandara et al., 2015b; Danthurebandara et al., 2015c). These data are shown below in Figure 8 and

Appendix B, and will be discussed in more detail in the second paragraph. As this chapter identifies sustainable ways of processing future waste, we believe it can provide policy relevant insights about how to develop and bring into practice sustainable waste management practices. Furthermore, this chapter could ideally serve as a theoretical background to the concept of the circular economy, which would otherwise risk to remain a simple word without precise meanings.

The next section discusses the different elements of the dynamic optimisation model. Based on this theoretical underpinning, different scenarios are simulated in the third section. These example simulations will focus on category two landfills, as this is the most representative type of landfill where those streams belong that lend themselves best to being valorised. Finally, the article concludes with a discussion and an overview of the most important findings.

4.2. Methodology

Numerical optimisation problems are known to serve at least two purposes. First, they make theory and methods less abstract and more meaningful and secondly, they can serve as a bridge from theory and general models to actual analyses of real-world allocation problems (Conrad, 1999; Epple and Londregan, 1993). By applying dynamic optimisation techniques to the EWM concept, this chapter provides a valuable contribution to the relatively new and unexplored area of applying such techniques to the waste management field. Below, all elements of the model are discussed.

4.2.1. Demand for waste disposal

To map the aggregate demand for waste disposal to the market price, an inverse demand curve is used. When speaking about waste disposal, we allude to those future waste streams that are available to be valorised by applying EWM practices. In general, we write $p_t = D(Q_t)$, where p_t is the marginal willingness to pay or the price per ton of future waste that is being offered in period t. We will assume that price decreases in the volume Q_t (so $D'(Q_t) < 0$). In our model, a linear inverse demand function is used, which is given by:

$$p_t = D(Q_t) = a - bQ_t \quad (15)$$

with: p_t = price in year t (€/ton)

Q_t = volume of future waste available for EWM in year t (ton)

a = choke-off price, intercept on price axis (€/ton)

b = slope of the inverse demand function

An important characteristic of the linear demand curve is the implied maximum choke-off price, a. When this choke-off price is reached, the equilibrium quantity on the EWM market falls to zero. Such an upper bound may result from the existence of a substitute, available at constant marginal cost MC = a. As substitutes for EWM practices, one can think of an increase in waste export, more direct recycling routes and even an increase in waste incineration. In the model, we assume capacity exhaustion occurs in t = T. At that time, the remaining landfill capacity falls to zero. The date of exhaustion, T, is unknown and must be determined along with the competitive volume and price paths.

4.2.2. Competitive landfill companies

The optimisation model assumes that profit-maximising EWM operators are working in a perfectly competitive market. The profits of these companies are created by performing WtM and WtE activities, thereby respectively creating secondary materials like metals, glass, fines and aggregates, and RDF (Refuse Derived Fuel) that can be used in a variety of ways to produce electricity.

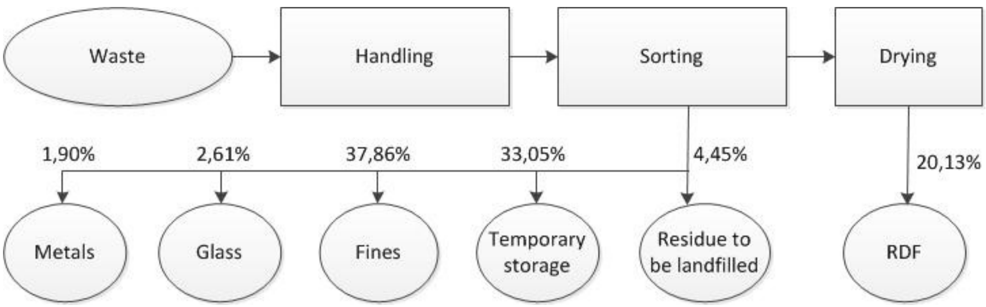

Figure 8: EWM process flow diagram showing output relative to waste input

The EWM process flow diagram in Figure 8 shows that the creation of EWM end products is preceded by handling, sorting and drying processes and accompanying processing costs. These processing costs are included in equation (16) below, followed by material and energy revenues, fines treatment costs, RDF processing costs, landfilling costs and storage costs. Storage costs are incurred for those fractions that have to be temporarily stored until proper valorisation can take place. These fractions include for example stainless steel slags and industrial sludge which are stored until proper valorisation routes are identified. In the model, the length of this average storage period was based on cautious expert estimations and set to ten years. With regard to the composition of the fines, 75% is processed as aggregates, 13% as ferrous metals and 12% as RDF (Danthurebandara et al., 2015a). Landfilling costs only apply to the waste fractions that have to be landfilled permanently. As to the imposed taxes, in case of EWM we speak of a waste tax instead of a landfill tax, as EWM operators mitigate the scarcity issue of landfill capacity by permanently landfilling only a minor part of all incoming waste streams. This fits in with the view of considering EWM facilities as ‘temporary storage places’. In terms of internalising remaining externalities, the function of these waste taxes remains valid. The waste tax, represented by parameter w, is levied at the gate of the EWM operator and thereby applies to all incoming waste streams. The percentages shown in Figure 8 were based on a characterisation study performed for the REMO site in Flanders (Danthurebandara et al., 2015), and are assumed to be indicative for the composition of future waste streams. Although this characterisation study describes a scenario in which historic, mixed waste streams would be valorised, the optimisation model developed in this chapter considers EWM practices that ideally should include not only more, but also better recycling practices within an optimised future waste management system.

Taking into account all aforementioned components and the information provided by the process flow diagram in Figure 8, when EWM operators are maximising their profits, they will try to offer processing volumes so as to:

$$\text{Maximise}_{Q_t} \pi = \sum_{t=0}^{T} \beta^t * \pi_t = \sum_{t=0}^{T} \beta^t * [p_t - w - c\_H - c\_Sort - (0.2013 * c\_D) +$$

$$(0.019 * p\_M) + (0.0261 * p\_G) - (0.3786 * c\_F) + (0.13 * 0.3786 * p\_M) + (0.12 * 0.3786 *$$

$$\mathrm{p\_E}) + (0.12 * 0.3786 * \mathrm{p\_Cert}) - (0.12 * 0.3786 * \mathrm{c\_RDF}) + (0.75 * 0.3786 * \mathrm{p\_A}) - (0.3305 * \mathrm{c\_Store}) + (0.3305 * \mathrm{r\_Store} * \beta^{10}) - (0.0445 * \mathrm{c\_}) - (0.2013 * \mathrm{c\_RDF}) + (0.2013 * \mathrm{p\_E}) + (0.2013 * \mathrm{p\_Cert})] * \mathrm{Q_t} \quad \mathrm{s.t.} \quad \sum_{t=0}^{T} \mathrm{Q_{t\_Res}} = \mathrm{S_0} \qquad (16)$$

with:

π_t = profit in year t (€)

w = waste tax (€/ton waste)

c_H = handling cost (€/ton waste)

c_Sort = sorting cost (€/ton waste)

c_D = drying cost (€/ton WtE waste)

c_F = fines treatment cost (€/ton fines)

c_RDF = RDF proc. cost (€/ton WtE waste)

c_Store = storage cost (€/ton waste)

p_M = metal price (€/ton metals)

p_G = glass price (€/ton glass)

p_E = electricity price (€/ton RDF)

p_Cert = green certificate price (€/ton RDF)

p_A = aggregates price (€/ton aggregates)

r_Store = revenue stored materials (€/ton)

S_0 = remaining landfill capacity year 0 (ton)

β = 1/(1 + δ) and δ is the discount rate

c_ = marginal landfill cost, including storage and cover costs (€/ton residue)

In the partial model, price parameter p_t is positve and represents the price that has to be paid to offer waste volumes at the gate of the EWM operator. To resolve the optimisation problem, GAMS modelling software was used in line with previous studies (Caplan, 2004; Conrad, 1999; Flakowski, 2004). By using a mixed complementary problem formulation, optimal future waste volumes to be offered at EWM facilities can be identified, taking into account the remaining landfill capacity in year 0. Moreover, evolving price paths can be modelled, together with shadow price, profit and social welfare figures. In the third section, optimisation techniques will be used to simulate three different scenarios. The first scenario is based on the current situation in Flanders, in which no EWM practices are applied and the average landfill tax amounts to €42/ton. In this scenario, as no EWM practices are applied and all waste input is landfilled, the EWM process flow shown in Figure 8 is not applicable and equation (16) only includes those variables that are directly related to landfilling, resulting in equation (17). As no EWM practices are applied in this scenario, an ordinary average landfill tax is imposed, represented by parameter l.

$$\text{Maximise}_{Q_t} \pi = \sum_{t=0}^{T} \beta^t [p_t - l - c] * Q_t \quad \text{s.t.} \quad \sum_{t=0}^{T} Q_t = S_0 \tag{17}$$

In the second and third scenario, those variables that are related to EWM are included in the equation as well, consistent with the process flow shown in Figure 8. Consequently, in these scenarios equation (16) is used in its full form. Whereas the second scenario still uses the current tax level of €42/ton, the third scenario is set up in a way that allows to investigate which waste taxation level is optimal from a societal point of view. It is important to note that all these scenarios focus on future waste streams. The first reason for this lies in the fact that this chapter wants to analyse how taxation schemes could be adjusted. As taxes for historic waste streams were already paid, it seems logical to focus on these streams for which the taxation scheme can still be adapted. Moreover, the current landfill taxation system was developed without taking into account EWM practices and their possible applications and results. The second reason has to do with the remaining landfill capacity that is available at this moment. With regard to the capacity issue, it is assumed that extra space that would be created by valorising historic waste streams is redesignated for non-disposal related uses, as is for example the case for the REMO landfill where the landfill site will be gradually developed into a sustainable nature park (Jones et al., 2013). As a result, the remaining capacity available at this moment will be used for temporarily storing future waste streams and disposing those residues that are to be permanently landfilled. By focussing on EWM practices that are applied to future waste streams, regardless of the fact whether ELFM practices are applied to historic waste streams and create extra capacity or not, this chapter only makes statements about how to optimally process future waste streams. Would one want to draw conclusions about how to optimally excavate and valorise historic waste streams and the effect this has on capacities and future waste handling, the used optimisation model would have to be further expanded.

4.2.3. Societal point of view

From a societal point of view, there are two more major aspects that are related to landfill taxation and landfilling in general. First of all, landfilling waste streams causes negative environmental externalities, like for example noise, odour, groundwater pollution and air emissions. As these externalities have an impact on society, they carry a cost, which is called an external cost. Implementing EWM practices however, involves significant external benefits. Although negative impacts like impacts of noise, light and visual disturbance on animal populations and eutrophication effects of nitrogen and sulphur deposition from the WtE plant remain present, positive external effects can outweigh the negative ones (Jones et al., 2013; Van Passel et al., 2013). First of all, EWM projects generate several positive, off-site environmental effects. CO_2 and low temperature heat for example, arising from the WtE plant, can be used in local horticulture to respectively fertilise the plants and heat the greenhouses. In this way, the use of primary fossil fuels is avoided. Furthermore, looking at the energy produced from the organic and the residual combustible waste fractions, WtE activities use locally available resources which reduces the dependence on the import of fossil fuel resources. A second positive effect is related to the production of secondary raw materials through WtM. This not only saves energy but also avoids land use for primary mining (Frändegård et al., 2013). Overall, the implementation of EWM goes hand in hand with a reduced use of fossil fuels for the production of electricity, heat and virgin materials, leading to an improved net carbon balance. Flemish research has indicated that, compared to a business as usual scenario in which no materials go in or out and energy is only recovered from methane, an EWM approach with energy and material recovery would lead to about 1 million tons less CO_2(eq) in a time span of 20 years. This corresponds to 15% less greenhouse gas emissions of which the main part is accounted for by emissions savings resulting from material recovery (de Gheldere et al., 2009; Van Passel et al., 2013).

The second aspect relates to a concept called the Marginal Cost of Public Funds (MCPF). Government revenues are typically expensive for society because of the disincentive effect for labour, tax dodging effects and administration costs (Barrios et al., 2013; Glomm et al., 2008; Schob, 1997). In contrast, landfill and waste taxes would be easier to monitor, especially in developed regions like Flanders. Indeed, there are only a few landfill sites and landfill monitoring would happen regardless of the fact whether taxes apply or not. The Marginal Cost of Landfill Taxation (MCLT) is therefore low. This gives:

$$\chi = \mathrm{MCPF} - \mathrm{MCLT} > 0 \qquad (18)$$

with: χ = relative benefit of shifting taxes

MCPF = marginal cost of public funds

MCLT = marginal cost of landfill or waste taxation

As in chapter 3, χ represents the benefit of using landfill or waste taxes instead of other, more expensive taxes like a labour tax for example. This again supports the double dividend hypothesis and fits in with the green tax shift debate (Groth and Schou, 2007; Schob, 1997). To be able to compare different scenarios from a sustainable point of view, including economic, environmental as well as a social figures, following welfare function is included:

$$\mathrm{W} = \sum_{t=0}^{T} \beta^{t} * \left[\pi_t + (w * Q_t) + \frac{(a-p_t)*Q_t}{2} + (w * \chi * Q_t) - (e * Q_t)\right] \qquad (19)$$

with: W = welfare (€)

β = 1/(1 + δ) and δ is the discount rate

π_t = profit in year t (€)

w = waste tax (€/ton waste)

Q_t = volume of future waste available for EWM in year t (ton)

a = choke-off price, intercept on price axis (€/ton)

p_t = price in year t (€/ton)

χ = relative benefit of shifting taxes

e = marginal externality cost of landfilling

This welfare function consists of the profit generated by EWM operators, tax revenues, consumer surplus, green tax reform benefits and externalities. The included tax revenues ($w * Q_t$) are actually transfer payments, as they are paid by the EWM operators and subtracted from their profit figures. Consequently, they have no net effect from a societal point of view. Although equation (19) is used in all three scenarios to generate welfare figures, it is of special use in the third scenario as it will be used there to investigate which waste taxation level maximises welfare and is therefore optimal from a societal point of view.

4.3. Example simulation results

Based on the process flow overview and the theoretical underpinnings given in the previous section, this section uses the input data shown in Appendix B to simulate three different scenarios. In the first scenario, as no EWM practices are applied, we use equation (17). In the second and third scenario however, EWM practices are applied and equation (16) is used in its full form.

4.3.1. Reference scenario without application of EWM

In this reference as-is scenario, no EWM practices are being applied and the average landfill tax to be paid amounts to €42/ton. By solving the optimisation problem and using equation (17) to define a value for parameter T, the results in Table 6 were obtained.

Table 6: Simulation without EWM

t	Landfilled volume (million ton)	p_t (€)	π_t (discounted, m€)	W_t (discounted, m€)	Shadow price (€)
0	1.04	96	40.62	84.63	39
1	1.00	100	39.09	75.82	43
2	0.96	104	37.41	67.66	47
…					
12	0.20	180	7.94	7.97	123
13	0.08	192	3.14	3.01	135
14	0	200	0	0	143
$\sum_{t=0}^{T}$	9.08		355.11	556.05	

As can be deduced from Table 6, the value obtained for variable T equals 14. This means that it takes only 14 years before the remaining landfill capacity is exhausted. The bottom row in Table 6 at the left-hand side shows the total waste volume brought to the landfill. In this scenario, where no EWM practices are being applied to valorise waste streams, all this waste is being landfilled and causes the landfill to be fully filled after already 14 years. This figure is consistent with the ones that were calculated in other research (OVAM, 2015a). As the scarcity issue in this scenario is strong, prices start at a high level causing the mark-up to be high. This creates a total discounted profit that is very high. As regards the shadow price, in the current context this measure can be regarded as an economic measure of scarcity, different from standard measures based on physical abundance (Krautkraemer, 2005). In Table 6, the shadow price represents the value of marginally loosening the constraint, that is, increasing the remaining landfill capacity. It can be seen that this shadow price rises at the rate of interest, reflecting the increasing opportunity cost as the capacity is reduced. This phenomenon is also known as the Hotelling Rule (Chermak and Patrick, 2002; Hotelling, 1931; Perloff, 2011). Below, all figures including the total discounted profit and welfare figures, are compared to scenarios with application of EWM.

4.3.2. Scenario with EWM and similar taxes

In this scenario, equation (16) is used and an average tax of €42 has to be paid for each ton of waste that is brought to an EWM facility. In the next two paragraphs, taking into account the significant external benefits generated by EWM practices, the externality cost is set to €30/ton instead of €60/ton. At the end of this section, a sensitivity analysis is included with respect to these externalilty cost values. After solving the optimisation problem for this scenario, the results in Table 7 were obtained.

Table 7: Simulation with EWM and similar taxes

t	Waste input (million ton)	p_t (€)	π_t (discounted, m€)	W_t (discounted, m€)	Shadow price (€)
0	1.60	40	0.0023	161.32	0.0014
1	1.60	40	0.0023	146.65	0.0016
2	1.60	40	0.0023	133.32	0.0017
…					
120	0.16	184	0	0	145
121	0.01	199	0	0	159
123	0	200	0	0	160
$\sum_{t=0}^{T}$	179.78		0.25	1,774.42	

Looking at Table 7, we see that the value of T has increased substantially from T=14 in the case of no EWM, to T=123. Of the total waste input that is shown in the bottom row of Table 7, only 4.45% residue is landfilled. The vast majority of all waste that is offered at the EWM facility gets valorised as material or energy. As a result, it will take much longer before the remaining capacity is fully filled and the impact of the scarcity issue is lower. This affects the prices, which start at a lower level than before and reduces considerably the mark-up that is earned. Accordingly, the total discounted profit is much smaller than before. One should be aware however, of the discounting process that exerts more influence in this case, as it takes much longer until the remaining capacity is fully exhausted. With regard to the shadow price, it can be seen that the value of the scarcity indicator is initially much lower than before. This is quite logical, taking into account that the remaining capacity is depleted at a much lower rate than when no EWM practices were being applied, making it less scarce. Although the total discounted profit is smaller in Table 7 than in Table 6, when we look at discounted total welfare we see that from a societal point of view and within our simulation structure, application of EWM is preferable. In Table 8, the different components of the total welfare figures are presented. When comparing the two scenarios with each other, one has to be aware of the differences that exist between both scenarios, with the timing difference as the most important one. Being cognizant of this, Table 8 indicates that within our simulation structure, tax revenues are higher when EWM is applied because the total waste volume offered for valorisation is larger. With regard to the consumer surplus and green tax reform figures, they are both also higher when EWM is applied. The higher consumer surplus is not only caused by the higher waste input, but also by the lower prices that have to be paid. Although total externalities are higher in case of EWM, one has to be aware of the fact that almost as many as twenty times more waste is processed during the years.

Table 8: Total discounted welfare components

	No EWM	EWM (w=42)
Component	Value	Value
Profit (m€)	355.11	0.25
Tax revenue (m€)	256.82	740.85
Consumer surplus (m€)	259.64	1,414.32
Green tax reform (m€)	51.36	148.17
Externalities (m€)	-366.88	-529.18
Total (m€)	556.05	1,774.42

Given the results in Table 6 and 7, dynamic overviews can be plotted. These are shown in Figure 9 with the right graph representing the scenario with and the left graph representing the scenario without EWM. These graphs clearly show the difference that EWM can make in terms of postponing the moment of full capacity exhaustion and in terms of price evolution. The coloured areas in both graphs depict the total future waste volume that is brought to a landfill or an EWM facility over T years. Again, it is important to note that the coloured area at the right hand site depicts the entire waste input of which the vast majority is valorised (the dark-coloured area) and only a small residual fraction (the light-coloured area) is landfilled. At the left hand side however, when no EWM is applied, the waste volume that is represented by the dark coloured area is fully disposed and equal to the initial remaining landfill capacity.

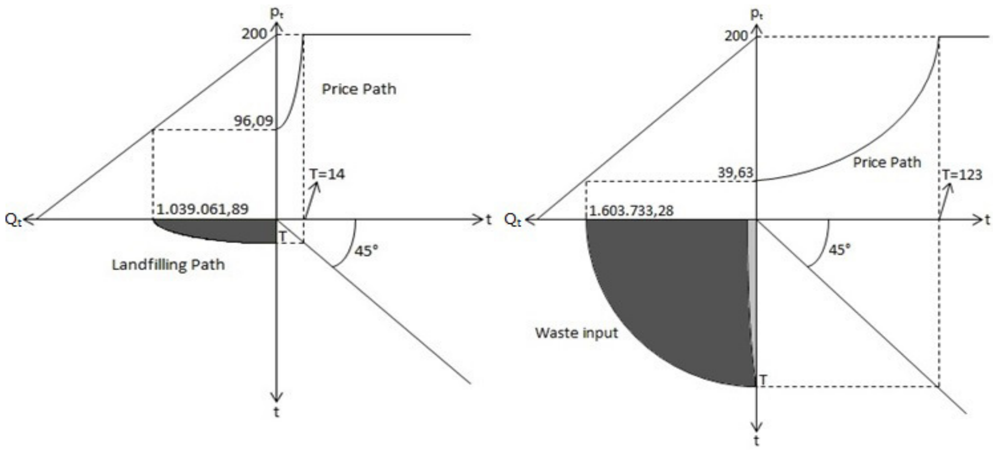

Figure 9: dynamic overview with (right) and without (left) applying EWM

4.3.3. Scenario with EWM and optimal taxes from a societal point of view

In this scenario, the dynamic optimisation model was set up in a way that allows to investigate which waste taxation level is optimal from a societal point of view, when EWM is applied. In this regard, the objective is to maximise the value of equation (19). As stated before, the implementation of EWM leads to an improved net carbon balance, thereby generating positive external effects. Based on the external benefits, it would be reasonable to assume that support mechanisms like for example lower taxation levels are justified. After solving the optimisation model, a waste tax of €50/ton was found to be optimal from a societal point of view. Higher tax levels would not only lower total profit, but also total welfare figures. After all, in that case the increase in tax revenue does no longer outweigh the decreases in profit and consumer surplus figures. Although this optimal waste tax level is higher than the average landfill tax level of the as-is scenario, actually the optimal waste tax level should be compared with the average landfill tax level that would optimise welfare in the reference scenario without EWM, being €93/ton. In comparison with this average landfill tax level, the lower optimal waste tax level of €50/ton refers to the positive external effects that are generated by EWM practices. In Table 9, the results of the EWM scenario with optimal waste taxes are shown.

Table 9: Simulation with EWM and optimal taxes

t	Waste input (million ton)	p_t (€)	π_t (discounted, m€)	W_t (discounted, m€)	Shadow price (€)
0	1.52	48	0.0012	161.80	0.00077
1	1.52	48	0.0012	147.09	0.00084
2	1.52	48	0.0012	133.72	0.00093
…					
126	0.26	174	0	270.24	126
127	0.14	186	0	132.23	139
128	0	200	0	0	152
$\sum_{t=0}^{T}$	179.78		0.14	1,779.79	

Comparing Table 9 with Table 7, we see that the value of T has slightly increased from T=123 in the case of the tax being equal to €42/ton, to T=128 with an optimal waste tax of €50/ton. Regarding the waste input, slightly less waste is processed each year when the average tax level is higher. As to the yearly waste volumes, their absolute values however are still slightly higher than the volumes that are collected in practice at this moment. This is not necessarily a problem. As EWM valorises waste, additional input volumes might possibly be withdrawn from waste streams that otherwise would be directed to less high-grade applications like incineration for example. Still, the vast majority of all input waste is valorised as material or energy and only 4.45% residue is landfilled. Prices start at a higher level than before, and the higher waste tax slightly increases the costs thereby lowering the (total) discounted profit. As can be seen in Table 7 and Table 9, total discounted profit levels are significantly lower than in the scenario without EWM. This is even the case with landfill tax levels in the no EWM scenario being equal to the optimal level of €93/ton. Taking into account the influence of the discounting process and the question whether the prices predicted by the Hotelling Rule for the reference scenario will rise that high in practice, it could still turn out that measures in support of EWM practices are needed. As a fiscal instrument, one can think of instruments like tax credits for example. When such tax credits would be assigned to EWM operators, profit levels could be boosted thereby supporting EWM operations. With regard to the shadow prices, we see that their values in Table 9 are lower than the values in Table 7. This is a logical consequence of the fact that in case of a higher waste tax level, the remaining landfill capacity is exhausted at a slightly lower rate, making the remaining capacity slightly less scarce and valuable. As the objective of this scenario was to identify the optimal waste tax from a societal point of view, welfare is maximised and higher than in both previous scenarios. In Table 10, the different components of the total welfare figures are presented. By means of comparison also the results from the previous scenario (w=42) are included. As can be seen, the higher waste tax level increases the total tax revenue and green tax reform figures. With regard to the consumer surplus, higher prices decrease the surplus when the tax level is equal to €50/ton. Although the externality figures are different from each other, in fact each scenario processes an equal amount of waste input. The shown difference is purely caused by the discounting process.

Table 10: Total discounted welfare components

	EWM (w=42)	EWM (w=50)
Component	Value	Value
Profit (m€)	0.25	0.14
Tax revenue (m€)	740.85	838.01
Consumer surplus (m€)	1,414.32	1,276.84
Green tax reform (m€)	148.17	167.60
Externalities (m€)	-529.18	-502.80
Total (m€)	1,774.42	1,779.79

Given the results in Table 9, the dynamic overview that could be plotted using the waste taxation level that is optimal from a societal perspective, is broadly similar to the one shown at the right hand side in Figure 9. The slight increase in the taxation level from €42 to €50/ton causes prices to start at a higher level than before, thereby postponing the moment of full capacity exhaustion with five more years. The total amount of waste input offered at the EWM facilities over T years is identical in both EWM scenarios, as is the total amount of residual fractions that are landfilled.

As a sensitivity check, the values of some uncertain input parameters were altered in order to examine the impacts these changes can have on the model outcomes. Whereas in the reference scenario an externality cost of €60/ton was used, this value was reduced to €30/ton when EWM came into play to take into account the significant external benefits created by these practices. Would these figures be reduced to respectively €50 and €25/ton, the optimal taxes in the no EWM and EWM scenario would be equal to respectively €90 and €47/ton. When the externalities in the EWM scenario would be further reduced to €20/ton, the optimal waste tax also decreases to €43/ton. This is all in line with the reasoning that lower externalities justify lower taxes. Also, when externalities are lower, total welfare should be higher. This can be shown in the calculations. In case the externalities would be equal to €20/ton, with an optimal waste tax of €43/ton, the total welfare would be equal to 1,950.93 million euros. Before, in case the externality cost was equal to €30/ton, this was only 1,779.79 million euros. The fact that profit levels increase when externality costs decrease is quite logical as in that case optimal tax levels are decreasing as well. The value of variable T slightly decreases when externality costs are lowered. Other parameter values that were altered within the context of this sensitivity check are the net income levels from the EWM practices. Would these net income levels be reduced with 25 or 50%, the optimal waste tax level that maximises welfare in the EWM scenario would still remain at the level of €50/ton. Considering the decreased net EWM income levels and the tax expenses that stay at the same level, prices increase and yearly processed waste volumes decrease, causing the profit levels to slightly decrease. Would these income levels be reduced with 25 or 50%, optimal welfare figures would decrease to respectively 1,767.90 and 1,756.05 million euros and variable T would slightly increase to 129 years.

4.4. Conclusions

As the current landfill taxation system was developed without taking into account EWM practices and their prossible applications and results, it may be difficult to find a balance between imposing landfill taxes and defining the taxation level on one hand and applying EWM practices on the other. After all, as landfill taxes have the effect of mitigating the scarcity issue of landfill capacity by reducing landfilled waste volumes, less material is made available for valorisation and the application of EWM practices also becomes less essential from a capacity point of view, thereby possibly eroding the application of EWM. Similar reasoning can be applied the other way around as EWM practices substantially reduce the volumes of permanently landfilled waste and thereby postpone the moment of full capacity exhaustion considerably. In turn, this has the effect that the use of landfill taxes only remains partially valid in terms of internalising external effects. In this regard, this chapter provided added value by developing economic optimisation modelling techniques that can be used to analyse how landfill taxation schemes can be adjusted and transformed into waste taxes in case EWM would be applied, thereby creating a combination that enables sustainable ways of processing future waste streams. To this end, the chapter included relevant EWM process flows and input parameters, and thereby developed a dynamic optimisation model that originates from the one presented in chapter 3. Technological data from a Flemish case study were used to run some example simulations. As shown, within our simulation structure applying EWM practices has the effect of significantly postponing the moment of full capacity exhaustion. In case the taxation level would stay at the same level of €42/ton, applying EWM would postpone the moment of capacity exhaustion with 109 years compared to the as-is scenario (from T=14 to T=123). As stated in this chapter, the implementation of EWM leads to an improved net carbon balance, thereby generating positive external effects. These positive effects would justify implementing support mechanisms like lower taxation levels. In the last scenario, the dynamic optimisation model was set up in a way that allows to investigate which waste tax is optimal from a societal point of view, when EWM is applied. Within our simulation structure, this is the case for a waste tax of €50/ton. Higher levels would not only lower total profit, but also total welfare figures as increases in tax revenue would no longer outweigh decreases in profit and consumer surplus figures. Although this optimal waste tax level is higher than the average landfilltax level of the as-is scenario, the optimal waste tax should actually be compared with that landfill tax level that would optimise welfare in case no EWM practices would be applied, being €93/ton. In comparison with this landfill tax level, the lower optimal waste tax level of €50/ton refers to the positive external effects that are generated by EWM practices. In conclusion, within our simulation structure we see that present policy approaches the optimum as the current landfill tax level is not that much different from the optimal waste tax of €50/ton. It is however important to note that this optimal tax level would only maximise welfare in case EWM practices are applied. Would no such practices be applied, a higher landfill taxation level of €93/ton would be justifiable. Consequently, within our simulation structure, and as the current Flemish landfill tax is slighty lower than the optimal levels, the choice that could be made is to further increase taxation levels or show commitment to EWM practices.

Regarding the profit levels of both EWM scenarios, they are both significantly lower than the profit level of the as-is scenario where no EWM practices are applied. This is even the case would landfill tax levels in the no EWM scenario be equal to the optimal level of €93/ton. Regarding the high price levels in the as-is scenario however, it can be questioned whether these prices predicted by the Hotelling Rule will increase that high in practice. After all, it is possible that landfill operators may lower their prices in order to attain the fixed landfill volume that was assigned to them in their landfill operating permit. In addition, it can happen that pressure on politicians becomes higher when an exhaustion of remaining landfill capacity is near. This might have the effect that additional capacity is created, thereby making the scarcity issue less pronounced and tempering the increase in prices. If it would still turn out that measures in support of EWM practices are needed, fiscal instruments such as tax credits could be used. By transferring part of the tax revenues back to EWM operators, their profit levels could be boosted thereby supporting EWM operations. In addition to this redistributive function, another advantage of such an instrument can be a controlling function as credits could only be assigned after certain EWM practices are actually carried out.

Based on the results of the Flemish example simulations, there are some lessons that can be generally applicable. By providing these lessons, this chapter could provide relevant information about optimising waste management systems within circular economy contexts. A first lesson is about the taxation level. Often it is believed that the higher the tax, the more effective waste management improvements can be realised (Krook et al., 2012). This chapter has shown however, that this is not necessarily the case within the used simulation structure. After all, when a particular taxation level is reached, higher levels would not only lower total profit but also total welfare as increases in tax revenue do not longer outweigh decreases in profit and consumer surplus figures. Another lesson is the fact that policy should take into account and keep an eye on the extent to which the capacity scarcity issue is reflected into much higher prices and profits. When necessary, policy could take appropriate measures in order to support EWM operations.

Although theoretical models are simplified road maps of reality, they can serve as an important bridge from theory and general models to actual analyses of real-world allocation problems. Evolving prices, costs, capacities and technologies can be taken into account at any time allowing that policy can constantly be adapted to prevailing market conditions. This also allows that future decisions can be made in a sustainable and socially responsible way. We do recognise that further research is needed to refine and further develop the used model. Aspects that deserve further attention are for example the assumption made in this chapter about the extra space that would be created by valorising historic waste streams and structures like tax credit systems that can assure that operators will keep their promises related to the processing of future waste streams.

CHAPTER 5.

A Hotelling model for the circular economy: Including recycling, substitution and waste accumulation[4]

[4] This section is based on Hoogmartens, R., Eyckmans, J., Van Passel, S. (2016). A Hotelling model for the circular economy: Including recycling, substitution and waste accumulation, Submitted.

Abstract

Non-renewable resources include a large variety of mineral deposits that have been formed by geological processes over millions of years. Although extraction of such resources still provides benefits as employment and economic revenues, it also contributes to negative externalities and increased resource scarcity. A central question in policy debates is how to optimally extract non-renewable resource stocks over time while taking possible substitutes and recycling into account. The present chapter responds to this challenge by developing a generic numerical optimisation model that can be used to simulate non-renewable resource practices and effects of different policy instruments within the material flow of a particular resource. By including recycling and substitution, the developed model extends the seminal cake-eating Hotelling-type model that dominates the resource economics literature. In addition to being generically designed, the added value of the model emerges especially when taking into account non-competitive market settings, interacting policy instruments and environmental externalities. The simulation examples shown in this chapter are some of a wide variety of possible simulations and the results shown are in line with expectations and intuition based on theoretical models. This reinforces the practical usefulness of the model in helping policy makers design sustainable resource practices.

5.1. Introduction

Non-renewable resources include a large variety of mineral deposits from which metals, fossil fuels and other processed minerals can be obtained. Although the extraction of these resources provides local employment and revenues, it is usually accompanied by negative environmental externalities. For example, quarrying sand and gravel can be noisy and dusty and traffic to the mining pit can create disamenities for neighbours. Furthermore, the natural environment can be damaged by run-off water, waste generation and visual pollution (Eckermann et al., 2012). Along with these negative aspects is often a problem of scarcity. As the crude forms of these non-renewable resources are created by long-term geological processes, their rate of formation is sufficiently slow – in timescales relevant to humans – that they are labelled as non-renewable (Perman et al., 2011). This, together with the intensive use of resources that enabled European wealth and wellbeing to grow, and strict demarcations of mining areas, causes remaining reserves to be limited and scarce (European Commission, 2011a). The European Union has recognised that the current utilisation of non-renewable resources in particular is non-sustainable and has identified resource efficiency as one of seven flagship projects to pursue in its Europe 2020 strategy (European Commission, 2011b). This flagship initiative, which had the aim of creating frameworks for policies to support the shift towards a resource-efficient and low-carbon economy, looks for ways to answer the following key question: What is the optimal extraction path over time for any particular non-renewable resource stock?

In practice, there is no straightforward answer to this question because non-renewable resources are quite heterogeneous and it is often unclear what policies should be undertaken in order to facilitate the transition towards a resource-efficient economy. Economic theory would suggest that companies have an incentive to exploit efficiency potentials to gain competitive advantages. The prevailing view is that increasing scarcity of non-renewable resources will be accompanied by a steady price increase that signals scarcity to consumers and triggers eco-innovations for substituting or limiting the use of scarce materials. However, efficiency potentials in the utilisation of natural resources have been underexploited and the price mechanism is often fundamentally flawed. Private resource owners are often more impatient than society as a whole, which leads to excessively fast exploitation. In addition, market prices do not reflect sufficiently to external costs in the absence of proper government regulation (Eyckmans and Dubois, 2014; Söderholm and Tilton, 2012). Based on these observations, implementing policy instruments to support a more sustainable resource use is justified. Moreover, this would be in accordance with the calls for 'true pricing' by internalising external costs and the green tax shift debate. At present, many European Member States have not tackled the shift from labour towards environmental taxation at all, even though environmental taxes can be a step towards reflecting the full external and social costs of resource extraction, utilisation and end-of-life practices (Bringezu, 2002; Wilts et al., 2014). Along with steering behaviour, these taxes would help to reorientate public finances away from labour taxation, which would benefit job creation and economic growth.

The discussion so far highlights the difficulty of identifying policies that trigger the transition towards a resource-efficient, circular economy. The challenge is exacerbated by the lack of appropriate methodologies that combine elements such as waste accumulation, recycling and substitution in a unified framework. In this regard, this chapter adds value by developing a generic optimisation model that can be used to simulate non-renewable resource practices and the effects that different policy instruments can have within the material flow of a particular substance. The generic optimisation model provides a tool for designing policies that support the transition towards a more resource-efficient economy, which can boost economic performance while reducing resource use and negative environmental externalities. Shifting the economy towards a more resource-efficient path should bring competitiveness and new sources of growth and jobs through cost savings from improved efficiency and better management of resources over their whole life cycle.

The next section provides more background on the type of modelling applied in this chapter and sets up the model with all of its components. In the third section, numerical simulations are carried out based on these theoretical underpinnings, showing typical outcomes and results that can be obtained. The article concludes with a discussion and an overview of the most important findings.

5.2. Hotelling modelling with recycling

Next to serving as a bridge between theoretical models and actual analyses of real-world policy designs, numerical optimisation problems can quantify the net effects of counteracting forces that theoretical models are usually unable to sign unambiguously (Conrad, 1999; Epple and Londregan, 1993; Flakowski, 2004). Although such optimisation problems are actually simplified roadmaps of reality, they can provide generally applicable and policy-relevant insights into how to develop resource efficiency and put it into practice. As in the previous chapters, the base of the model developed in this chapter lies with the well-known Hotelling model (Hotelling, 1931). According to the Hotelling rule, the shadow price of a non-renewable resource increases at the rate of discount along the socially optimal extraction path. This rising shadow price reflects the increasing opportunity cost as remaining non-renewable resource reserves are consumed. Private profit maximising resource owners interacting on a competitive commodity market will choose an extraction path that coincides with the socially optimal one if the private and social discount rates are equal (Chermak and Patrick, 2002; Perloff, 2011).

Several theoretical models on resource extraction and recycling were developed in the 1970s. In a study carried out by Smith (1972) for example, a rudimentary model was used that emphasises only those elements essential to the recycling problem. Later, Lusky (1975) developed an integrated model of conservation and recycling in a framework of a natural resource cycle, and Hoel (1978) studied the optimal path of extraction and recycling under various assumptions about the environmental effects of recycling and the assimilative capacity of the environment. In addition to these theoretical models, also numerical simulation models in the same spirit were published. In the study carried out by Weikard and Seyhan (2009) for example, a resource extraction model was built for a competitive fertilizer market and included different recycling options. Seyhan et al. (2012) also focused on the extraction and recycling of Phosphorus, and developed a resource-specific model. Compared to these studies, our model adds more value by developing a comprehensive generic optimisation model that can be used to simulate non-renewable resource practices and effects of different policy instruments within the material flow of a particular resource. In addition to being generically designed, our model includes recycling, substitution and waste accumulation in a unified framework, and is able to simulate non-competitive market settings, first-best welfare maximisation scenarios, interacting policy instruments and environmental externalities linked to different stages of the material flow.

5.2.1. Economic actors in decentralized market model

The model involves four different types of economic actors: (i) consumers, (ii) mine owners, (iii) suppliers of substitute material and (iv) recyclers.

5.2.1.1. <u>Consumers</u>

The representative consumers choose to consume an amount of non-renewable resources, Q_t, to maximise their utility and take their budget constraints into account. In the model, utility is an increasing and strictly concave function of consumption, so that $U' \geq 0$ and $U'' < 0$. Furthermore, there is a numéraire good, v_t, the price of which is normalised to unity. Making use of this numéraire good facilitates value comparisons as all relative prices in the model can be expressed in terms of this numéraire as a tradable economic commodity. It is further assumed that the income of the consumers is exogenous and that no intertemporal savings or borrowings take place. In the model, the exogenous income is denoted by $\bar{y}_t$ and is strictly larger than zero. The price of the consumption good is denoted by p_t, and can be supplemented with a consumption excise tax t^q. Combining all these elements provides the following constrained optimisation problem:

$$\max_{v_{t,}Q_t} v_t + U(Q_t) \quad \text{s.t.} \quad v_t + [p_t + t^q]Q_t \leq \bar{y}_t \tag{20}$$

Forming the Lagrangian of this consumer problem gives us:

$$L(v_t, Q_t, \lambda_t) = v_t + U(Q_t) + \lambda_t[\,\bar{y}_t - v_t - [p_t + t^q]Q_t] \tag{21}$$

In equation (21), parameter λ_t represents the Lagrange multiplier of the consumer's budget constraint or marginal utility of extra income. Taking the derivative of this equation with respect to the numéraire good v_t gives us the first-order condition $\lambda_t = 1$, whereas differentiating with respect to the consumption level Q_t gives us:

$$U'(Q_t) - p_t - t^q \leq 0 \quad \perp \quad Q_t \geq 0 \tag{22}$$

Equation (22) indicates that consumers buy consumption goods up to the point at which their marginal utility of consumption equals the full price of the good. This full price consists of the purchasing price p_t, supplemented with the consumption excise tax t^q. Concerning the range within which these prices can fluctuate, the model imposes an upper limit in terms of a choke-off price. When this choke-off price is reached, the equilibrium quantity on the market falls to zero, meaning that the demand is choked off at this price. When this upper bound is reached, people who would otherwise use this resource switch demand to a substitute non-renewable resource or to an alternative final consumption good that does not use this resource as an input. With regard to the numerical implementation of the demand for the consumption good, the model uses a standard linear inverse demand curve in order to differentiate closed form solutions. In the model, we allow for the possibility that the intercept of this demand curve changes over time; for instance, in order to reflect changes in real income, preferences or population over time. This gives following inverse linear demand function:

$$U'(Q_t) = a_t - bQ_t \tag{23}$$

The utility function necessary to calculate welfare and corresponding with this inverse demand function is given by the integral under the marginal utility function:

$$U(Q_t) = \int_0^{Q_t} U'(x)dx = \int_0^{Q_t} [a_t - bx]dx = aQ_t - \frac{b}{2}Q_t^2 + \text{constant} \tag{24}$$

This utility is derived directly from the consumption of the non-renewable resource. As to some comparative statics, further differentiating equation (22) shows that the consumption level Q_t decreases when price p_t or excise tax t^q increases, which is an intuitive result:

$$\mathrm{U}''\mathrm{dQ} = \mathrm{dp} + \mathrm{dt}^{\mathrm{q}} \quad \Longrightarrow \quad \frac{\mathrm{dQ}}{\mathrm{dt}^{\mathrm{q}}} = \frac{\mathrm{dQ}}{\mathrm{dp}} = \frac{1}{\mathrm{U}''} \leq 0 \qquad (25)$$

5.2.1.2. ***Mining companies***

A second economic actor concerns the representative mining company that extracts the non-renewable resource as virgin material and sells it directly to the consumers. The virgin extraction rate is represented by parameter $\mathbf{q^v}$. The total initial stock of this virgin material is given by $\mathbf{S_0}$ and is strictly larger than zero. As this total stock is fixed, the developed model can be classified as a kind of cake-eating model of non-renewable resource depletion (Weikard and Seyhan, 2009). In each period, mining companies decrease the remaining stock by mining virgin resources. Throughout the model, this remaining stock should always be nonnegative; using a linear demand function, we assume that virgin resource extraction stops in a finite time at period t = T. The marginal cost of virgin material production is assumed to be based on a linear cost function. This means that the marginal cost is constant at every point in time, although it can evolve over time as a result of factors such as technological progress. In the model, this marginal production cost is represented by parameter $\mathbf{c_t^v}$. Next to this cost parameter, we foresee the possibility of introducing a revenue-based extraction tax $\mathbf{t^v}$. Being revenue-based, this tax is equivalent to an increase in the extraction costs. The related environmental motives for taxing resource extraction identified in the literature are: (i) to decrease the rate of extraction, (ii) to focus on all generated environmental externalities and (iii) to encourage the substitution of secondary and recycled materials for virgin material (Söderholm, 2011).

The mining sector itself is modelled as a standard Hotelling non-renewable resource problem, with every mining company maximising its flow of discounted profits. With $\beta_t = \frac{1}{[1+\delta]^t}$ denoting the private discount factor and δ the private discount rate, mine owners decide when to extract and sell the mined, non-renewable resources in order to maximise the present value of the resource. This gives the following maximisation problem:

$$\max_{q_t^v\ (t=1,2,\ldots,T)} \pi^v = \sum_{t=1}^{T} \beta_t [p_t - c_t^v - t^v] q_t^v \qquad (26)$$

s.t. $\begin{cases} S_{t+1} - S_t = -q_t^v \quad \forall t = \{1,2,\dots,T\}, S_0 > 0 \\ \sum_{t=0}^{T} q_t^v = S_0 \\ q_t^v \geq 0 \quad \forall t = \{1,2,\dots,T\} \end{cases}$

The first restriction included in equation (26) indicates that the remaining resource stock at the beginning of period t+1 is equal to the remaining stock at the beginning of period t, decreased by the virgin extraction that takes place in period t. The third restriction ensures that the supply of virgin material is nonnegative. Writing the Lagrangian for this dynamic program gives us:

$$L = \pi^v = \sum_{t=1}^{T} \beta_t [p_t - c_t^v - t^v] q_t^v - \sum_{t=1}^{T} \beta_{t+1} \lambda_{t+1} [S_{t+1} - S_t + q_t^v] \tag{27}$$

In equation (27), the remaining stock constraint was multiplied by the discount factor β_{t+1} in order to simplify the calculations. Taking the derivatives of this equation with respect to the virgin extraction rate (control variable) and the remaining resource stock (state variable), respectively, provides the two following expressions:

$$\frac{\partial L}{\partial q_t^v} = \beta_t [p_t - c_t^v - t^v] - \beta_{t+1} \lambda_{t+1} \leq 0 \quad \perp \quad q_t^v \geq 0 \tag{28}$$

$$\frac{\partial L}{\partial S_t} = \beta_{t+1} \lambda_{t+1} - \beta_t \lambda_t \leq 0 \quad \perp \quad S_t \geq 0 \tag{29}$$

After some calculations, the first-order condition with respect to the state variable S_t can be rewritten as:

$$\lambda_{t+1} - \lambda_t \leq \delta \lambda_t \quad \perp \quad S_t \geq 0 \tag{30}$$

Similarly, the first-order condition with respect to the control variable q_t^v can be rearranged to yield:

$$p_t - c_t^v - t^v \leq \lambda_t \quad \perp \quad q_t^v \geq 0 \tag{31}$$

In these equations, parameter λ_t represents the net price of the resource, also called the shadow price. Combining the latter two equations, the well-known Hotelling rule for the optimal extraction of a non-renewable resource can be stated as:

$$\frac{\lambda_{t+1} - \lambda_t}{\lambda_t} = \delta \quad \Leftrightarrow \quad \frac{[p_{t+1} - c_{t+1}^v - t^v] - [p_t - c_t^v - t^v]}{[p_t - c_t^v - t^v]} = \delta \tag{32}$$

Equation (32) shows that, along an optimal extraction path, the net price or shadow price of the non-renewable resource increases at the rate of discount ρ. In other words, the discounted net price of this non-renewable resource is constant along the most efficient resource extraction path. By formulating the Hotelling rule in this way, it can be seen that the Hotelling rule is actually only a special case of a general asset-efficiency condition. After all, this condition states that the present value of any efficiently managed asset will remain constant over time.

5.2.1.3. ***Substitute suppliers***

A third economic actor is the representative substitute supplier. This supplier allows for the possibility that substitute material, such as imported material from abroad, can come to the market. Substitution takes place when the price of the non-renewable virgin resource rises to such an extent that it makes alternatives economically more attractive. Would a substitute come onto the market, its full price would function as a choke-off price, at which a switch is made from virgin to substitute material. The perfectly elastic supply of this substitute is represented by variable q^s. We assume that this substitute material can be imported at a fixed cost c^s. Next to this cost parameter, we foresee the possibility that authorities levy an import duty t^s on the material. The supply schedule of the substitute material is given by the following first-order condition:

$$p_t - c_t^s - t^s \leq 0 \quad \perp \quad q_t^s \geq 0 \qquad (33)$$

If the substitute material comes onto the market, it holds that q_t^s is strictly larger than zero, and $p_t = c_t^s + t^s$. Otherwise, if the price was lower than the sum of import costs and duties, the substitute material would not come onto the market and q_t^s is equal to zero.

5.2.1.4. ***Recyclers***

Apart from virgin and substitute material, we also allow for the possibility that a representative recycler processes end-of-life waste with the intention of supplying recycled material. We assume that there is no market for waste and that the recycler can therefore obtain this waste for free. Furthermore, we assume that there is no free disposal of waste in terms of illegal dumping or street litter and that there is no piling up of waste with consumers. All produced waste volumes are given to the recycling sector, which in turn perfectly complies by accepting the whole stream. In processing this waste, represented by variable w_t, the recycler chooses a recycling effort γ_t as to maximise profits. As γ_t represents the percentage of waste that is recycled, its value lies in the range [0,1]. The profit of the recyclers consists of revenue from selling recycled material at price p_t. In the model, recycling has an increasing and convex cost function, so that $r' \geq 0$ and $r'' > 0$, with r representing the recycling unit cost that is function of parameter γ_t. The non-recyclable fraction is disposed of at a full price p^d per unit. This parameter includes the gate fee that is charged at the landfill and a possible landfill or disposal tax. Such a disposal tax can provide a strong economic incentive whose relative strength is such that one would expect the recycling to be a consequence of the benefits of having to avoid disposal as opposed to the avoidance of the virgin extraction tax. Together with this extraction tax, the tax on waste disposal should provide a strong incentive to employ recycled materials rather than to extract virgin materials and dispose of old ones (Ecotec, 2001; Söderholm, 2011). Summarising the above provides the following profit maximisation problem:

$$\max_{\gamma_t} \pi_t^r = p_t \gamma_t w_t - r(\gamma_t) w_t - [1 - \gamma_t] w_t p^d \qquad (34)$$

Taking the derivative of this equation with respect to recycling effort γ_t gives us the following first-order condition:

$$p_t - r'(\gamma_t) + p^d \leq 0 \quad \perp \quad \gamma_t \geq 0 \qquad (35)$$

In a competitive recycling market, the marginal unit cost of recycling, $r'(\gamma_t)$ should be equal to the price of the virgin resource. The recycling cost function was calibrated in such a manner that it would result in logical recycling efforts, with parameter γ_t increasing with increases in the full price of the virgin resource. In the model, it is assumed that $r'(0) = 0$ and that the limit of $r'(\gamma_t)$ tends to plus infinity when γ_t approaches one. This ensures the existence of an interior solution, as $p_t + p^d$ is always larger than zero. In the context of generating some comparative statics, it would be interesting to know how recycling depends on variables like the price of material p_t and the full disposal cost p^d. Totally differentiating equation (35) delivers:

$$r'' d\gamma_t = dp_t + dp^d \quad \Rightarrow \quad \frac{d\gamma_t}{dp_t} = \frac{d\gamma_t}{dp^d} = \frac{1}{r''} > 0 \qquad (36)$$

This equation reveals intuitive comparative statics results: the higher the price of material and the higher the full cost of disposal of recycling residues, the higher the recycling effort chosen by a profit-maximising recycling firm. These increasing recycling efforts reduce the pressure on demand for virgin materials, help to reuse valuable materials that would otherwise be wasted, and reduce energy consumption and greenhouse gas emissions from extraction and processing (European Commission, 2011; Pittel et al., 2010).

The existence of an interior solution does not guarantee positive profits for the recycler. Therefore, to ensure that a potential solution to the model would not entail a loss scenario for the recycler, the following zero-profit condition is included:

$$p_t\gamma_t - r(\gamma_t) - [1-\gamma_t]p^d \leq 0 \quad \perp \quad w_t \geq 0 \qquad (37)$$

Clearly, if recycled material has no value ($p_t = 0$) or if landfill costs are high and material prices are low, there is no viable recycling market. Recyclers can never cover their costs and would leave the market. In this case, waste could be piled up with consumers for example. In this chapter however, next to including the zero-profit condition, the model is calibrated in a manner that recyclers do not incur losses and thus prevent a piling up of waste by processing it.

Finally, the solution for the recycler maximisation problem determines the amount of recycled material that is supplied to the market as:

$$q_t^r = \gamma_t w_t \qquad (38)$$

5.2.2. Market equilibrium and market balance

With all of the aforementioned equations in mind, we can formulate the market equilibrium for both the materials and the goods market. For the goods market, consumer demand Q_t should equal supply q_t in every period:

$$Q_t = q_t \quad \forall t = 1,2,\dots,T \qquad (39)$$

For the materials market, total material demand should equal total supply, which consists of the virgin, substitute and recycled materials that are assumed to be perfect substitutes:

$$q_t = q_t^v + q_t^s + q_t^r \quad \forall t = 1,2,\dots,T \qquad (40)$$

Finally, we must specify the flow of material throughout the life cycle of the consumption good. It is assumed that material quality does not deteriorate with recycling, so recycled material can be used in the production of new consumption goods, which in turn can be recycled again without incurring quality losses. With regard to the durability of the functional relationship between past consumption and waste generation, the model can be set up in different ways. The first way is to assume that goods are not durable at all and give rise to waste immediately after consumption, with $w_t = q_t$. Alternatively, we can assume that consumption goods only last for one period; this would imply that $w_t = q_{t-1}$. Another more general approach is to assume that, for example, one half of the goods live two periods, one quarter live for three periods and another quarter live for one period, such that:

$$w_t = \frac{1}{4}q_{t-1} + \frac{1}{2}q_{t-2} + \frac{1}{4}q_{t-3} \qquad (41)$$

Obviously, different variations of equation (41) are possible, as long as the coefficients sum up to one. These coefficients can be interpreted as the probabilities that goods produced in period t break down in the future periods t+1, t+2, t+3,… In general, we can write:

$$w_t = \sum_{\tau=1}^{t-1} \phi_\tau q_{t-\tau} \quad \forall t = 1,2,\dots,T \qquad (42)$$

In equation (42), parameter ϕ_τ represents the breakdown probabilities, which should sum up to one so that $\sum_{\tau=1}^{T} \phi_\tau = 1$. This approach is sometimes called the residence time or population balance model (Müller et al., 2014) and different statistical density functions can be used to model the lifetime of the consumption good, like the commonly used bathtub curve for example. Another option that can be used to set up a durable relationship between waste and past consumption is to use a so-called 'in use stock' (IUS) or accumulation relationship. In this case, the IUS in period t would be modelled as:

$$\mathrm{IUS}_{t+1} = \mathrm{IUS}_t + q_t - w_t \quad \forall t = 1,2,\dots,T \tag{43}$$

As can be seen in equation (43), the function is recursive and the IUS in period t consists of all material supplied to the market up to and including period t (inflow). As waste is extracted from the material flow for the purpose of recycling, the corresponding waste volume is deducted from the IUS (outflow). Top-down and bottom-up approaches are both used in the literature to quantify the inflow and outflow of material contained in the IUS (Müller et al., 2014). With regard to the waste fraction that becomes available for recycling, it can then be assumed that a particular percentage α of the IUS so far is offered at the recycling plant:

$$w_t = \alpha \mathrm{IUS}_t \tag{44}$$

As shown in equation (34), only part of the waste that is offered at the recycling plant gets recycled, with the remaining residue part being landfilled. Therefore, in the model landfills increase according to the following equation of motion:

$$\mathrm{LF}_{t+1} = \mathrm{LF}_t + [1 - \gamma_t] w_t \quad \forall t = 1,2,\dots,T \tag{45}$$

In equation (45), parameter LF_t represents the amount of waste that has been landfilled up to period t.

Finally, environmental externalities can be linked to different stages of the material flow like the virgin material extraction (q_t^v), the recycling process (q_t^r) or production of substitute material (q_t^s). In addition to flow pollution problems, stock pollution problems can also be modelled; for example, if landfills (LF_t) would cause negative environmental externalities. The framework can also accommodate externalities linked to the use phase of the consumption good (Q_t). In general, we write the environmental externalities as follows:

$$e_t = \varepsilon^v q_t^v + \varepsilon^r q_t^r + \varepsilon^s q_t^s + \varepsilon^{LF} \mathrm{LF}_t + \varepsilon^Q Q_t \tag{46}$$

5.2.3. Monopolist mine owner

In section 5.2.1, the representative mining company, recyclers and producers of the substitute material were all assumed to operate in a competitive, decentralised market setting. However, for the mining of virgin material in particular, it is often difficult to maintain the assumption of competitive market behaviour given the high level of market concentration. Therefore, it would be interesting to analyse alternative market structures, in particular monopolistic virgin resource owners. A monopolistic mine owner faces a more complex optimisation problem. First of all, like any monopolist, he or she can influence the instantaneous equilibrium market price by altering its supply. However, the virgin material residual demand is defined as total market demand minus the demand served by recycled and substitute material. The output choice of the monopolist virgin material supplier influences the material's price, which will also have an effect on recycling efforts being made and substitute material supply possibly. Secondly, a forward-looking monopolist must take into account the impact that his or her current supply of virgin material has on the availability of waste that forms the input for the recycling industry in subsequent periods. Because derivation of explicit first-order conditions for this scenario is complicated,[5] we programmed an explicit maximisation problem to solve the monopolist's profit maximisation problem, taking into account the supply behaviour of substitute material producers and recyclers, both immediately and in the future. Hence, the profits of the mine owner are defined as the sum of the discounted profit flows:

$$\max_{q_t\,(t=1,2,\ldots,T)} \pi^v = \sum_{t=1}^{T} \beta_t [P(Q_t) - c_t^v - t^v] q_t^v \quad (47)$$

In equation (47), parameter β_t still represents the private discount factor. This discount factor might be different from the social discount factor that is used in the first-best welfare scenario below. As the monopolist takes into account the fact that part of the total supply comes from the recycled and substitute material

[5] See Swan (1980) for an interesting theoretical model of a monopolist anticipating future recycling of its material. Note, however, that this is not a Hotelling-type model but instead focusses on steady-state solutions in the absence of exhaustibility constraints.

suppliers, the first-order conditions of these alternative suppliers are included in the model as constraints.

5.2.4. First-best welfare optimisation

Apart from the market scenarios defined above, we also consider a welfare optimisation scenario. In order to be able to formulate the first-best welfare optimisation problem, we must first define the social welfare function. In the model, social welfare is defined as the sum of utility minus the production costs of the virgin, substitute and recycled material suppliers and the cost of all externalities during the lifetime of the good. Taxes and subsidies are left out of this equation, as these are just redistributions of income and profits. This gives us following equation, with variable W representing welfare:

$$W = \sum_{t=1}^{T} \tilde{\beta}_t \left[U_t - c_t^v q_t^v - c_t^s q_t^s - r(\gamma_t) w_t - e_t\right] \quad (48)$$

Note that in equation (48), a social discount factor ($\tilde{\beta}_t$) is used instead of the private discount factor β_t. In practice, companies often employ a higher discount rate than social planners because they account for risk and are under pressure from their investors to deliver short-term returns (Jagannathan et al., 2016). According to the Hotelling rule, the higher discount rate implies a more rapid exhaustion of a non-renewable resource stock, leaving less for future generations. In turn, this implies that remaining resource stocks are exploited at a faster rate than is socially efficient.

In addition to the social discount factor, equation (48) takes into account externalities that arise at different stages of the materials' life cycle (virgin material extraction, recycling, landfilling). These externalities are represented by parameter e_t and were defined in expression (46) above.

5.3. Methodological example simulations

In order to underline the generic applicability of the model, this chapter elaborates a numerical example and shows typical outcomes and results that can be generated based on the theoretical underpinnings presented in the previous chapter.

5.3.1. Case description

The input parameters used in this chapter are based on a realistic case in which a non-renewable resource is extracted and used in the production of a consumption good. To make the descriptions more clear and intelligible, we refer to this non-renewable resource as sand. Given knowledge of the different equations presented in section 5.2, together with the different case input parameters shown in Appendix C, it is possible to obtain example results with respect to initial, interim and final market prices; shadow prices; recycling efforts; and supplied volumes of virgin, substitute and recycled sand. The time horizon to the time of remaining virgin sand reserve exhaustion, T, is unknown and is treated as an endogenous variable. To resolve the optimisation problems, GAMS modelling software was used, in line with previous studies (Caplan, 2004; Conrad, 1999; Flakowski, 2004). For this GAMS implementation, a mixed complementarity program (MCP) format was adopted. By using first-order conditions to set up the model, the main advantage of this kind of formulation lies in its flexibility and speed in solving complex economic models. In this chapter, non-linear programming is particularly used to derive optimal price and extraction paths (Flakowski, 2004).

5.3.2. Simulation results

Figure 10 shows the market price of sand for the different market scenarios. As this figure shows, the price path in the first-best welfare scenario is slightly different from the one of the competitive market scenario. This difference is caused by a divergence in recycling, as more recycling takes place in the first-best welfare scenario. In the competitive scenario, the market price of sand increases from about €5/ton to €12/ton, which is equal to the choke-off price level. As the marginal cost of mining sand is assumed to be constant, the market price path in the competitive scenario follows the Hotelling rule, with the shadow price increasing over time at the rate of discount. With regard to the period of full virgin sand reserve exhaustion, the value obtained for parameter T equals 57, which means that it takes 57 periods before the initial remaining virgin sand reserve is completely exhausted.

Looking at the monopolistic scenario, Figure 10 demonstrates that the monopolist will restrict output, resulting in an initial market price that is higher than in competitive markets. However, as the comparison with the competitive scenario also shows, the rate of price increase is slower and, eventually, an effect of this monopolistic market is to increase the time horizon over which the sand is extracted. As a result, in the monopolistic scenario it takes 91 periods to fully deplete the initial virgin sand reserve. Although this can give the impression that a monopolist mitigates the scarcity issue, it is important to realise that market power is not the right way as the limited supply of sand leads to welfare losses. This is confirmed by the welfare figures included in Table 11 below.

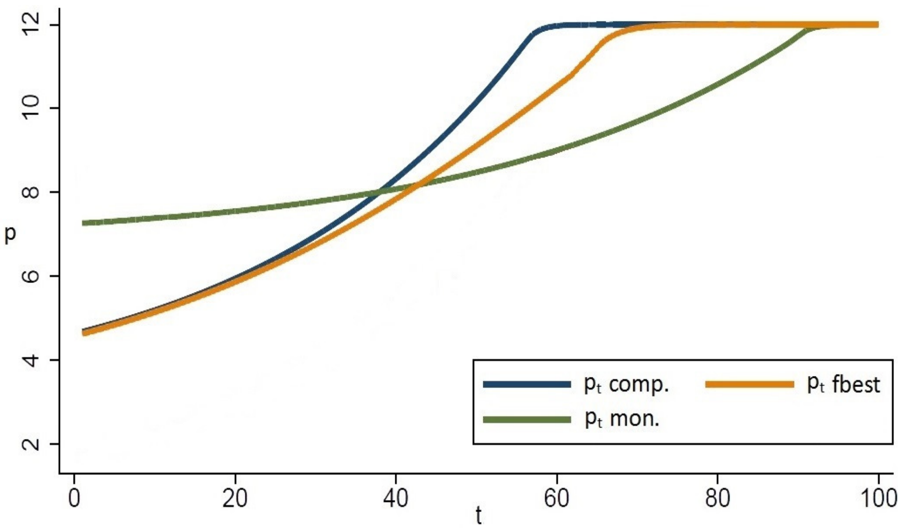

Figure 10: Market price (€)

Connected with the price graph in Figure 10, Figures 11, 12 and 13 show the supplies of virgin (q_t^v), substitute (q_t^s) and recycled (q_t^r) sand for respectively the competitive, monopolistic and first-best welfare scenario. These figures also show the total supply (q_t), which is equal to the sum of the different sorts of supply. Again, it can be observed that a monopolist spreads his or her extraction activities more over time by restricting output and raising prices initially, thereby deferring the time of complete exhaustion of the virgin sand reserve further into the future. It is important to note that the supply curves of substitute material, such as imported sand, coincide with the x-axis and are therefore equal to zero in this simulation example. This is a consequence of the fact that, in the current model example, the cost of supplying substitute material is set higher than the choke-off price. As a result, the substitute never comes into the market and its supply volume is equal to zero.

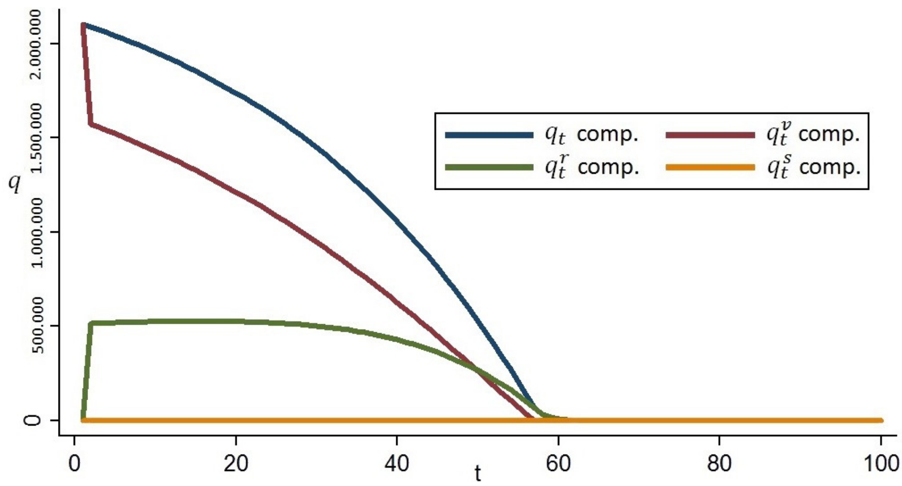

Figure 11: Virgin, substitute, recycled and total sand supply (tons), competition

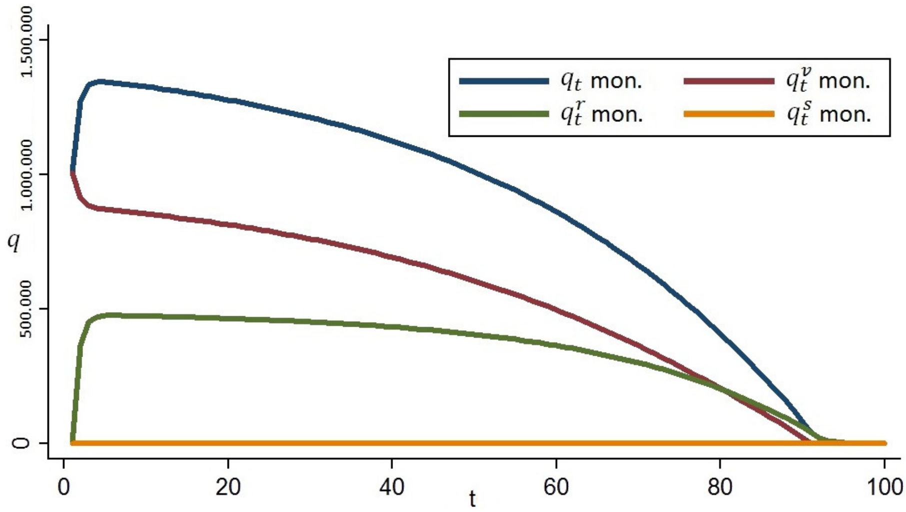

Figure 12: Virgin, substitute, recycled and total sand supply (tons), monopoly

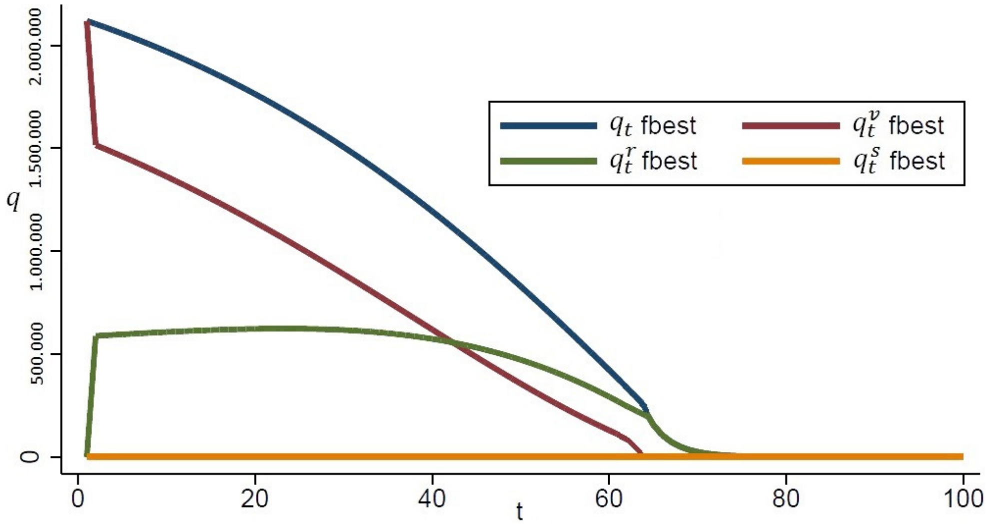

Figure 13: Virgin, substitute, recycled and total sand supply (tons), first-best welfare

Figure 14 shows the recycling efforts (γ_t) that the price-taking recyclers choose to maximise their profits. As the figure shows, the first-best welfare maximisation scenario generates the highest recycling efforts. In this scenario, more recycling takes place and the share of recycled supply relative to the total supply is larger than in the competitive scenario. This causes the exploitation period of virgin sand reserve to be longer in the first-best welfare scenario than in the competitive scenario. However, the monopolistic scenario dominates with regard to postponing the moment of virgin sand reserve depletion. It can be seen from the above figures and Figure 14 that although recycling effort γ_t increases over time, the supply of recycled material q_t^r decreases over time. This is caused by the fact that total supply q_t also decreases, thereby reducing the total amount of end-of-life waste that becomes available for recycling.

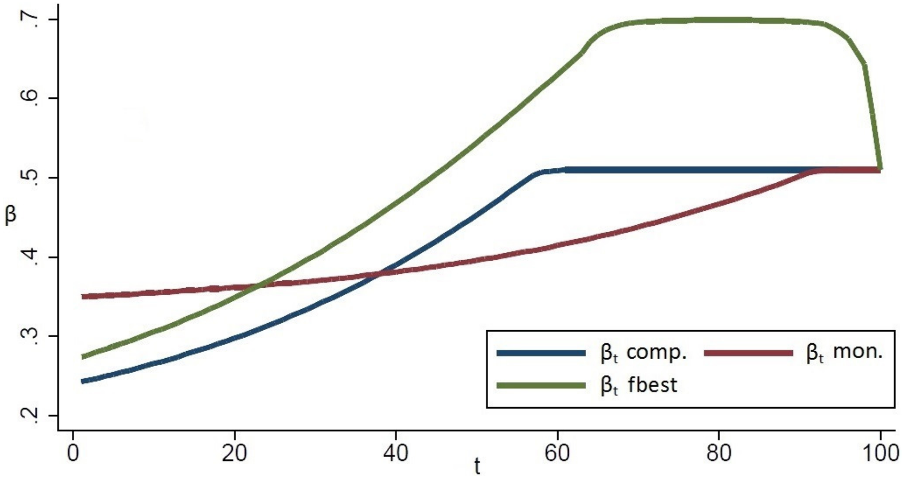

Figure 14: recycling effort γ_t (%)

Table 11 summarises all of the above figures and provides an overview of the most important simulation results generated to date.

Table 11: Simulation results without substitute

Parameter	Competition	Monopoly	First-best
T (periods)	57	91	64
$q_{tot.}^v$ (tons)	52,323,999	52,323,999	52,323,999
$q_{tot.}^r$ (tons)	24,464,174	33,035,173	34,284,201
$q_{tot.}$ (tons)	76,788,173	85,359,172	86,608,200
$W_{tot.}$ (€)	304,867,890	294,066,759	329,964,544

In all of the above figures, substitute materials do not come onto the market because the cost of supplying them is higher than the actual choke-off price. In terms of the competitive scenario, this situation can also be represented by Figure 15, with the brown colored area representing the total amount of virgin sand extraction and the green area representing the total amount of supplied recycled material. In Figure 15, the total volume of supplied virgin sand that is represented by the brown area is equal to the initial virgin sand reserve.

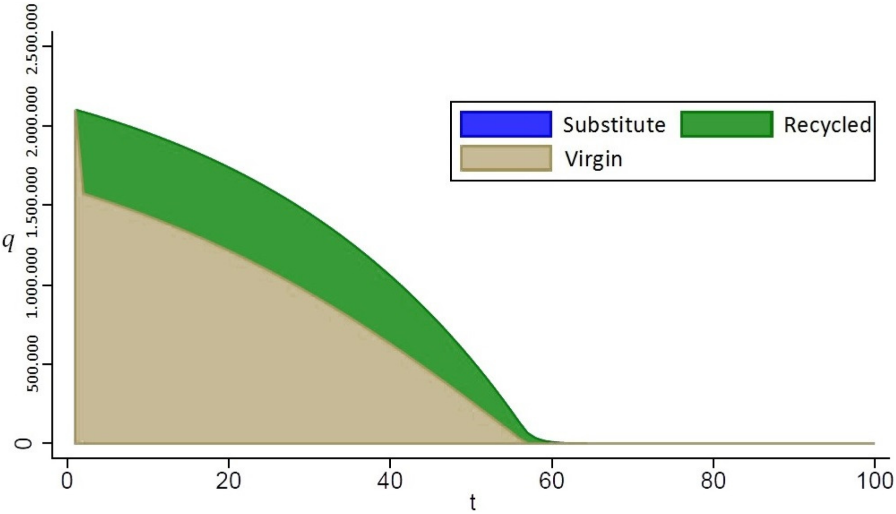

Figure 15: supplies in competitive scenario, no substitute (tons)

However, in constructing Figure 16, changes were made with respect to the cost of supplying substitute material (from €50/ton to €10/ton) so that this cost becomes lower than the initial choke-off price (€10/ton versus €12/ton). As a consequence of this change, the substitute comes onto the market when the market price reaches a level that makes it economically interesting for substitute suppliers to put their substitute for virgin sand onto the market. When this substitute comes onto the market, its full price functions as a new choke-off price at which a switch is made from virgin to substitute material. Still focusing on the competitive scenario, Figure 16 shows that, in this simulation, the substitute comes onto the market after 50 periods. At that moment, the switch is made from virgin to substitute material and the full price of the substitute becomes the new choke-off price. From that point on, no more virgin sand is mined. As the substitute material suddenly takes over the role of virgin material, the initial remaining virgin sand reserve is not fully exhausted in this simulation. This implies that the total virgin sand extraction represented by the brown colored area in Figure 16 is no longer equal to the initial virgin sand reserve.

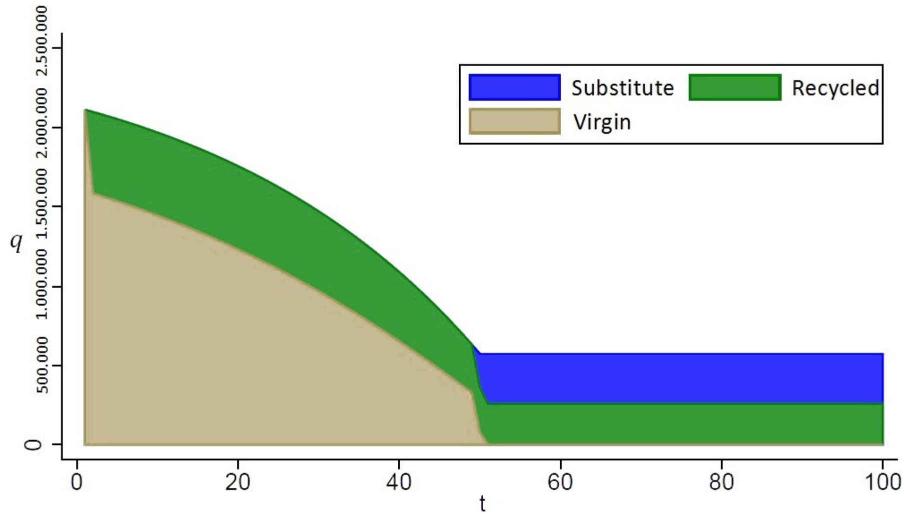

Figure 16: Substitute material in the competitive scenario (tons)

Table 12 shows how the simulation results change when the substitute material makes it to the market. Substitute, recycled and total material quantities were calculated using a timeframe of 100 periods, in accordance with the presented figures.

Table 12: Simulation results with substitute

Parameter	Competition	Monopoly	First-best
T (periods)	50	82	43
$q^{v}_{tot.}$ (tons)	51,291,998	51,448,287	46,146,017
$q^{s}_{tot.}$ (tons)	16,070,572	5,708,448	39,031,980
$q^{r}_{tot.}$ (tons)	36,335,063	36,966,120	57,718,415
$q_{tot.}$ (tons)	103,697,633	94,122,855	142,896,412
$W_{tot.}$ (€)	326,278,208	302,723,287	358,184,755

Following the above simulations, it would be interesting to investigate what would happen if the values of some other input parameters changed. Given the enormous variety of possible simulations, the multiplicity of parameter interactions and corresponding effects, and with our aim of presenting some explicable and intelligible results, we chose to make some further simulations to examine how results change when, for example, policy would levy an extraction tax (t^v) of €2/ton and when the private discount rate would be higher than the social one. In these simulations, we still let the substitute come onto the market. Beginning with the last adjustment, the private discount rate is raised to 10%, with the social discount rate still being equal to 3% (the value of parameter δ increases from 0.03 to 0.10). In the competitive scenario, these adjustments result in the equilibrium price path having a steeper slope than before. This is a logical consequence of the rise in the private discount rate as, according to the Hotelling rule, prices grow at the market interest rate. The steeper slope means that the new choke-off price level is reached more quickly than before. This implies that the length of time for which virgin sand is mined is shorter than before and equals 35 periods. The substitute comes earlier onto the market, causing the extraction of virgin sand to cease after just 35 periods. The same reasoning applies in the monopolistic scenario. The time horizon over which the sand is extracted is still longer than in the competitive scenario but shorter than before and equals 64 periods. The same social discount rate as before applies in the first-best welfare maximisation scenario. For this reason, the corresponding price path does not change and was not included in Table 13.

Building upon the simulation model with the higher private discount rate, we examined what would happen when a tax of €2/ton would be levied on the extraction of virgin sand (the value of parameter t^v increases from €0 to €2/ton). Logically, we found that an extraction tax would be equivalent to a rise in the extraction costs. Consequently, the introduction of an extraction tax raises the initial market price and slows down the rate at which this market price increases. It is important to note that the Hotelling rule remains valid, so the shadow price still increases over time at the rate of discount. Because the market price increases less rapidly than before, it takes longer before the choke-off price level is reached. In the competitive scenario, the extraction tax lengthens the time until the virgin sand reserve is exhausted from 35 to 45 periods. Regarding the monopolistic scenario, it takes 20 more periods to fully deplete the initial reserve when an extraction tax of €2/ton is imposed. Table 13 presents an overview of the various simulations described above, including Table 12 as a comparative basis.

Table 13: Simulation overview

		$c^s = 10$	$c^s = 10$, $\delta = 10\%$	$c^s = 10$ $\delta = 10\%$, $t^v = €2/ton$
Competition	T (periods)	50	35	45
	$q^v_{tot.}$ (tons)	51,291,998	52,036,993	51,985,372
	$q^s_{tot.}$ (tons)	16,070,572	20,851,557	17,713,981
	$q^r_{tot.}$ (tons)	36,335,063	33,290,875	36,180,395
	$q_{tot.}$ (tons)	103,697,633	106,179,425	105,879,748
	$W_{tot.}$ (€)	326,278,208	307,590,863	325,015,248
Monopoly	T (periods)	82	64	84
	$q^v_{tot.}$ (tons)	51,448,287	52,214,352	52,082,032
	$q^s_{tot.}$ (tons)	5,708,448	11,863,672	5,391,312
	$q^r_{tot.}$ (tons)	36,966,120	37,882,666	37,498,705
	$q_{tot.}$ (tons)	94,122,855	101,960,690	94,972,049
	$W_{tot.}$ (€)	302,723,287	317,266,585	289,075,994

5.4. Conclusions

Debates on supporting the transition towards a more resource-efficient and low-carbon economy have focused on how to identify optimal extraction paths over time for any particular non-renewable resource reserve. This chapter adds value to this discussion by developing and introducing a generic numerical optimisation model that can be used to simulate the effects that different policy instruments can have within the material flow of a particular substance. The model is flexible enough to allow for different assumptions regarding behaviour of market participants (profit maximisation in a competitive or monopolistic market setting). The modelling framework is also capable of comparing decentralised market-based scenarios with social welfare maximising scenarios.

By using a fixed initial non-renewable resource reserve, a kind of cake-eating model was built, similar to the well-known Hotelling model. Several extensions were added that, to our knowledge, had never previously been combined together with such a Hotelling model. The first extension relates to the inclusion of a recycling sector in which recyclers choose a recycling effort in order to maximise profits. Consequently, recycling is an endogenously defined function within the optimisation model. The second extension concerns the feature of allowing for the possibility that a substitute material can come onto the market at a fixed price. If such a substitute – such as imported material from abroad – came onto the market, its full price would function as a choke-off price level at which the switch is made from virgin to substitute material. This substitute would actually form a third supply source, next to virgin and recycled material. Throughout the developed model, the full material flow system that includes these different supply sources is taken into account by imposing appropriate material balance constraints. Thirdly, we introduced different policy instruments (extraction, production or consumption taxes, waste taxes, etc.) that can be used to adjust for different environmental externalities linked to different stages of the material's life cycle. Fourthly, different degrees of product durability can be simulated by selecting different functional relationships between past consumption and future waste generation.

As the various simulation examples have shown, the results are all in line with expectations based on theoretical insight and intuitive reasoning. This indicates that the developed model is able to produce meaningful results that are based on a well-founded, realistic and stable methodological structure. In addition, the model is capable of quantifying effects that are very hard to assess in purely theoretical models. An example is the impact on market prices, recycling efforts and date of exhaustion of virgin material reserves in the case of a farsighted monopolist producer of virgin material who anticipates future recycling of the waste containing the material that he or she brings to the market today. It is important to note that the optimisation model was developed to be generically applicable. By this generic design, the provided optimisation model can be of great value to policy makers in the field of designing and supporting sustainable practices for all sorts of non-renewable resources. After all, input parameters can be adapted to different circumstances and characteristics that make the set of non-renewable resources rather heterogeneous, and appropriate formulations can be used to simulate competitive, monopolistic or first-best welfare optimisation scenarios. With all these features, the developed model can be used to make countless simulations as a means to support a transition towards a more resource-efficient, circular economy.

Although the modelling framework adds value to the existing literature, we are aware of different possibilities for future research in this area. We believe that the most important areas are the following. First, the model could be expanded to allow for different jurisdictions that are capable of setting their own policy instruments, in order to maximise their domestic welfare. Such a model could be used to investigate the international policy competition, perhaps leading to a "race to the bottom" in externality taxes. Secondly, in the current version of the model, producers of the consumption goods have only chosen production volumes and have not been able to adjust quality aspects of their goods, such as longevity and material intensity and green design. Allowing for a more realistic set of choices for producers would definitely enrich the model. Thirdly, instead of the zero-profit condition a lower bound could be imposed on the recycled material price parameter to assess what would happen in terms of for example waste accumulation when material prices would fall below this lower bound, causing negative profits for recycling firms that as an effect may leave the market. Fourthly, we assumed that there is no market for waste and that recyclers obtain all waste for free; however, waste can have a positive scrap value and recyclers would in that case be willing to pay a positive price for post-consumer waste as a secondary resource. In addition, depending on the waste regulation and technology assumption, it is also possible that consumers have to pay a price to dispose of their waste or a waste charge that can be used to cover recycling costs when the scrap value of goods would be negative. Hence, modelling the market for post-consumer waste in more detail constitutes an interesting avenue for future research. Finally, it could be interesting to take a closer look at the effects of recycling on material quality deterioration.

Each of these extensions will however add considerable complexity to the model. Only by using a consistent numerical simulation modelling framework such as the one we have presented in this chapter will it be possible to investigate these more complicated but realistic scenarios.

CHAPTER 6.

Conclusions

6.1. Research questions

In the past 30 years, global material extraction more than doubled from around 36 billion tonnes in 1980 to almost 85 billion tonnes in 2013. This is mainly caused by growing populations and an increasingly intensive and extensive use of natural resources. As all those natural resources are taken out of their natural deposits which are created by long-term geological processes, natural resource reserves continue to get scarcer as resources are extracted. In this regard, putting in place Sustainable Materials Management (SMM) policies is crucial as SMM can improve the environment by reducing the amount of resources human economic activities require as well as diminishing the associated impacts and improving resource security.

In practice, it is a challenge to identify optimal, sustainable policies that trigger the transition towards a resource-efficient, circular economy. A first reason for this is the fact that even though a proliferation of sustainability assessment methodologies allows analysis of all aspects of sustainability, it also induces confusion. After all, methodological differences and different weights for environmental, economic and societal aspects lead to conflicting assessment results for policy and business matters. Furthermore, while broadening the different methodologies to conduct sustainability assessments, fragmented developments by a variety of research disciplines led to vague distinctions and as a consequence, synergies between assessment methodologies became harder to identify. A second reason stems from the fact that unregulated resource and waste markets lead to socially-undesirable outcomes in the presence of scarcity and externality issues. In practice, these issues are rarely reflected in market prices. Even though environmental taxation can be a step towards reflecting the full external and social costs of resource extraction, utilisation and end-of-life practices, the shift from labour towards environmental taxation is not tackled at all in a lot of European Member States. Caused by, inter alia, the complexity of the decision making chain within which SMM policies have to operate and the mix of policy instruments that is required, SMM policies are distinctive which makes them harder to identify. This difficulty is further exacerbated by the heterogeneity of non-renewable resources and by the lack of appropriate economic optimisation modelling techniques that can be used to make relevant simulations.

Taking into consideration the challenge of identifying appropriate SMM policies, this dissertation focuses on examining the socio-economic aspects of SMM. More specifically, it studies the roles that sustainability assessment, economic optimisation methodologies and policy instruments may play in the transition towards SMM. In this regard, this dissertation provides knowledge with respect to the following main research question:

How can sustainability assessment and economic optimisation methodologies be designed and applied to support the transition towards Sustainable Materials Management?

This research question is operationalised by the different subquestions in sections 6.1.1. to 6.1.4. The subquestion of section 6.1.1. relates to the first part of the dissertation that structures existing micro level product and project related sustainability assessment tools. The following three subquestions, presented in sections 6.1.2. to 6.1.4., develop complementary economic optimisation modelling techniques as a means to support policy in developing and implementing sustainable practices with regard to scarce landfill and non-renewable resource reserve problems. By answering these subquestions, this dissertation handles the complementary methodological structure that consists of problem-oriented approaches like LCA, LCC and CBA, and systems-oriented approaches like economic optimisation and equilibrium modelling techniques. A graphical overview of this complementary structure is shown in Figure 17 below.

6.1.1. What are similarities, differences and connections between LCA, LCC and CBA sub-methodologies?

In order to interpret and integrate conflicting assessment results, the second chapter of this dissertation provided a structured framework that shows the interrelationships between different sub-methodologies that exist within widely applied methodologies such as LCA, LCC and CBA. This framework is shown in Figure 17, where it is presented within the complementary structure created in this dissertation. By using different kinds of arrows, the framework illustrates how different sustainability assessment tools interact. Full arrows indicate that different methodologies can be combined. Environmental LCA (eLCA) and environmental LCC (eLCC) for example do not only define system boundaries, time spans and functional units in a similar way, but they also share the steady state approach without discounting of impacts. This is important as eLCC is primarily set up as an assessment method that is carried out in combination with eLCA. As an eLCC only includes real money flows, the risk for double counting with environmental impacts included in eLCA is minimised. By applying the same system boundaries, time span, functional unit and steady state cost model as eLCA and eLCC, also a social LCA (sLCA) is compatible. Together, these three sub-methodologies form the relatively new and comprehensive tool called Life Cycle Sustainability Assessment (LCSA). With respect to the dashed arrows shown in the framework, they indicate that although a methodology can provide useful input to another methodology, both remain stand-alone methods. The monetised, non-internalised environmental costs for example that are included in a full environmental LCC (feLCC) can be identified by eLCA, but they both remain stand-alone methods in order to avoid double counting. When the framework includes '∈' as a symbol, this indicates that a methodology is part of another methodology. eLCC for example builds further on financial LCC (fLCC), by considering the full life cycle of products. Building on the framework, chapter 2 further focused on key aspects that need to be considered when different sub-methodologies are extended to integrated sustainability assessments. Looking at LCA for example, both eLCA and sLCA lack an economic dimension that can be found in the form of eLCC. When combining these tools however, some key aspects have to be taken into account. After all, used metrics and data requirements are different while scope and time frames are similar. With respect to Cost-Benefit Analysis (CBA), this methodology is also able to carry out full sustainability assessments. There are however some dissimilarities between LCA and LCC on the one hand and CBA on the other. Firstly, LCA and LCC can be catalogued as product related assessments while CBA mostly focuses on projects or policies. Secondly, LCA and LCC focus on the (full or economic) life cycles of the assessed products while CBA, by focusing on the lifetime of a particular project, makes the lifetime of used products secondary. Thirdly, while LCA and LCC are tools comparing different products, CBA typically calculates an NPV value that has an economic meaning compared to a zero reference. A fourth key aspect shows itself in the perspective on labour requirements. In sLCA and sLCC, job creation is usually regarded as a benefit, while an sCBA can consider labour as a cost when new jobs are created for already highly demanded skilled workers. A last key aspect is the way scarcity is dealt with. LCA sees scarcity as a societal problem and counts recycling or use reduction of a scarce material as a benefit, while in CBA, scarcity is not necessarily a problem as economic markets can internalise scarcity. In that case, prices represent the full cost to society if the private discount rates are comparable to the social optimal level. In addition to these key aspects, chapter 2 showed that important connections exist: (i) LCA, LCC and CBA can all cope with social inequality, (ii) processes such as valuation techniques for LCC and CBA are common, (iii) both LCA and CBA have a connection with Environmental Impact Assessment (EIA) and (iv) LCA can be used in parallel with LCC to form the LCSA method. Finally, chapter 2 referred to a study on automotive glass recycling which was carried out in Flanders (Belgium), with eLCA and fCBA results that point to different policy recommendations. By suggesting actions that can contribute to a more comprehensive understanding of this problem however, chapter 2 illustrated how the developed framework can help to overcome policy problems caused by confusing assessment results.

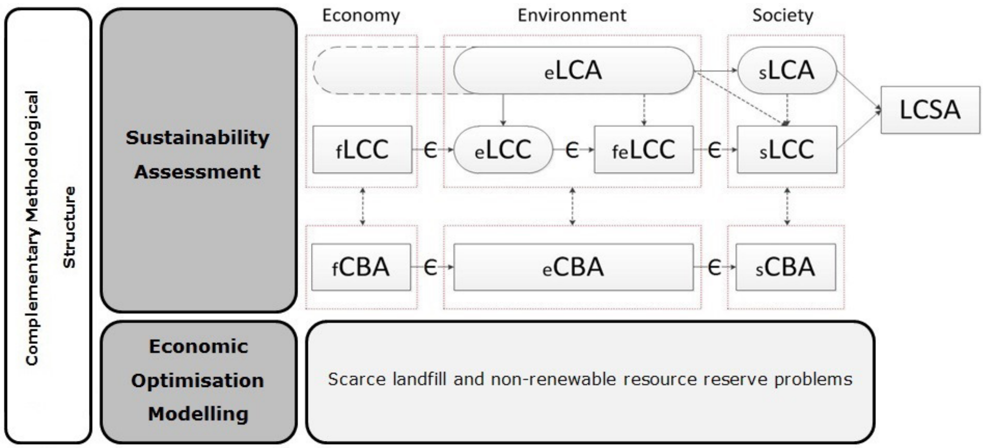

Figure 17: Assessment framework within complementary methodological structure

6.1.2. Do landfill taxes support a sustainable way of managing remaining landfill capacities and waste?

By developing a Hotelling based economic optimisation model, the third chapter of this dissertation provided a tool for identifying the best allocation of landfill volume over time and examining the effects landfill taxes have on scarce remaining landfill capacities. Carrying out example simulations based on Flemish data, chapter 3 defined optimal landfill and price paths by running the algorithm which includes maximising the profits of landfill companies taking into account that the sum of all yearly landfilled volumes should equalise the initial remaining landfill reserve volume. As could be seen, in our simulations the introduction of a landfill tax has the effect that yearly landfill volumes decrease considerably. When a landfill tax of 60€/ton would be used, it would take 42 years for full landfill reserve exhaustion to occur, whereas this period would be shortened to only 20 years with no landfill taxes. Although discounted total profit falls when landfill taxes are used, discounted total welfare increases considerably (from €132,563,772 to €239,109,115). This difference is mainly achieved by the value of the Marginal Cost of Landfill Taxation (MCLT) being smaller than the value of the Marginal Cost of Public Funds (MCPF). With regard to the optimal landfill tax level from a welfare maximisation point of view, it was shown that this tax level is equal to 55€/ton. Compared with the results of the scenario in which the landfill tax was equal to 60€/ton, this optimal tax level also results in a higher profit figure and shortens the period until landfill capacity exhaustion with five years. With respect to the total level of externalities, only the discounted values are smaller when landfill taxes are being used, as eventually the same amount of waste is landfilled which is equal to the initial remaining landfill capacity. To conclude, we can say that within our simulation structure, the added value of a landfill tax -from a broad societal point of view and knowing that remaining landfill capacity is scarce- is considerable in terms of welfare gain. In practice however, care should still be taken not to jeopardise the competitiveness of the concerned industry.

6.1.3. How to combine landfill taxes and EWM in order to enable sustainable ways of processing future waste streams?

Building on the third chapter, chapter 4 went one step further by including the so-called Enhanced Waste Management (EWM) concept. In this chapter, economic optimisation modelling techniques were developed that can be used to analyse how landfill taxation schemes can be adjusted and transformed into waste taxes in case EWM would be applied, thereby creating a combination that enables sustainable ways of processing future waste streams. As the first results in chapter 4 showed, within our simulation structure applying EWM practices has the effect of significantly postponing the moment of full capacity exhaustion. In case the taxation level would stay at the current level of €42/ton, applying EWM would postpone the moment of capacity exhaustion with 109 years compared to the as-is scenario without EWM (from 14 to 123 years). Next to these results, the fourth chapter investigated which waste tax would be optimal from a societal point of view, when EWM is applied. In this case, it was shown that a waste tax of €50/ton is optimal from a societal point of view. Higher tax levels would not only lower total profit, but also total welfare figures. Although this optimal waste tax level is higher than the average landfill tax level of the as-is scenario without EWM, the optimal waste tax level should actually be compared with that landfill tax level that optimises welfare in case no EWM practices would be applied, being €93/ton. In comparison with this landfill tax level, the optimal waste tax level of €50/ton refers to the positive external effects that are generated by

EWM practices. Comparing current taxation levels with optimal ones, chapter 4 showed that within our simulation structure, present policy approaches the optimum as the current average landfill tax level is not that much different from the optimal waste tax of €50/ton. It is however important to note that this optimal tax level only maximises welfare in case EWM practices are applied. Would no such practices be applied, a higher landfill taxation level of €93/ton would be justifiable. Consequently, within our simulation structure, and as the current Flemish landfill tax is slightly lower than these optimal levels, the choice that could be made is to further increase taxation levels or show commitment to EWM practices. With regard to the profit levels of the EWM scenarios, chapter 4 showed that they are both significantly lower than the profit level of the as-is scenario without EWM. This is even the case would landfill tax levels in the no EWM scenario be equal to the optimal level of €93/ton. Regarding the high price levels in the as-is scenario, it can be questioned however whether these prices predicted by the Hotelling rule will increase that high in practice. After all, it is possible that landfill operators may lower their prices in order to attain the fixed landfill volume that was assigned to them in their landfill operating permit. In addition, it can happen that pressure on politicians becomes higher when exhaustion of remaining landfill capacity is near. This might have the effect that additional capacity is created, thereby making the scarcity issue less pronounced and tempering the increase in prices. If it would still turn out that measures in support of EWM practices are needed, fiscal instruments such as tax credits could be used to boost the profit levels of EWM operators. At the same time, such instruments could serve as a control measure to assure that EWM practices are actually carried out. Based on the results of the Flemish example simulations, there are some lessons that can be generally applicable. By providing these lessons, this chapter could provide relevant information about optimising waste management systems within circular economy contexts.

6.1.4. How to optimally extract scarce non-renewable resource reserves over time, taking into account recycling and substitutes?

Continuing in the context of scarce reserves but focusing on the scarcity of non-renewable resource reserves, chapter 5 adds value to the discussion on how to optimally extract these resource reserves by developing and introducing a generic numerical optimisation model that can be used to simulate the effects that different policy instruments can have within the material flow of a particular substance. By using a fixed initial non-renewable resource reserve, a kind of cake-eating model was built, based on the well-known Hotelling model. Several extensions were however added that, to our knowledge, were never combined all together with such a Hotelling model before. The first extension relates to the inclusion of a recycling sector in which recyclers choose a recycling effort in order to maximise profits. Consequently, recycling is an endogenously defined function within the optimisation model. The second extension concerns the feature of allowing for the possibility that a substitute material can come onto the market at a fixed price. If such a substitute – such as imported material from abroad – came onto the market, its full price would function as a choke-off price level at which the switch is made from virgin to substitute material. This substitute would actually form a third supply source, next to virgin and recycled material. Throughout the developed model, the full material flow system that includes these different supply sources is taken into account by imposing appropriate material balance constraints. As a third extension, chapter 5 introduced different policy instruments (extraction, production or consumption taxes, waste taxes, etc.) that can be used to adjust for different environmental externalities linked to different stages of the material's life cycle. Fourthly, different degrees of product durability can be simulated by selecting different functional relationships between past consumption and future waste generation. Next to these extensions, the developed model is flexible enough to allow for different assumptions regarding behaviour of market participants (profit maximisation in a competitive or monopolistic market setting). Moreover, the modelling framework is capable of comparing decentralised market-based scenarios with social welfare maximising scenarios. A numerical example about sand extraction is included in the fifth chapter to show outcomes and results that can be generated. Regarding the simulation results, chapter 5 revealed intuitive comparative static results: the higher the price of virgin material and the higher the full cost of disposing recycling residues, the higher the recycling effort chosen by a profit-maximising recycling firm. Looking at different market scenarios, the fifth chapter showed that a divergence in recycling efforts causes the price path in the first-best welfare scenario to be slightly different from the one of the competitive market scenario. After all, in the first-best welfare scenario, more recycling takes place. With respect to the monopolistic scenario, it was demonstrated that the monopolist restricts output, thereby creating an initial market price that is higher than in competitive markets. However, as the comparison with the competitive scenario also showed, the rate of price increase is slower and, eventually, an effect of the monopolistic market is to restrict output and increase the time horizon over which the sand is extracted. As a result, whereas it takes 57 periods before the initial remaining sand reserve is completely exhausted in the competitive scenario, in the monopolistic scenario this takes 91 periods. Although this can give the impression that a monopolist mitigates the scarcity issue, it is important to realise that market power is not the right way as the limited supply of sand leads to welfare losses. This was confirmed in chapter 5 by the corresponding welfare figures. When the cost of supplying substitute material becomes lower than the initial choke-off price, the substitute comes onto the market when the market price reaches a level that makes it economically interesting for suppliers to put their substitute for virgin sand onto the market. When this happens, the full price of the substitute functions as a new choke-off price at which a switch is made from virgin to substitute material. Consequently, in that case extraction of virgin material ceases earlier than in the case where no substitute comes onto the market. With regard to a private discount rate, often higher than a social one, chapter 5 included some extra simulations to show that a higher private discount rate leads to steeper slope price paths. As a result of this steeper slope, the choke-off price level is reached more quickly than before, implying that the length of time for which virgin sand is mined is shorter than before. Finally, the simulations in chapter 5 showed the consequences of levying a tax on the extraction of virgin sand. In both the competitive as well as the monopolistic scenario, levying this tax has the same effect as a rise in the extraction costs. Consequently, the introduction of an extraction tax raises the initial market price, slows down the rate at which the market price increases and lengthens the time to complete exhaustion of the initial sand reserve. The fact that all simulation results presented in chapter 5 are in line with expectations based on theoretical insights and intuitive reasoning indicates that the developed model is able to produce meaningful results that are based on a well-founded, realistic and stable methodological structure. In addition, the model is capable of quantifying effects that are very hard to assess in purely theoretical models. All this, together with the generic design of the model, ensures that the developed model can be used to make countless simulations as a means to design and support sustainable practices for all sorts of non-renewable resource practices.

6.2. Policy recommendations

Based on the contents of this dissertation, insights can be formulated that can support policymaking in developing and implementing sustainable materials management practices, thereby triggering the transition towards a resource-efficient, circular economy. With regard to the first part of the dissertation, consisting of chapter 2, it was shown that although different sustainability assessment methodologies can be applied in a rigorous way, they might lead to conflicting assessment results for many policy and business issues. To overcome this problem, the second chapter presented a framework that structures the interrelationships between different sub-methodologies and suggested actions that can contribute to a better understanding of case study results. Looking at these actions, it is important to be as comprehensive as possible when comparing different alternatives. This means including the different pillars of sustainability, namely environment, economy and society. In this respect, the developed framework could come in handy by showing which sub-methodologies can and cannot be combined when searching for an appropriate combination of methodologies. Would one want to carry out a project evaluation from the viewpoint of society as a whole for example, a social CBA could assess the environmental, social and financial aspects of the project in one coherent methodology. On the other hand, would one want to compare the sustainability of different products, the environmental LCC, environmental LCA and social LCA sub-methodologies are sufficiently similar so that they can be used in a complementary way. Software tools are being developed to integrate the different pillars of sustainability more efficiently in practice. Policy makers should be aware however, of the fact that such tools are still relatively new. Consequently, permanent vigilance is needed when using such recent developments for decision-making purposes.

In the second part of this dissertation, chapter 3, 4 and 5 developed economic optimisation modelling techniques as a means to support policy in developing and implementing sustainable practices with regard to scarce landfill and non-renewable resource reserve problems. Within the simulation structures of chapters 3 and 4, it was shown that landfill taxes reduce landfilled waste volumes and thereby postpone the moment of full remaining landfill capacity exhaustion. Furthermore, it was shown how landfill taxation schemes could be adjusted and transformed into waste taxes in case EWM would be applied. Within our simulations it was indicated that, in comparison with an optimal landfill tax level, the optimal waste tax would be lower, referring to the positive external effects that are generated by EWM practices. With respect to the absolute height of taxation levels, both chapter 3 and 4 provide an important insight. Often it is believed that the higher the tax level, the more effective waste management improvements can be realised (Krook et al., 2012). Within the simulation structure of chapters 3 and 4 however, it has been shown that this is not necessarily the case. After all, when a particular taxation level is reached, higher levels would not only lower total profit but also total welfare. In other words, a turning point is reached from where total welfare starts to decrease. Another lesson to be learned is the fact that policy should take into account and keep an eye on the extent to which the landfill capacity scarcity issue is reflected into much higher prices and profits. If it would turn out that measures in support of EWM practices are needed, fiscal instruments such as tax credits could be used. By transferring part of the tax revenues back to EWM operators, their profit levels could be boosted thereby supporting EWM operations. In addition to this redistributive function, another advantage of such an instrument could be a controlling function as credits could only be assigned after certain EWM practices are actually carried out. Although the simulation results in chapter 3 and 4 apply to the Flemish situation, the insights that were provided can be generally applicable. Consequently, these insights could be considered in debates on policy interaction and optimal policy mixes with respect to a transition towards sustainable waste management practices within circular economy contexts.

In chapter 5, a generic economic model was developed that can be used to simulate non-renewable resource practices. Next to including recycling, substitution and waste accumulation in a unified framework, the model lends itself to simulating numerous situations with different market settings, policy instruments and environmental externalities for example. As all simulation results were in line with expectations based on theoretical insights and intuitive reasoning, the developed model was approved to produce meaningful results that are based on well-founded, realistic and stable methodological structure. This reinforces the practical usefulness of the developed model in helping policy makers design policies that support the transition towards a resource-efficient economy. This shift towards a more resource-efficient path should bring competitiveness and new sources of growth and jobs through cost savings from improved efficiency and better management of resources over their whole life cycle.

6.3. Future research

In chapter 2, different key aspects and connections were discussed that should be taken into account when future efforts are made to broaden LCA, LCC and CBA to full sustainability assessments. In broadening these methodologies and combining different sub-methodologies, economic monetisation of environmental and societal aspects is often an obstacle. As a consequence, it is advisable to move towards generally agreed monetisation standards, in order to achieve well-founded results that can be compared with each other in a reliable way. Obviously, this is a big challenge for both research and policy; but it is important would one want to identify the most sustainable products and projects from an environmental, economic and social point of view. Along with tackling this monetisation issue, further research and permanent vigilance is needed to develop more comprehensive tools that allow to integrate the three pillars of sustainability more efficiently in practice. Methods like 'Ecocosts 2012', 'Ecovalue08' and 'Stepwise 2006', that allow valuing outputs of a life cycle inventory by converting them to eco-costs, show that integration is not an insurmountable problem. However, such methods should be further developed before one can speak of a fully integrated sustainability assessment methodology.

As to the economic optimisation modelling techniques that focused on scarce landfill and non-renewable resource reserve problems, further research can be done by taken into account evolving prices, costs, capacities, accessibilities, qualities and technologies. This way, policy can constantly be adapted to prevailing market conditions. Furthermore, this also allows that future decisions can be made in the most sustainable and socially responsible way, thereby ensuring that policy makers can address the needs of a resource-efficient, circular economy in an optimal way. Further research can also be undertaken to further refine the models that were developed. In this context, attention could go to different market structure types and it would be interesting to investigate the effects of relaxing the no free waste disposal assumption. Specifically with respect to the optimisation model that focuses on optimal landfill capacity use and waste management practices, aspects that deserve further attention are for example the extra landfill capacity that could be created by valorising historic landfilled waste streams and the creation of more capacity when pressure on politicians becomes higher when an exhaustion of remaining landfill capacity is near. These aspects might have the effect that additional capacity is created, thereby making the scarcity issue less pronounced and tempering the increase in prices. Also with regard to the composition of future incoming waste streams, more refinement as regards the different input data would be valuable.

Looking at the developed generic optimisation model that focuses on the extraction of non-renewable resources, different possibilities for future research exist. Firstly, the developed model could be expanded to allow for different jurisdictions that are capable of setting their own policy instruments, in order to maximise their domestic welfare. Such a model could be used to investigate the international policy competition, perhaps leading to a 'race to the bottom' in externality taxes. Secondly, in the presented version of the model, producers of the consumption goods have only chosen production volumes and have not been able to adjust the quality aspects of their goods, such as longevity, material intensity and green design. Allowing for a more realistic set of choices for producers would definitely enrich the model. Thirdly, instead of the zero-profit condition a lower bound could be imposed on the recycled material price parameter to assess what would happen in terms of for example waste accumulation when material prices would fall below this lower bound, causing negative profits for recycling firms that as an effect may leave the market. Fourthly, we assumed that there is no market for waste and that recyclers obtain all waste for free; however, waste can have a positive scrap value and recyclers would in that case be willing to pay a positive price for post-consumer waste as a secondary resource. In addition, depending on the waste regulation and technology assumption, it is also possible that consumers have to pay a price to dispose of their waste or a waste charge that can be used to cover recycling costs when the scrap value of goods would be negative. Hence, modelling the market for post-consumer waste in more detail constitutes an interesting avenue for future research. Finally, it would be interesting to take a closer look at the effects of recycling on material quality deterioration. Each of these extensions however, will add considerable complexity to the developed model. Only by using a consistent numerical simulation modelling framework, such as the one developed in this dissertation, will it be possible to investigate these more complicated but realistic scenarios.

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APPENDICES

APPENDIX A: environmental impacts automotive glass study

Source: (OVAM, 2013a,b)

Environmental impacts per kg recycled ELV glass, Shredding route		
Impact category	Unit	Value
Climate change	Kg CO2 eq	0.010
Ozone depletion	Kg CFC-11 eq	1.3E-09
Terrestrial acidification	Kg SO2 eq	2.4E-05
Freshwater eutrophication	Kg P eq	-1.1E-07
Marine eutrophication	Kg N eq	9.5E-07
Human toxicity	Kg 1,4-DB eq	-1.4E-04
Photochemical oxidant formation	Kg NMVOC	2.8E-05
Particulate matter formation	Kg PM10 eq	7.9E-06
Terrestrial ecotoxicity	Kg 1,4-DB eq	2.1E-06
Freshwater ecotoxicity	Kg 1,4-DB eq	-1.6E-06
Marine ecotoxicity	Kg 1,4-DB eq	7.3E-06
Ionising radiation	Kg U235 eq	-5.0E-04
Agricultural land occupation	M2a	-1.9E-05
Urban land occupation	M2a	-3.5E-04
Natural land transformation	M2	-3.4E-06
Water depletion	M3	-0.0013
Metal depletion	Kg Fe eq	-3.7E-04
Fossil depletion	Kg oil eq	0.0034

Environmental impacts per kg recycled ELV glass, Dismantling route		
Impact category	Unit	Value
Climate change	Kg CO2 eq	-0.28
Ozone depletion	Kg CFC-11 eq	-5.5E-09
Terrestrial acidification	Kg SO2 eq	-8.5E-04
Freshwater eutrophication	Kg P eq	-4.9E-05
Marine eutrophication	Kg N eq	-4.8E-05
Human toxicity	Kg 1,4-DB eq	-0.050
Photochemical oxidant formation	Kg NMVOC	-3.6E-04
Particulate matter formation	Kg PM10 eq	-2.3E-04
Terrestrial ecotoxicity	Kg 1,4-DB eq	-1.2E-05
Freshwater ecotoxicity	Kg 1,4-DB eq	-0.0012
Marine ecotoxicity	Kg 1,4-DB eq	-9.1E-04
Ionising radiation	Kg U235 eq	-0.031
Agricultural land occupation	M2a	-0.0094
Urban land occupation	M2a	-0.0010
Natural land transformation	M2	-1.3E-05
Water depletion	M3	-0.0063
Metal depletion	Kg Fe eq	-0.0090
Fossil depletion	Kg oil eq	-0.046

APPENDIX B: input data EWM

General data		Source
Discount rate (%)	10	Industrial reference
Remaining landfill capacity (ton)	9,083,513[a]	OVAM, 2015a
Slope inverse demand function	0.0001	Case study
Choke-off price (€/ton)	200	Case study
Average landfill tax (€/ton)	42[b]	OVAM, 2015b
Marginal landfill cost (€/ton)	15[c]	Case study
Unit externality of landfilling (€/ton)	60[d]	Case study
MCPF-MCLT	0.20[e]	Case study
Waste Composition	Output share	
Metals (wt%)	1.90	Characterisation study
Glass (wt%)	2.61	Characterisation study
Fines/Aggregates (wt%)	37.86	Characterisation study
Temporary Storage (wt%)	33.05	Characterisation study
Residue to be landfilled (wt%)	4.45	Characterisation study
RDF (wt%)	20.13	Characterisation study
EWM		
Handling cost (€/ton)	5	Danthurebandara et al., 2015a; Van Passel et al., 2013
Sorting cost (€/ton)	17	
Drying cost (€/ton)	3	
Fines treatment cost (€/ton)	5	
Thermal treatment cost (€/ton)	117	
Storage cost (€/ton)	9	
Ferrous metals (€/ton)	200	
Glass (€/ton)	6	
Aggregates (€/ton)	10	
Electricity (€/ton RDF)	90[f]	
Green certificates (€/ton RDF)	77.55[f]	
Energy efficiency (%)	27	
Green energy factor (%)	47	

a. Situation in 2014 with regard to Flemish category 2 landfills. When EWM practices are implemented, it is taken into account that space has to be reserved for the temporarily stored fraction.
b. it is assumed that 80% of all landfilled waste is combustible and 20% is non-combustible.
c. this includes storage and cover costs. When EWM practices are implemented, these are only applicable to the 4,45% residue that is to be landfilled.
d. only when no EWM practices are implemented. As EWM practices involve significant external benefits, this figure is decreased when they are implemented.
e. cautious estimate based on Barrios et al., 2013 and Danthurebandara et al., 2015a
f. based on an electricity price of €60/MWh and a green certificate price of €110/MWh.

APPENDIX C: input data sand extraction simulations

Input parameters	
a_0: starting value intercept inverse demand function, choke-off price	12
c^v: marginal cost virgin production	3
c^s: marginal cost substitute production	50 → 10
t^q: tax on consumption	0
t^v: tax on virgin production	0 → 2
t^s: tax on substitute production	0
p^d: full disposal price recycling residues	5
S_0: Initial resource stock at time zero	52,324,000
IUS_0: Initial in use stock at time zero	0
LF_0: Landfilled waste volume at time zero	0
δ: private discount rate	0.03 → 0.10
Sdr: social discount rate	0.03
γ_0: starting value recycling rate	0.30
e: externalities	0
Calibrating parameters	
b: Absolute value slope of inverse demand function	6/1,724,000
g_0: starting value marginal recycling cost function parameter	6/log(1-0.30)

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