Municipal Solid Waste Management Problems: An Applied General Equilibrium Analysis

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Abstract

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About 40% of the entire budget spent on environmental problems in the Netherlands is reserved for the waste management problem. Regardless of the amount spent on waste management, the quantity of municipal solid waste generated still increases. It has up till now proven impossible to decouple generation of municipal solid waste and income growth.

This thesis investigates the policy options that can be used to reduce generation of municipal solid waste and looks specifically at the direct and indirect effects of introducing unit-based pricing. Two types of unit-based pricing are distinguished: a full unit-based pricing scheme, in which municipalities charge a variable price for collection of both organic waste and rest waste, and a selective unit-based pricing scheme, in which municipalities only charge a unit-based price for the collection of rest waste. It presents a modeling framework to simulate the waste market in the Netherlands. The model includes several municipalities as sources of waste, consumer preferences, economies of scale, transport costs, and several kinds of emissions caused by waste treatment. In this thesis specific focus was given to the possibility of waste leakage, where consumers pollute the organic waste stream with rest waste.

The model was used in a stylized example with numerical data based on the Netherlands in 2000. The results show that the selective unit-based pricing scheme is the most effective policy tool to reduce generation of municipal solid waste. Due to the effects of waste leakage, however, it is not advisable to introduce unit-based pricing in every municipality. The results show that it is not cost effective to introduce selective unit-based pricing for waste collection in larger municipalities. In these municipalities the effects of waste leakage are too costly. The degree of pollution is so high that part of the organic waste stream cannot be composted and will have to be incinerated, thus greatly increasing the costs of treating organic waste. Only in small municipalities with a relatively large number of environmentally concerned consumers selective unit-based pricing can be introduced. Larger municipalities may consider introducing full unit-based pricing. This policy tool, however, only stimulates prevention and not recycling, thus the effects for reducing generation of rest waste are limited.

Voorwoord

Op het schrijven van een proefschrift zijn tal van zinspreuken van toepassing. Spreuken zoals 'Aken en Keulen zijn niet op een dag gebouwd', 'de laatste loodjes wegen het zwaarst' en 'de aanhouder wint', waren zeker van toepassing op mijn proefschrift. Toch vind ik de stelling van Ronday nog het meest toepasselijk: 'Promoveren is vaak een weg naar niets en het gaan naar nergens totdat je het bereikt hebt'. Na vijf jaar heb ook ik mijn doel bereikt en ligt het proefschrift hier in gebonden vorm. Hoewel de weg zeker niet zonder hobbels is geweest en ik me af en toe wanhopig afvroeg of het ooit wel wat zou worden, kan ik toch met voldoening en plezier terugkijken op de afgelopen jaren en kan ik me nu vol overgave op mijn nieuwe werk bij APE storten. Natuurlijk zou het me niet gelukt zijn zonder de hulp van anderen, die ik dan ook in dit voorwoord wil bedanken.

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Den Haag, oktober 2003

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Part I

Concepts and background

" Waste itself is a human concept; everything in nature is eventually used. If human beings carry on in their present ways, they will one day be recycled along with the dinosaurs." (Peter Marshall)

1 General introduction

1.1 Definition and classification

The majority of human activities will inevitably result in the generation of waste due to the imperfect utilization of energy and resources. There are numerous definitions of what exactly constitutes waste, and many classifications, which attempt to categorize waste flows. According to the European Environmental Protection Act (1990), "waste is any substance, which constitutes scrap material or any effluent or other unwanted surplus substance arising from the application of a process, or any substance or article, which requires to be disposed of as being broken, worn out, contaminated or otherwise spoiled."

Waste poses a highly complex and heterogeneous environmental problem. The characteristics of waste are highly dependent on the materials of which it consists. For example, the characteristics of nuclear waste and organic waste are very different, both with respect to their natural absorption capacity and impact on human health. Yet they have one thing in common: both waste types are by-products of human activity and although they physically contain the same materials as found in useful products, they differ from useful products due to their lack of value (White *et al.*, 1997).

The existence, and more specifically, the treatment of waste can cause environmental damage as well as health risks. Different categories of waste cause different problems. For example, the health risks associated with toxic waste are much greater than those relating to municipal solid waste. Depending on the type of waste that must be handled, different legal regulations may be necessary to control the environmental and economic effects of waste treatment.

Waste may be categorized with respect to the source that generated it (WMC, 2003d). Waste types distinguished according to this classification are: (1) municipal solid waste, which is generated by households and contains the so-called 'rest waste', as well as organic waste, glass, paper and other recyclable materials (2) residual waste that is generated by waste treatment facilities like composting units and incineration plants, (3) industrial waste, which is generated by industrial sectors (4) construction waste, which is generated by the construction and demolition sectors, (5) contaminated soil and (6) other waste, which is a diverse set of smaller types of waste categories including, for example, waste originating from hospitals and non-contaminated soil.

Other classifications, for example, based on composition of waste rather than its origin, also exist. Such classifications regard toxic waste and organic waste as separate categories. However, according to the above classification, toxic waste may be included in every category: from *municipal solid waste* to *other waste*; organic waste is part of both the category *municipal solid waste* and *industrial waste*.

In general, one can argue that there are five main categories of socially acceptable waste handling options available, namely (1) prevention, (2) re-use and recycling, (3) composting, (4) incineration and (5) landfilling. Naturally not every waste handling option is suitable for every category of waste. Each waste handling option has its own economic and environmental characteristics.

Waste prevention or minimization is usually the most favored waste handling option, but may be difficult to achieve in our consumer society. *Re-use and recycling of waste* have clear environmental advantages. By re-using and recycling materials, less virgin materials need to be used, ultimately resulting in a closed production cycle in which no or at least very few virgin materials are actually required. The economic costs of re-use and recycling, however, are substantial, and there may be technical problems preventing re-use and recycling on a large-scale. Moreover, it should be noted that even recycling and re-use might cause environmental damage.

The first two categories are typical examples of ways of reducing waste flows. The next three categories are examples of treating waste in order to get rid of it. *Composting* organic waste is one of the most favored methods of waste treatment. By transforming organic waste into compost, at least part of it can still be usefully employed. In the Netherlands, the *incineration of waste* is the preferred way of treating non-organic waste. Energy can be obtained through incinerating waste. Incineration provides a major contribution to reaching the targets set by the European government for the use of energy from renewable resources. *Landfilling of waste*, which was predominant up until a decade ago, is the least preferred option for waste treatment. Although it is relatively cheap, it also leads to relatively high environmental risks due to emissions into the air and groundwater. In the Netherlands, landfilling sites are legally required to provide permanent aftercare to reduce the possibility of future spills.

The category hazardous waste deserves some special attention. According to the laws of both the European Union and the United States, hazardous waste must be handled more carefully than common municipal solid waste. Hazardous waste can either be a liquid, solid or sludge that is a by-product of a manufacturing process. It can also be the result of commercial products, such as battery acid or industrial solvents, which have been discarded. The treatment of this waste type can have serious environmental effects. In the United States, hazardous waste may be landfilled but only in specially designed and extra secure landfill sites. Since 2002, it is no longer possible to landfill hazardous waste in the European Union; it must either be incinerated or treated in another way. Following several scandals involving the dumping of hazardous waste in developing countries, both the European Union and the United States have adopted laws forbidding the export of hazardous waste.

1.2 The waste management problem

The increasing scale of economic activity, *i.e.* industrialization, urbanization, rising standards of living and population growth, has led to a sharp increase in the quantity of waste generated. The environment has a limited capacity for waste assimilation. If too much waste enters the environment rather than being recycled or reused, the assimilative capacity of the environment is put under too much stress to be able to handle the total quantity of waste generated. This may result in pollution and resource degradation and consequently economic damage (Turner, 1995).

According to the mass balance principle, which can be derived from the first law of thermodynamics[1], mass inputs must equal mass outputs for any process. This implies that any virgin materials used in both the production and consumption process must eventually be returned to the environment as higher entropy waste products or pollutants (Ayres, 1989). It is not yet possible to achieve an one hundred percent recycling rate. A society is, however, to some extent able to choose the quantity and quality of waste it will generate.

Waste can be treated in several ways. It can be composted, incinerated, or landfilled. Until a decade ago, landfilling of waste was very popular in the Netherlands. Landfilling, however, is also the least environmentally friendly waste treatment option. The government has, therefore, implemented several laws to render landfilling less attractive. One of the most successful policy measures was the introduction of a high landfilling tax. Due to this landfilling tax, landfilling became very expensive. The price of landfilling combustible waste is actually higher than the cost of incinerating it. This price incentive stimulated the industrial sectors to reduce waste generation. Over the last 10 years, the overall recycling percentages in the industrial

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[1] The first law of thermodynamics, the law of conservation of mass/energy, states that physical processes always require conservation of energy/mass. In other words, energy and matter cannot be created or destroyed (Perman *et al.*, 1996).

sector increased from about 70% to almost 90%. Households recycle far less, only about 40%. The government still faces a difficult task in trying to solve the municipal solid waste problem.

The municipal solid waste flow accounts for about 40% of all waste that requires treatment. This waste category presents perhaps the greatest waste management problem in the Netherlands. By nature, municipal solid waste is one of the most difficult sources of waste to manage due to its complex composition and diverse sources of generation (Read, 1999). Since every household in the Netherlands generates municipal solid waste, it is difficult to control this waste flow. To re-use, recycle or compost waste, the government is dependent on the households. If a household chooses to not recycle or separate waste, there is essentially nothing the government can do, since it is far too expensive to check the quality and quantity of waste recycled or composted in every household. Any attempt to reduce the municipal solid waste flow by increasing the price of collection, usually results in some form of illegal dumping. Consumers can, for example, dump waste in their neighbor's bin, take it to work with them, or dump it in a nearby field or forest. Households can also illegally dispose of rest waste by dumping it in the organic or recyclable waste stream. By polluting these waste streams they increase the costs of recycling and composting significantly. The quantity of waste illegally disposed of differs a lot between municipalities. Depending on the environmental preferences of the households, some municipalities will have more significant problems with illegal disposal than others.

When designing an efficient waste management plan, it is important to consider the interactions between the waste treatment sector, on the one hand, and the rest of the economy on the other. Waste management policies aimed at reducing waste generation at the production side ignore the behavior of the households such as the choice of waste reduction and disposal decisions. The effects of the policy may therefore be less beneficial than expected. Subsequently, policies designed to reduce waste generation by private households can lead to households demanding products with less waste content, thus influencing the producer decisions, but may also lead to increased illegal disposal by private households.

Waste treatment costs are dependent on how and where the waste is treated. Due to economies of scale, a smaller waste treatment unit is more expensive than a large one. The quantity and quality of waste to be treated will have a significant impact on the optimal location choice of waste treatment units. An efficient waste management plan should take these spatial aspects into account. Each municipality should decide on the basis of the quality and quantity of waste they collect, where and how to treat the waste.

In short, a satisfactory analysis of municipal solid waste policies demands a comprehensive framework in which production, consumption, disposal stages, and spatial aspects are included. In this thesis, such an analysis is presented. Using a general equilibrium model of the waste market, I will demonstrate the effectiveness of several waste management policies. My analysis will include the effects of consumer preferences, recycling, prevention, economies of scale of waste treatment units, transport costs and both quality and quantity of municipal solid waste.

1.3 Waste generation, market distortions and incentives

Following the Second World War, the generation of waste has increased rapidly in the Netherlands. Since 1950, the quantity of waste generated has more than tripled, from about 17 Mtonnes in 1950 to about 67 Mtonnes in 2000 (WMC, 2003e). During the sixties and seventies in particular there was a sharp increase in national income, which resulted in a substantial rise in waste generation. The European Environment Agency (EAA, 2000) has demonstrated that waste generation in the European union is still coupled with economic growth, making it impossible to pursue economic growth without creating increasingly serious waste management problems. A particularly close link exists between economic growth and the waste generated by the construction industry, as well as between economic growth and municipal solid waste. The generation of other types of waste, such as industrial and agricultural waste, is still on the increase, but the quantity of these types of waste grows more slowly than the annual rise in welfare due to successful implementation of waste management policies (Dijkgraaf *et al*., 1999).

In the Netherlands, the government managed to decouple economic growth and the generation of both industrial and construction waste. The generation of municipal solid waste, however, is still clearly coupled with economic growth. The government has failed to achieve its targets in this respect. This failure should be attributed primarily to the presence of market distortions in the waste sector. Three important factors have led to these market distortions, namely: (*i*) a flat fee-pricing system (*ii*) virgin material biased regulations and (*iii*) the so-called ‘killer-contracts’.

The flat fee-pricing system generates the first market distortion. In a flat fee-pricing scheme, the private households pay a fixed amount of money per year for the collection of municipal solid waste. The total amount of the fee charged is not dependent on the actual quantity of waste generated. Most municipalities choose this kind of pricing system because it is quite expensive to keep track of the actual quantity of waste generated per household. The most important problem created by this pricing system is a missing link between waste generation and the price of collection. Private households therefore have no price incentive to reduce the quantity of waste they generate.

Virgin material biased policies lead to the second market distortion. Virgin-material biased policies inadvertently promote the use of virgin materials instead of recycled materials. Miedema (1983) shows that because the price of waste collection and disposal is not incorporated into the price of virgin materials, virgin materials are too cheap in comparison to recycled materials. As long as the costs of waste disposal are not internalized in the price of virgin materials, the demand for virgin materials will be higher than socially optimal.

The third market distortion is one specific to the Netherlands. In the Netherlands, so-called *killer-contracts between municipalities and waste treatment facilities* exist. The killer-contracts between municipalities and *incinerators* have often been the focus of discussion. However, to a lesser extent, killer-contracts also exist between municipalities and *composting units*. These contracts specify the quantity of waste that the municipality will deliver to the facility and the price they will pay for disposing of it. These contracts provide the municipalities with an incentive to keep the quantity of municipal solid waste generated by the private households constant so that they can fulfill their contracts (see also De Jong and Wolsink, 1997).

Several studies have already analyzed the effects of market distortions in the municipal solid waste market. An extensive overview of the current literature can be found in Chapter 2. Most of these studies have concentrated on solving the problems caused by the flat fee-pricing system. By replacing the flat fee-pricing system with a unit-based pricing system, it is in theory possible to negate the market distortion. In a unit-based pricing system, households pay a variable fee to the municipalities for the collection of municipal solid waste; the fee charged will in some way depend on the actual quantity of waste generated. Several differentiating pricing systems are possible: for example a weight-based pricing system, which bases its price of collection on the total weight of waste collected; a frequency-based pricing system, which bases the price of collection on the frequency it is collected and a volume-based pricing system, which bases its price on the volume of waste collected. In the following paragraph, a brief overview is given of the most important articles in the field of waste management and waste policies.

Wertz (1976) was the first to analyze the effects of a user charge on municipal solid waste disposal. He found that there was a distinctive negative relation between the price of municipal solid waste disposal and the actual quantity of municipal solid waste generated.

Miedema (1983) analyzed the effects of other distorting characteristics of the municipal solid waste market, such as virgin material-biased tax policies, virgin material-biased policies, and indirect subsidization of virgin materials. He advocated the introduction of virgin material taxes as a means of motivating efficient waste disposal practice.

Jenkins (1993) developed a model where households maximize utility, which positively depends on the consumption of goods and negatively on the quantity of recycling. A disposal charge for municipal solid waste collection is included in the budget constraint. She found that the quantity of municipal solid waste generated is sensitive to the price of municipal solid waste collection. In particular, she found that the average price elasticity for municipal solid waste collection equaled –0.12.

Hong *et al.* (1993) derived a household recycling choice model and a demand function for municipal solid waste disposal. They applied the model to a sample of households from the Portland, Oregon metropolitan area and found a positive though small relation between an increased price of waste collection and the quantity of municipal solid waste generated.

Miranda *et al.* (1994) analyzed the effects of introducing a unit-based price on waste disposal behavior. They collected data from 21 cities throughout the United States over an 18-month period. They ascertained that introducing unit pricing and recycling-programs could have a dramatic effect on the quantity of municipal solid waste generated.

Sterner and Bartelings (1999) found that the introduction of an unit-based pricing system for the collection of municipal solid waste combined with the launch of a ‘green’ shopping campaign and the introduction of recycling centers had a dramatic effect on the quantity of municipal solid waste generated. This study focused on the attitudinal variables that influenced the quantity of municipal solid waste generated by households, and discovered that economic incentives, although important, are not the only driving force behind the observed reduction of municipal waste. Given a proper recycling structure, households are willing to invest more time in recycling and composting than can be purely motivated by savings on their waste management bill.

Each of these empirical studies concludes that waste generation is sensitive to user fees. The introduction of user fees can lead to a substantial reduction in municipal solid waste generation, especially if they are combined with programs that increase the public awareness about the municipal solid waste problem. The imprudent construction of waste collection fees, however, might not have the desired effect and can encourage illegal dumping, burning or other improper kinds of disposal (Fullerton and Kinnaman, 1995).

Although most of these studies agree that a flat fee-pricing system is not optimal, they differ on what the optimal policy to minimize cost of disposal should be. Studies like Miedema (1983), Jenkins (1993), Strathman *et al.* (1995), and Linderhof *et al.* (2001) propose the introduction of a ‘downstream’ tax, for example a unit-based pricing system.

Other studies, such as Fullerton and Kinnaman (1995,1996); Palmer and Walls (1997); Fullerton and Wu (1998) and Choe and Fraser (1999), favor an ‘upstream’ tax, like a deposit refund system or an advanced disposal fee on price of the consumption good, to internalize the waste treatment costs in the price of the product. In a deposit-refund system, consumers pay an extra amount of money (the deposit) to the seller. If the consumers return the remainder of the product to the seller, they will get the deposit back. The recyclable waste that is thus collected is then sent to either a re-use center or a recycling unit. They fear that a ‘downstream’ tax will be non-optimal due to huge implementation and enforcement costs.

1.4 Objectives of the study

Recent literature, as described in Section 1.3, has provided some insights into the kind of effects that market distortions can have on the municipal solid waste market. These studies demonstrated how the introduction of a unit-based price, recycling subsidies and taxes influenced both the quantity of municipal solid waste generated and the total costs spent on waste treatment. These studies, however, have neglected several important aspects of the waste management problem.

First of all, they have not fully considered the impact of the environmental preferences of private households on the quantity and quality of waste they generate. In this thesis, I will study how different types of consumers react to the introduction of unit-based pricing for waste collection and how their preferences determine the quality of waste they generate. Furthermore, I will show how these results may influence the design of waste management plans.

Secondly, although some of these studies identified the illegal dumping of waste as a household strategy for waste reduction, they did not consider an alternative method of illegal disposal, namely the dumping of rest waste in the organic or recyclable waste stream. This has important consequences for the treatment of organic and recyclable waste, and in this thesis I will illustrate how this behavior can be included in the analysis.

Thirdly, these studies did not cover the spatial aspects of the waste management problem in the context of a general equilibrium analysis. Deciding where waste is to be treated is an important aspect of the waste management problem and this decision is influenced by both the quantity and the quality of waste that is generated. In this thesis, a fixed set of waste management locations, several sizes of waste treatment units, economies of scale, and transport costs are included in a general equilibrium framework for the waste market.

In this thesis, I aim to contribute to the understanding of waste management in the following ways:

- *By providing an analysis of how the incentive structure of the consumers, emission restrictions, interrelations between the municipal solid waste sector and the rest of the economy and the spatial aspects of the waste problem influence the optimal municipal solid waste management plan.*

- *To assess whether a flat fee-pricing system, a unit-based pricing system for the collection of rest waste, a unit-based pricing system for the collection of organic and rest waste, or a recycling subsidy is the preferable policy option to minimize the social costs of municipal solid waste treatment.*

- *To gain insight into how to develop a more efficient municipal solid waste management plan, which solves inefficiencies caused by market distortions present in the municipal solid waste market.*

The objectives of this thesis lead to five key research questions:

1) What are the most important environmental and economic topics with regard to the municipal solid waste management problem?

2) How does the market distortion caused by the flat fee-pricing system influence municipal solid waste generation and how can these negative effects be sufficiently reduced?

3) How great a problem is waste leakage and how is waste leakage influenced by household attitudes?

4) How is the choice of the optimal location of waste treatment facilities influenced by the quantity and quality of municipal solid waste generated by consumers and, moreover, how will the spatial aspects of the municipal solid waste management problem in turn influence the successfulness of introducing unit-based pricing?

5) What kinds of policy changes can be recommended to minimize the total social costs of municipal solid waste treatment for our society?

The first research question deals with the focus of the research project. On the basis of a literature research, I will provide a detailed illustration of the municipal solid waste management problem and outline the kind of environmental and economic issues that are involved in it.

The second research question focuses specifically on one market distortion in the municipal solid waste market, namely flat fee-pricing. As mentioned earlier, the flat fee-pricing system can cause inefficiently high quantities of municipal solid waste to be generated. This thesis will pay special attention to the effects of the flat fee-pricing system and policy alternatives.

The third research question deserves some introductory comments. The choice between waste treatment options does not solely depend on the preferences of the municipalities who collect municipal solid waste, but also on the kind of waste that is generated. Not all waste is suitable for incineration or composting. For example, municipal solid waste consists of several categories of waste, namely glass, paper, hazardous waste, organic waste, and rest waste. The category rest waste is quite diverse and consists of several different types of materials like plastics, aluminum, but also glass, paper and organic waste. Glass and paper can be recycled, hazardous waste must be incinerated or treated otherwise, and organic waste may be composted. Rest waste will be incinerated. The recyclable and organic waste streams, however, should not be polluted with rest waste. Dumping rest waste in the recyclable and organic waste stream, which will subsequently be referred to as "waste leakage", means that it will be far more costly to treat this waste, for the rest waste has to be separated from the other waste types.

The *fourth research question* concerns the interaction between the quality and quantity of municipal solid waste and the choice of waste treatment units. To minimize the cost of waste treatment, it is possible to concentrate only on minimizing the quantity of municipal solid waste that is generated. In this case, the treatment of waste is left out of the equation. Another method is to concentrate solely on how and where municipal solid waste should be treated. Both of these methods, however, do not consider the interactions between the quantity and the quality of waste generated and the optimal waste treatment method. For example, a small quantity of organic waste of a good quality could well be treated in a small composting unit. A large quantity of waste of a lower quality may only be treatable in a larger composting unit. As the quantity and quality of waste generated is not fixed, but may be influenced by policies, it is important to take this interaction into account.

The choice for the optimal waste treatment location strongly depends on the characteristics of the municipality concerned, the distance, the economies of scale, and the environmental characteristics. Depending on both the quality and the quantity of waste collected, municipalities may prefer either a smaller or a larger waste treatment unit. Since municipalities are very diverse in size and nature, it is difficult to design an optimal waste management plan that is suitable for every municipality. The optimal municipal solid waste management plan must reflect the preferences of both the municipalities and its inhabitants. Some municipalities may wish to charge consumers for the quantity of waste they generate because of the 'polluter pay principle', which says that every polluter should be charged for the environmental costs they cause. Other municipalities may choose a flat fee due to the 'equality principle', as poorer households will, in relative terms, pay more than more affluent households when a unit-based pricing system is implemented. It will, therefore, be impossible to design an optimal national waste management plan without taking into account the individual characteristics of the municipalities in question.

Finally *the fifth research question* concerns policy recommendations based on this thesis. I will specifically illustrate the kind of situations in which it is advisable for a municipality to introduce a unit-based pricing system for municipal solid waste collection.

The focus of this thesis is to provide insight into the interrelations between the waste sector, consumer behavior and the rest of the economy. The applied general equilibrium technique will be used as a modeling technique. In particular, the Negishi format is employed as the preferred modeling technique (see Section 1.5). To answer the research questions, I will need to answer the following modeling questions:

a) How can interactions between the waste sector, government policies, and the rest of the economy be modeled?

b) How can the flat fee-pricing system be introduced to a general equilibrium setting?

c) How can spatial aspects of the waste management problem, such as a fixed set of possible location of waste treatment units, economies of scale and transport costs, be introduced to a general equilibrium framework?

This thesis is part of the research program *Material Use and Spatial Scales in Industrial Metabolism* (MUSSIM), funded by The Netherlands Organization for Scientific Research (NWO), which aims to develop an economic framework for modeling the physical side of the economy in economic models. The research program seeks to develop a framework and method of analysis that is based on dynamic optimization and simulation. Furthermore, the program integrates economic processes and decisions on the use of materials (environmental and resource economics), physical flows and processes related to use of these materials (industrial metabolism), and decisions on spatial allocation and transport affecting these materials flows (regional and international economics).

The MUSSIM-research program is divided into three research projects. Each research project examines a different aspect of material use in the economy. This thesis will thus focus only on municipal solid waste streams in the Netherlands. I will, therefore, disregard any possibilities of export of either waste or secondary materials. For further information on the economic, environmental and social costs and benefits of international trade in secondary materials at different spatial scales, see van Beukering (2001). For more details on the relationship between material flows and economic and spatial structure of production in the Netherlands for selected materials and extensive input-output models for the Dutch economy, see Hoekstra (2003).

1.5 Conceptual framework

The main modeling tool used in this thesis is the applied general equilibrium modeling technique. I have chosen the general equilibrium setting because I would like to analyze the main interactions between economic behavior, waste generation, and resource use. The possibility of analyzing the interactions between several markets at once is the strength of general equilibrium modeling. By choosing a general equilibrium format it is possible to study the effects that a policy change concerning municipal solid waste has on the waste treatment sector, the recycling sector, the production sector and the virgin material sector.

Shoven and Whalley (1993) provide an excellent description of the main aspects of a general equilibrium model:

"The term general equilibrium corresponds with the well-known Arrow-Debreu model (see Arrow and Hahn, 1971). The number of consumers in the model is specified. Each consumer has an initial endowment of N commodities and a set of preferences, resulting in demand functions for each commodity. Market demands are the sum of each consumer's demands. Commodity market demands depend on all prices, and are continuous, nonnegative, homogeneous of degree zero (*i.e.* no money illusion), and satisfy Walras' law (*i.e.* that at any set of prices, the total value of consumer expenditures equals consumer incomes). On the production side, technology is described by either constant-returns-to scale activities or non-increasing-returns-to-scale production functions. Producers maximize profits. The zero homogeneity of demand functions and the linear homogeneity of the profits in prices (*i.e.* doubling all prices doubles money profits) imply that only relative prices are of any significance in such a model. The absolute price level has no impact on the equilibrium solution Equilibrium in this model is characterized by a set of prices and levels of production in each industry such that that the market demand equals supply for all commodities (including disposals if any commodity is a free good). Since producers assumed to maximize profits, this implies that in the constant-returns-to-scale case, no activity (or cost-minimizing technique for production functions) does any better than break even at equilibrium prizes ".

Shoven and Whalley (1993) p. 1-2

General equilibrium models are economy-wide models in the sense that they cover all major economic transactions. The reason for modeling all relevant markets simultaneously is the existence of complex interactions in an economy. Partial models are based on the *ceteris paribus* conditions, *i.e.* the remainder of the economy is assumed to be constant during policy simulations. As long as the ceteris paribus condition holds, partial models are fine, and the complications and data-requirements of general equilibrium models can be safely avoided. If, however, there are significant linkages between different markets, a partial analysis may lead to inaccurate and perhaps biased results due to the existence of indirect effects[2]. In an extreme case, the indirect effects, as captured by general equilibrium models, may outweigh the direct effects, as captured by partial models. This can result in opposite policy recommendation (Thissen, 1998).

General equilibrium models can be built in different formats, such as the Computable General Equilibrium (CGE) format, the Negishi format, the full format, and the open economy format. Each of these formats has its strengths and weaknesses, for more information see Ginsburgh and Keyzer (1997). The models presented in this thesis are all written in the Negishi format. I have chosen this format, as it is especially suitable for the implementation of externalities, such as environmental pollution and waste generation; and price rigidities, like a zero marginal price for waste collection. In contrast to, for example, the CGE format, the Negishi format is able to calculate the equilibrium solution in the case of price rigidities without requiring additional proof that a general equilibrium solution has been found. Moreover, the Negishi format is particularly suitable for incorporating multiple consumers given that it can maximize several utility functions at the same time.

The Negishi format can, however, only be written in the primal form, which is a weakness of this type of modeling. This means that in the model only production sets exist. Prices are calculated exogenously from the model. In the primal format the equilibrium solution if found by one or several mathematical programs using some iterative procedure on parameters to find a fixed-point solution. In the dual form, which for example is used in the computable general equilibrium format, net supply and input demand are explicit functions of prices. The model is solved by a system of nonlinear equations. The advantage of the dual form over the primal form lies in the way in which the model is solved. As it is based on a system of nonlinear equations, the computation and parameter estimation are normally far less difficult than the computation in the primal form. Thus the dual form will find an equilibrium solution much faster than the primal form. Nevertheless, I feel that this disadvantage does not offset the strong points of the Negishi format.

In this thesis, the general equilibrium framework is used to analyze the interactions between the waste treatment sector, the consumption sector, the production sector, the recycling sector, and the extraction sector. The main elements of the conceptual framework are shown in Figure 1-1. Several production sectors are distinguished. Each of these production sectors uses virgin materials and recycled materials to produce goods. These goods are consumed by the consumption sector. The

[2] The indirect effects capture the interactions between different markets. Any change in one market can result in a change within another, which in turn can again affect a change in the original market.

consumption sector consists of several types of private households and a government consumer. The consumption of products results in waste. Waste can be either recycled or treated. If waste is recycled it is transformed into recycled material, which can be used in the production process. Three methods of waste treatment are distinguished. Waste can be composted, incinerated or landfilled. Each waste treatment option will have its own costs and benefits. All waste treatment options create emissions but, for example, composting will cause far less environmentally damage than incineration or landfilling.

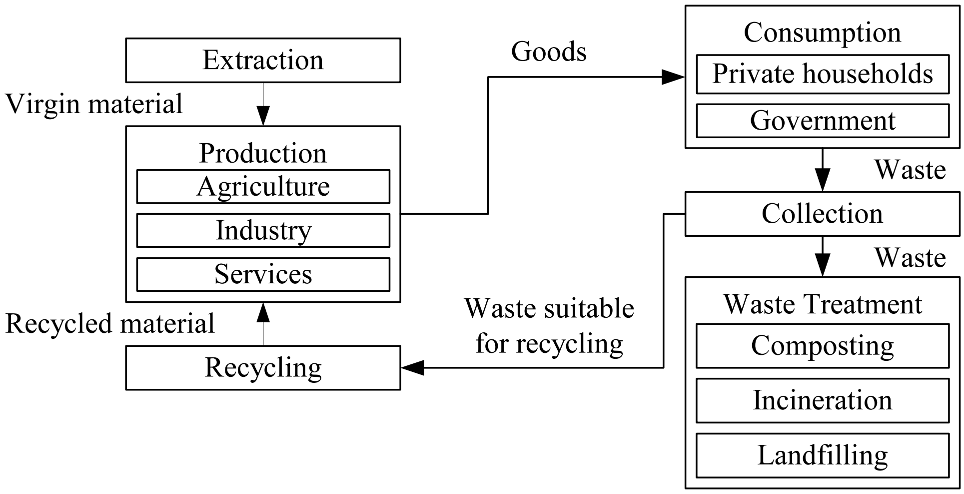

Figure 1-1 The main elements of the conceptual framework

The models presented in this thesis are comparatively static in nature. This means that I will compare a benchmark case with several scenarios. By introducing a policy change in a scenario and comparing the optimal outcome with the benchmark case, it is possible to analyze the expected changes in the economy due to the policy change. The focus of this thesis is on comparing one equilibrium state of the economy with another. How such an equilibrium state is reached after the introduction of a policy change is of less importance; the complications of designing a dynamic equilibrium model can, therefore, be avoided.

In this thesis, I have developed three types of models to analyze the waste management problem. Each of these models focuses on a slightly different aspect of the waste management problem. Each will be applied in a stylized example with numerical data used from the Netherlands. The results of these calculations will be carefully discussed and they will show the main workings of each model. The order in which I present these models reflects a logical development from a rather basic to a more complex level.

The first model is a basic applied general equilibrium model that focuses on the entire life cycle of a product. The model is fairly aggregated to prevent over-complication. All of the different stages that a product goes through, from extraction as virgin material, to production, consumption, recycling and final disposal by landfilling, incineration or composting are included in the model. By including the entire lifecycle of the product, it is possible to analyze how changes in generation of municipal solid waste can affect the use of virgin and recycled materials, consumption patterns and the choice of final waste disposal options.

The second model is more focused on the consumption sector and details of the waste collection sector. Since the focus of the model is slightly less broad than the previous model more detailed information about the different waste streams generated by households and household preferences are included. In this model, the production sectors are aggregated to one sector. Thus only one good is produced and consumed in the model.

Finally, like the second model, the third model focuses on the consumption sector and the waste treatment sector. In this model, detailed information about the *spatial aspects* of the waste treatment problem, *i.e.* where waste is generated and where it should be treated, are considered. Several municipalities and several locations of waste treatment facilities will be included in the model. This model provides insight into how changes in municipal solid waste generation influences the optimal location of waste treatment units and thus the transport cost caused by transport of waste. The analysis encompasses alternative settings for the locations of waste treatment units given a set of locations and sizes of waste treatment units, economies of scale and transport costs.

The models are all built in GAMS (General Algebraic Modeling System). This is an optimization program, which is - among other things - quite suitable for building complex general equilibrium models. The complete computer-code for each model is shown in appendix I.

1.6 Outline of the thesis

To gain insight into how to develop the most efficient municipal solid waste management plan, this thesis has been organized into seven chapters, starting with this introduction (*Chapter 1*). This section describes the main contents of the subsequent chapters in this thesis. Please note that the chapters have been written in such a way that they can be read and published independently. Some explanations and footnotes may thus necessarily be repeated in Chapters 4, 5 and 6. Table 1-1 gives a short overview of the characteristics and scope of each chapter.

Chapter 2 provides a general overview of the municipal solid waste management problem. Particular attention is paid to (1) several market distortions, which cause waste generation to be inefficiently high, (2) the choice between waste treatment options and the waste hierarchy, and (3) the spatial aspects of the waste management problem.

Table 1-1 Overview of the structure of the thesis

Chapter	Characteristic	Scope of the chapter
2	Conceptual	Descriptive analysis of the waste management problem
3	Conceptual / descriptive	Descriptive analysis of the waste market and current policies in the Netherlands
4	Modeling	Model for analysis of effectiveness of waste management policies
	Application	Analysis of effectiveness of introducing a unit-based price as compared to introducing recycling subsidies
5	Modeling	Model for analysis between waste quality and consumer preferences
	Application	Analysis of the effectiveness of introducing a unit-based price including that 'waste leakage' could occur
6	Model	Extension of the model of Chapter 5 to include location specific waste treatment centers, transport costs and several municipalities.
	Application	Analyzing spatial aspects of waste treatment problem in relation to size waste treatment units
7	Conclusions	Summary, conclusions and recommendations

Chapter 3 offers background information about waste flows and waste policies in the Netherlands. The Dutch waste market will be described in detail and insights into the financial and environmental costs of waste treatment, the generation of waste and the effects of waste management policies will be discussed. This chapter provides a detailed description of the economic and environmental costs of three different waste treatment options, namely *incineration*, *landfilling* and *composting*. Data presented in this chapter will be used in model applications in other chapters. Moreover, this chapter focuses on a description of the current waste management policies in the Netherlands and illustrates how these policies have developed over time.

Chapter 4 presents an analysis of the efficiency of Dutch waste policies. It focuses specifically on the problems associated with a flat fee-pricing system for collection of waste. In this chapter, a general equilibrium model is developed, which represents the municipal solid waste market. In the analysis, several important actors have been included, namely producers, consumers, municipalities, waste treatment units, and recycling units. The analysis focuses on how market distortions resulting from a flat fee-pricing scheme can be introduced to a general equilibrium format. The model is employed in a stylized example with numerical data from the Netherlands in 1996 used to demonstrate the effects of flat fee-pricing has on the generation of municipal solid waste. Introducing several policy options in the model and comparing the results to the benchmark case can help to find the most desirable policy option for the reduction of rest waste.

In *Chapter 5* a more detailed analysis of the interactions between the consumption sector and the waste treatment sector is presented. In this chapter, the focus is on the effectiveness of introducing a unit–based fee for the collection of municipal solid waste. Introducing such a fee may lead to a reduction in waste generation but it may also lead to an undesirable impact on the environment. Such a fee provides households with incentives to generate lower quality organic waste as a form of dumping. An applied general equilibrium model is presented that incorporates low quality organic waste, high quality organic waste, and rest waste, and includes the possibility of substitution between the generation of these three types of waste. The model is used to analyze the effectiveness of introducing a unit-based pricing scheme as compared to a flat fee-pricing system.

In *Chapter 6*, the model described in Chapter 5 is extended to include some important spatial aspects of the waste management problem, in particular the location of the waste treatment facilities in relation to transport costs and economies of scale. The model includes several municipalities. Each municipality has the choice of transporting their waste to a small, medium or large waste treatment facility. The model includes transport costs and economies of scale for different sizes of waste treatment facilities. The model also demonstrates that low quality waste can be expensive to treat, thus showing the direct disadvantages of waste leakage. This model is applied in a numerical example with data collected from the Randstad area in 2000. By extending the basic model of Chapter 5, a more extensive analysis can be given about the effectiveness of introducing a unit-based pricing scheme as compared to a flat fee-pricing system.

Chapter 7 contains the summary and main conclusions of this thesis. The five research questions will be answered in this chapter. Finally, policy recommendations and recommendations for future research are also given.

2 Economics of waste management: key problems

2.1 Introduction

The increasing scale of economic activity, *i.e.* industrialization, urbanization, rising living standards and population growth, has inevitably led to a sharp increase in the total quantity of waste generated in our society. This large and increasing mass of redundant goods, by-products, and organic and inorganic residue must be dealt with in one way or another. The environment has a certain capacity for assimilation of waste, but this capacity is not infinite. If too much waste enters the environment rather than being recycled or re-used, the assimilative capacity of the environment is put under too much stress and this results in pollution, resource degradation, and economic damage (Turner, 1995).

In the Netherlands, the quantity of waste generated increased sharply due to the rise in population growth and welfare throughout the last century. The quantity of municipal solid waste generated has been steadily increasing since the beginning of the 20th century. During the sixties and seventies, there was a sharp increase in income, which resulted in a substantial rise in waste generation. Since the eighties, a proportional relationship between the gross domestic product and the quantity of municipal solid waste generated has emerged. This is illustrated in Figure 2-1.

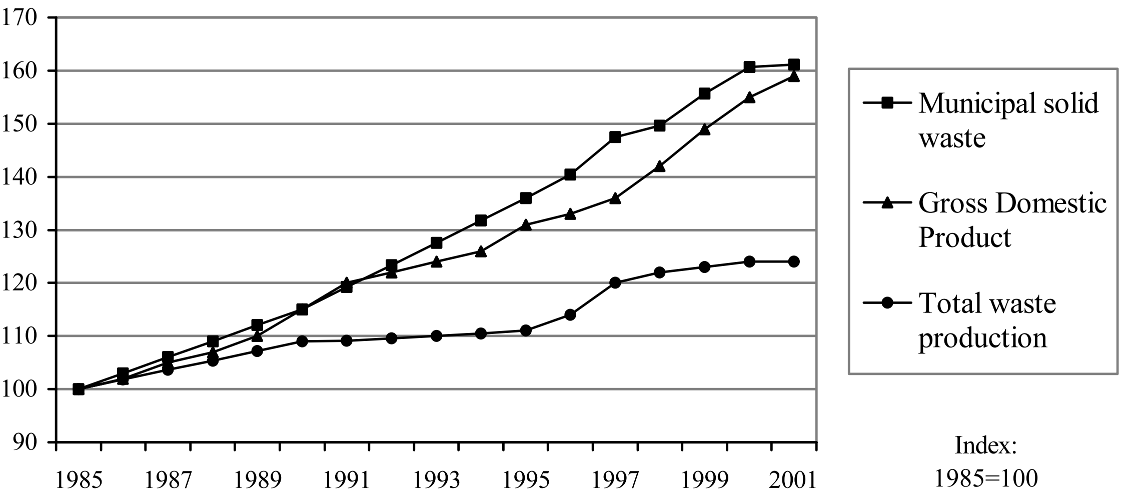

Figure 2-1 The development of the gross domestic product and the production of waste in the Netherlands, 1985-2001.

Figure 2-1 reveals a decoupling of the growth of the gross domestic product and the growth of total waste generation. This is mostly due to the steady increase of recycling in several production sectors. In the construction and demolition sector, for example, a recycling rate of 94% has been achieved. The growth rate of municipal solid waste generation is still linked to the growth rate of the gross domestic product. Although policy makers aimed to decouple income and waste generation, they have failed to achieve this for this particular waste stream.

Economic growth has led to an enormous increase in economic welfare. Material wealth has increased significantly and the quantity of goods available to the consumer has grown sharply. Due to the laws of thermodynamics, economic production and consumption always generate some pollution and waste. It is not possible to recycle for a full 100%. A society, however, can to some extent choose how much waste it generates through prevention, re-use, or recycling. By subsidizing recycling or by taxing landfilling, for example, the government can influence the quantity of waste generated. To design an efficient management plan, the government must balance the social benefits of a particular economic activity with the social costs (including disposal) related to this activity.

Waste treatment, such as composting, incineration, and landfilling, creates many problems for our society. It is costly to treat waste. For example, it leads to environmental problems and takes up valuable space. Available evidence shows that industrial countries are trying to cope with an increasing number of problems caused by disposal of waste. To build a waste treatment unit, a site has to be found that is technically suitable, *i.e.* the right soil and not too expensive, and socially acceptable. Some countries, such as the USA, Germany, and the Netherlands have a shortage (either locally or nationally) of sites that are technically suitable for building landfill or incineration units. This means that even if it is sociably acceptable to build a landfill or incineration unit, there simply is not enough space available to construct one. In other industrial countries there may be enough sites available, which are technically suitable for building an incineration or landfill site, but in these countries there is a shortage of possible landfill and incineration sites that are socially acceptable. The NIMBY (Not In My Back Yard)-syndrome plays an important role in the process of deciding on a possible disposal site (Turner, 1995).

National policy makers in the EU face an additional problem because the European Commission and Council have decreed that the 'proximity principle' should be an accepted part of all members states waste management policy. According to the proximity principle, '*provisions must be made to ensure that as far as possible waste is disposed of in the nearest suitable waste treatment centers*'. Thus, the export is not permitted, as it would place an unfair burden on the environment of the importing country (see for more information Monkhouse and Farmer, 2003). The proximity principle only applies to waste that must be incinerated or landfilled. Recyclable waste can be exported if adequate proof is given that the importing country is actually going to recycle the imported waste.

The municipal solid waste problem is still relatively new and policymakers are trying to cope with it in the best way possible. Considerable research has already been conducted on this topic. This chapter surveys the literature on the major questions and theories in the area of municipal solid waste management:

- How should the municipal solid waste market be regulated to reduce the generation of municipal solid waste?
- What is the optimal mix of waste treatment options?
- Where should waste disposal units be located, considering social, political, and economic preferences as well as pure technical aspects?

Section 2.2 discusses the optimal regulation of the municipal solid waste market and inefficiencies that are present in the current municipal solid waste market. Section 2.3 deals with the question of whether there is an optimal waste treatment method. Section 2.4 looks at ways of determining the optimal location of a waste disposal unit. Finally, Section 2.5 concludes the chapter.

2.2 Waste generation: the optimal policy mix

One of the most fundamental questions regarding the waste management problem concerns the ‘optimal’ quantity of waste that a society should generate. Most environmental scientists argue that we should not generate waste at all. Natural cycles like, for example, the hydrological cycle or the carbon cycle are closed, which means that waste generated during these cycles will be re-used as inputs. The industrial cycle should be fashioned after the natural cycle, thus we should try to close the material cycle and re-use or recycle all materials we consume. This idea of ‘treating the economy as a living organism’ is called industrial metabolism (Anderberg, 1998; Ayres and Simonis, 1994). Presently, our society is nowhere near to closing the industrial cycle. This cycle still extracts high-quality materials, such as fossil fuels and ores, from the earth and returns them to the earth in degraded forms; it only re-uses part of its waste.

From an economic point of view, it may not be necessary to fully close the material cycle. It is often forgotten that both recycling and re-use of materials have financial and environmental impacts, which makes it undesirable to completely eliminate waste generation (Pearce and Turner, 1993). Both environmental scientists and environmental economists, however, agree that too much waste is currently being generated (see for example Graig, 2001). The question remains just how much waste should be generated and how the waste market can be regulated to produce the 'optimal' quantity of waste. Fricker (2003) argues that the only sustainable way of reducing waste generation is by reducing consumption. The majority of environmental economists, however, do not share this view. In the next section, a number of policy instruments, which can be used to control waste generation, will be discussed.

2.2.1 Waste generation and the pricing mechanism

Waste management in most countries is still dominated by inefficient pricing, institutional and legal structures. The primary virtue of the pricing mechanism, *i.e.* the market, is that it gives consumers an idea of the costs of producing a particular product and offers producers insight into how consumers value a product (Löfgren, 1995). Naturally, the pricing mechanism only supplies the correct information if the market is undistorted. Solid waste management pricing is mostly based on a flat fee system. Households pay a fixed charge, the so-called flat fee, for the collection of waste. The amount of the fee is independent of the quantity of waste that is actually generated, thus consumers have no price-incentive to reduce the generation of waste, and thus larger quantities of waste are disposed of than is socially desirable.

Figure 2-2 illustrates the demand curve for waste collection services[1]. As the price of these services declines, the demand for these services increases. In the case of household waste disposal, the price of disposing one extra unit of waste equals zero, as the price is independent of the quantity of waste disposed of.

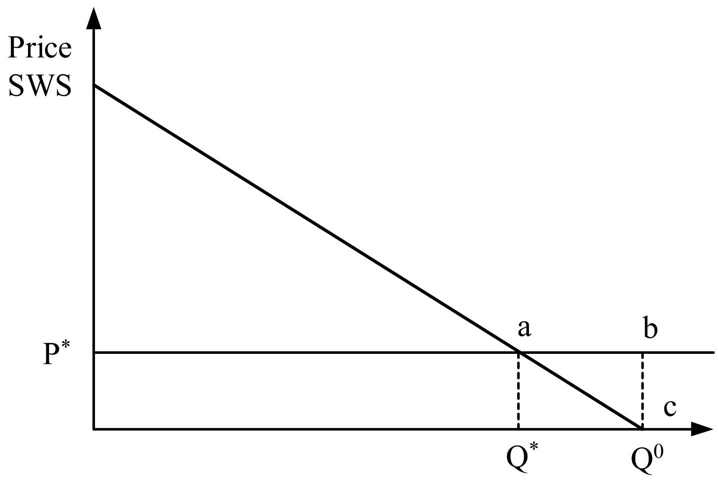

Figure 2-2 The demand curve for solid waste services (SWS)
Source: Jenkins (1993)

[1] The demand curve shown in Figure 2-2 is just an illustration of a possible demand curve. In reality, it may well be that the demand for solid waste services is not linearly related to the price of these services.

The quantity of waste disposal services demanded is equal to Q^0 and so consumption in terms of disposal costs is not restrained. If the price, *i.e.* the marginal costs of waste disposal, is equal to zero then Q^0 will be the optimal quantity of waste disposal. If, however, the marginal costs of waste disposal are positive, the demand for solid waste services is clearly higher than optimal. Assume, for example, that the social costs of waste disposal are equal to P^*, then the optimal demand for solid waste services will be equal to Q^*. Society faces a net total cost equal to the triangle *abc* caused by the inefficiently high demand for waste disposal services. Only when the disposal fee is equal to the exact marginal costs of waste disposal will the demand for waste disposal services equal the optimal quantity of waste disposal (Jenkins, 1993).

Finding the optimal disposal fee, however, poses several problems. The optimal disposal fee should cover both the marginal financial and the marginal environmental costs of municipal solid waste disposal and treatment. It is, therefore, important to quantify all external effects caused by waste treatment. However, as Figure 2-2 clearly demonstrates, the flat fee-pricing scheme will always lead to a non-optimal quantity of waste generation since the marginal costs of waste disposal are most assuredly positive.

It is important to note that the flat fee and the quantity of waste generated are not unrelated. The flat fee is determined by the quantity of waste generated in previous years. The flat fee will completely or partly cover the costs of collection and treatment of municipal solid waste. The flat fee, however, will not provide households with an incentive to reduce waste generation, as the marginal price of waste generation equals zero.

2.2.2 Finding the optimal policy mix

A lot of research has been done to determine the optimal policy mix to both stimulate consumers to generate less rest waste as well as to encourage more recycling and composting. The findings of these studies are discussed below using a simple general equilibrium model built by Kinnaman and Fullerton (1999).

In the model developed by Kinnaman and Fullerton, *n* identical households are distinguished. Each of these households maximizes utility (*u)* over consumption (*c)*. Consumption generates waste and this waste must either be disposed of as waste (*g)* or be recycled *(r).* The function *c(g,r)* represents all possible combinations of waste and recycling given a certain level of consumption. Consumer *i* maximizes utility given the price of consumption (p^c), the price of garbage disposal (p^g), the price received for recycled materials (p^r) and the available income (y).

$$Max\ u_i = u_i[c_i(g_i, r_i)] \qquad i = 1,...,n \tag{2.1}$$

Subject to the budget constraint:

$$y_i = p^c c_i(g_i, r_i) + p^g g_i - p^r r_i \tag{2.2}$$

According to this model, the production sector produces the consumption good with the input of virgin material (*v)* and recycled material (*r*). The production function *f* represents the production possibility set of the producer. He maximizes profits (π) given the prices p^v en p^r:

$$Max\ \pi = p^c f(v,r) - p^v v - p^r r \tag{2.3}$$

In the equilibrium solution, consumers will choose optimal levels of recycling and waste disposal. All recycled material is used by the production sector and the producers choose an optimal mix between using virgin material and recycled material.

In this simple model, the external effects created by waste disposal are disregarded, which is not a realistic assumption. Disposal leads to many environmental externalities, such as the pollution of ground water and emissions that contribute to the problem of climate change, acidification and other environmental problems. Assume that household utility is influenced by the total quantity of waste generated in society: $u_i = u_i(c,G)$ where $u_G < 0$ and $G=ng$. The solution found by the model described in equation 2.1 to 2.3 does not represent the optimal levels of recycling and disposal. For a positive *G*, *u(c,G)* will always be lower than *u(c)*. If consumers fail to internalize the social external costs of waste treatment in their utility function, the calculated levels of recycling will be too low and the level of waste disposal will be too high.

To internalize the external costs created by waste treatment in the price of waste disposal, economists have proposed the use of several taxation or subsidy schemes. To stimulate household recycling, the government may choose to tax waste disposal (at rate t^g), subsidize recycling efforts of households (at rate s^{hr}), or impose an advanced waste disposal fee at the time of purchase (at rate t^c). The maximization problem for the individual household is thus defined as:

$$Max\ u_i = u_i[c_i(g_i, r_i), G] \tag{2.4}$$

Subject to the budget constraint:

$$y_i = (p^c + t^c)c_i(g_i, r_i) + (p^g + t^g)g_i - (p^r + s^{hr})r_i \tag{2.5}$$

To directly stimulate the use of recycled materials the government can choose to tax the use of virgin materials (at rate t^v) or subsidize the use of recycled materials (at rate s^{fr}).

The profit maximization problem transfers into:

$$Max\ \pi = p^c f(v,r) - (p^v + t^v)v - (p^r + s^{f\ r})r \tag{2.6}$$

Levying a tax (t^g) on the generation of waste is the most direct approach to internalize the external costs of waste disposal. Most municipalities in the Netherlands and other countries throughout the world charge a flat fee for the collection of waste, either through local property or income taxes. This means that the marginal private costs of generating municipal solid waste (p^g+t^g) equal zero whereas the marginal social costs of generating municipal solid waste are positive. The introduction of a positive tax t^g can induce households to internalize the social costs of waste disposal in their decisions about generating waste.

Wertz (1976) was the first to estimate the effects of a unit-based price for municipal solid waste by comparing data from a municipality in the USA, which charged a user price for the collection of waste, with data from the rest of the USA that was representative of municipalities, which charged flat fees, for the year 1970. His results suggest that the introduction of user prices reduces the generation of municipal solid waste.

On the basis of panel data from 12 cities in the United States, Jenkins (1993) estimated that introducing a unit-based pricing could reduce the social costs of waste treatment by about \$125 per tonne of waste. This would improve social welfare by \$650 million a year, roughly \$3 per person per year. Fullerton and Kinnaman (1994) estimated similar welfare increases by analyzing the effects of the introduction of a volume-based pricing system in Charlottesville, Virginia. Podolsky and Spiegel (1998) calculated the largest social welfare increase. They estimated that the introduction of a unit-based pricing resulted in social welfare benefits of \$12.80 per person based on an analysis of cross-section data from a town in New Jersey (USA).

One advantage of the unit-based pricing system is that it is directly based on the ‘polluter pays principle’ as established in the framework directive on waste of the European Union, which, among other things, rules that the cost of waste disposal should be borne by the individual who generates it. The ‘polluter pays principle’ is generally accepted as instrument of justice given that it not only charges the polluter for the administrative and environmental costs generated by their behavior, but it also encourages the polluter to mend his ways (Perman *et al.*, 1996). Goddard (1995), however, raises an interesting question regarding the ‘polluter pays principle’ namely, who is the actual polluter in this case? Is it the consumer who generates the waste by consuming the product, is it the producer who designs a product that contains either too much waste or is not recyclable, is it the package designer, who pays little attention to the waste content of his design, or is it the retailer who desires packaging that keep transaction costs low? It is impossible to answer this question. Goddard demonstrates that it is more appropriate to consider which of the actors is in the best position to control the waste flow. A well-informed consumer would be the proper person to make personal consumption choices. By getting the prices of waste disposal right, the consumers can decide on their own how much municipal solid waste should be prevented or recycled.

Another advantage of the unit-based price is that it ensures an efficient allocation of resources without requiring the other tax and subsidy instruments mentioned above (Fullerton and Wu, 1997 and Palmer and Walls, 1994). If a unit-based price is introduced, households will start to consume, recycle, and dispose waste in such a way that the marginal benefits of consumption and recycling are equal to the marginal costs of disposal. In such a case, the market will provide the proper prices for consumer goods, recycling and disposal. For example, if consumers start to recycle more waste, recycled material becomes cheaper. Thus producers will start to use more recycled material without needing an extra incentive of the government in terms of a recycling subsidy (Kinnaman and Fullerton, 1999). In fact, Dinan (1993) shows that introducing both a unit-based price on waste disposal and a subsidy on the use of recycled material is inefficient as this basically subsidizes the use of recycled material twice.

Several studies, however, have illustrated that the introduction of a unit-base price will lead to significant transaction costs, thus it may be inefficient to introduce such a pricing system. First of all, the administrative costs of introducing a unit-based price may exceed the social benefits of lower waste generation. Fullerton and Kinnaman (1996) estimate that the administrative costs of introducing a unit-based price on the bases of an ‘expensive bag’ in Charlottesville, Virginia could exceed the $3 per person social benefits mentioned before. Linderhof *et al.* (2001), however, reveal that in Oostzaan the cost of waste collection and disposal did not increase after the introduction of a weight-based price for waste collection. Furthermore, they show that the costs invested in the introduction of the weight-based pricing system are compensated by the lower cost of waste treatment due to the reduction of waste. These results depend largely on the individual municipality. In the case of Linderhof *et al.*, the average consumer in the municipality was very environmentally friendly oriented. Thus, consumers were more than willing to recycle and prevent waste. It can be expected that results in other municipalities would be less positive.

Secondly, Dinan (1993) showed that a uniform unit-based price for all types of waste might be inefficient if materials within the waste stream led to different social costs. For example, the treatment of hazardous waste, such as flashlight batteries, will generate far greater social costs than the treatment of recyclable waste, such as old newspapers. The unit-based price collection of flashlight batteries should, therefore, be higher than the unit-based price for collection of old newspapers. Other studies, such as Walls and Palmer (2001), Eichner and Pethig (2001), and Calcott and Walls

(2002), support these results. A solution would be a selective unit-based pricing system based on the social costs of disposing the material in question, this would of course be rather expensive to implement.

Thirdly and most seriously, the unit-based pricing system may promote the illegal disposal of waste. Households may start to dump their waste in their neighbors' bins, dispose of it at work, illegally dump waste, or burn it themselves. Such behavior leads to large social costs and has been identified as one of the most serious obstacles to the introduction of a unit-based pricing for waste collection. Both Dobbs (1991) and Fullerton and Kinnaman (1995) demonstrate that if illegal disposal is a possibility, it may be optimal to have a negative tax on waste disposal, *i.e.* legal waste disposal should be subsidized. In such a case, policy makers would be better off implementing other policy instruments to reduce waste generation. Fullerton and Kinnaman (1996) estimate that about 28% of the decrease in waste generation may be caused by increased illegal disposal. Empirical studies, like Jenkins (1993) and Miranda and Aldy (1998), also report instances of increased illegal dumping. These results, however, are contradicted by other empirical studies. For example, Miranda *et al.* (1994), Strahman *et al.* (1995), Nestor and Podolsky (1998), Podolsky and Spiegel (1998), Sterner and Bartelings (1999) and Linderhof *et al.* (2001) found no significant evidence of increased illegal disposal.

Despite the three disadvantages mentioned above, the unit-based price is one of the most effective policy options to provide an incentive to increase prevention and home composting. None of the other policy tools can significantly influence the consumers' choice to prevent waste. Therefore, Calcott and Walls (2002) find that a modest disposal charge will always be part of the set of optimal policy instruments. Shinkuma (2003) even goes a little farther, arguing that even if illegal disposal is an option, the unit-based pricing system will still provide a second best optimum as long as the price of recycled material is positive. Only if the price of recycled material is negative, should another policy tool like the deposit-refund system be considered.

As mentioned above, the unit-based price alone may not provide an efficient solution to the municipal solid waste problem and it may be necessary to use other policy instruments. Miedema (1983) was one of the first to evaluate the potential use of other policy instruments. He found that the introduction of a tax on the use of virgin materials (t^v) provides more welfare gains than a subsidy on the use of recycled materials (s^{fr}), a unit-based price for waste collection (t^g) or an advantaged disposal fee internalized waste disposal costs in the price of a product (t^c). Although both the unit-based pricing scheme and the advantaged disposal fee can reduce waste generation and increase recycling, the necessary fee must be high, which can stimulate illegal disposal. Besides the transaction costs of introducing these systems and the social costs of disposal are too great to be cost-efficient. A tax on the use of virgin materials provides an incentive to use fewer virgin materials and at the same time boost the market for recycled materials. Miedema thus favors the virgin material tax above other policy instruments.

Pearce and Turner (1993) conclude that both virgin material taxes and unit-based pricing could indeed be used to correct market distortions in the waste market. A later study by Bruvoll (1998) supports the conclusion that the introduction of a virgin material tax will result in significantly higher use of recycled materials and lower quantities of waste generated thus considerably reducing social costs of waste disposal. Moreover, taxing virgin material may be an efficient tool for reducing air emissions, thus reducing social costs of using virgin materials.

Conrad (1999) demonstrates that a virgin material tax provides firms with a strong incentive to reduce waste generation; a much stronger incentive than the unit-based price provides. Not all studies agree with this result. For example, Dinan (1993) shows that only those producers who are able to substitute virgin material for recycled material will do so. This means that a large number of industries that have no real option to substitute virgin material will not be induced to use more recycled material at all, thus the demand for recycled materials will not be affected as much as could be expected. Furthermore, a local virgin material tax will not affect the export of recycled materials. Since a large part of recycled materials, like paper and plastic, are exported to the United States, the virgin material tax will be less effective.

Palmer and Walls (1994) demonstrate that a virgin material tax indeed encourages the use of recycled materials, but that it also discourages production and consumption. Thus the social costs of a virgin material tax are too high to be efficient. Not all studies, however, agree on the macro-economic effects of a virgin material tax. Bruvoll *et al.* (1999) show that the improvement in environmental quality caused by increased recycling could increase productivity and in turn curb the decline in production and consumption. Fullerton and Kinnaman (1995) conclude that the introduction of a virgin material tax is only useful in addressing the environmental problems caused by extraction of virgin materials (strip mining) and cannot be used to tackle the environmental problems caused by waste generation. Palmer and Walls (1997) share the same opinion. They advocate that the deposit-refund system is a more useful policy instrument.

Besides taxing virgin material, the use of recycled material can be stimulated by subsidizing recycling. The use of a recycling subsidy has been extensively researched. Fullerton and Wu (1998) argue that if unit-based prices cannot be implemented due to risk of illegal disposal, recycling subsidies aimed at the firm (s^{fr}) can indeed improve welfare and should be chosen as the preferred tool to minimize the social costs of waste disposal. Palmer and Walls (1994) find that recycling subsidies for both firms and households (s^{fr} *en* s^{hr}) supply a more optimal mix between waste generation and recycling. If, however, a recycling subsidy would be the sole instrument to be implemented it could excessively stimulate both production and consumption, and thus waste generation. They advocate that it would be better to introduce a recycling subsidy combined with a consumption tax (t^c). Only this combination may induce consumers to recycle more and reduce waste generation (Palmer *et al.*, 1997). This subsidy/tax system is similar to a deposit refund system.

Other studies have also identified the deposit-refund system as the optimal policy instrument to reduce waste generation and increase recycling. With the help of a general equilibrium model, Fullerton and Kinnaman (1995) illustrate how a deposit-refund system could significantly reduce social waste disposal costs and that it should be implemented as the preferred policy instrument whenever illegal waste disposal is a serious option. Palmer *et al.* (1997) show that the introduction of a deposit-refund system would have effectively reduced waste generation by 7.5% in 1990, thus resulting in a social welfare benefit of \$33 per tonne of waste. Other economic studies, like Dobbs (1991), Dinan (1993), Palmer and Walls (1994), and Atri and Schellberg (1995) favor the use of a deposit/refund system.

In the optimal deposit/refund system the deposit equals the marginal social costs of waste disposal. The refund is equal to the social waste disposal costs minus the marginal social costs of recycling. If the marginal social costs of recycling equal zero, the refund matches the deposit exactly. The deposit can be levied on either the production or the sales of a product and the refund can be given to either the producer or the consumer. If the refund is returned to the consumer, the consumer has a direct incentive to increase recycling. This increase in recycled materials will in turn drive the price of recycling down. Thus the demand for recycled material will increase. If the refund is given to the firms, the firms will start to demand more recycled material, this will increase the price of recycled materials thus giving the consumers an incentive to recycle more (Atri and Schelberg, 1995). In addition, both Fullerton and Wu (1998) and Eichner and Pethig (2001) demonstrate that the deposit/refund provides an incentive to change the design of the product to improve its recyclability.

High transaction costs are the greatest disadvantage of the deposit/refund system. The most optimal deposit/refund system would be a differentiated system for various materials. This system, however, is also the most expensive. Palmer *et al.* (1995) calculate that the marginal costs of reducing waste by 10 percent are equal to \$45 per tonne waste reduced. In comparison, the marginal cost of reducing waste by 10 percent with the use of a recycling subsidy would be equal to \$98 per tonne of waste. These marginal costs, however, do not include the administrative costs of implementing the system. If these costs are included, the marginal costs of the deposit/refund system will increase sharply. However, Palmer *et al.* find that in the case of California the marginal costs including the administrative cost of a deposit/refund system are still expected to be lower than the marginal costs including administrative costs of a recycling subsidy, thus showing that the deposit/refund system is more cost-efficient than the recycling subsidy.

Dinan (1993) asserts that if the costs of introducing a deposit/refund system are high, the government should select those waste materials, which either cause a large part of the municipal solid waste stream, like old newspapers, or of which disposal causes large social costs, like flashlight batteries. Fullerton and Wolverton (2000), however, point out that it is not necessary that the deposit and the refund are exactly equal to each other, nor is it necessary that the deposit and the refund are placed on the same actors in the market place. Thus, as suggested by Palmer *et al.* (1995), the deposit-refund may be placed upstream to avoid dealing with private households, consequently avoiding substantial transaction costs. Calcott and Walls (2002) support these results. They show that in a system where the producers of goods that are recycled pay a tax-upfront, which equals the refund received by recyclers, and the producers of goods that are not recycled pay an advanced disposal fee, which equals the marginal costs of treatment, can indeed provide a second-best equilibrium solution.

Besides tax and subsidy instruments, the government also has command and control policies at its disposal. Most of the command and control options are aimed at promoting waste recycling. The three most popular command and control policy tools in both the United States and the European Union are: 1) Product specific minimum content standards, 2) Material specific utilization requirements and 3) Producer responsibility (Goddard, 1995).

1) The product specific minimum content standards specify the minimum quantity of recovered materials that a product must contain. Materials most commonly mentioned are aluminum, steel, glass, and paper used in packaging. The product specific minimum content standards have been applied in various parts of the United States and various countries in the European Union. Although widely applied, little research has been done to examine the effectiveness of the minimum content standards. One exception is the research conducted by Palmer and Walls (1997). In this study, Palmer and Walls evaluate the effectiveness of minimum content standards as compared to taxes and subsidies. They find that while the material content standard can lead to lower output of municipal solid waste, it can also lead to inefficient use of other production factors such as labor. To negate these unintentional effects, it may be necessary to implement additional taxes and subsidies. Moreover, to set efficient standards, extensive knowledge about the production function of a firm is necessary. This information is not always available. If industries are heterogeneous, setting a uniform, optimal standard may be impossible.

2) The material specific utilization requirements specifies that producers must recover and use a certain predefined percentage of specific materials, like packaging and non-

durable goods, that normally would be disposed of. Materials that are normally used are paper, glass, aluminum, steel, and plastic. New Jersey, for example, passed a law in 1987 that 25% of all waste had to be recycled. Municipalities were forced to develop and submit a recycling plan as part of their solid waste plans. The law stated that three materials had to be recycled; the municipalities were allowed to choose which three materials. Most municipalities chose paper, aluminum, and glass since these materials take up a large part of the typical household waste stream. The law was indeed effective and after two years the state of New Jersey was well on its way to realizing the 25% recycling rate (Callan and Thomas, 1996).

3) The producer responsibility specifies that producers of goods ultimately destined for disposal directly responsible for the collection, recycling, the disposal, and the associated financial and external costs. Walls (2003) demonstrates how difficult it is for the government to design and implement cost-effective environmental policies to spur producer responsibility. She warns of the danger that the costs of introducing producer responsibility may outweigh the benefits of such a system. Runkel (2002) shows how producer responsibility can influence both durability and welfare. He argues that under perfect competition, the producer responsibility can achieve a first-best equilibrium solution. Even if no perfect competition exists, the producer responsibility will raise welfare as compared to a situation where producers do not receive a price incentive to limit waste disposal.

Germany was the first country to introduce a far-reaching producer responsibility program in 1991. This program, which is known as the ‘green dot’ program, sets extensive recycling targets for several materials, such as glass, paper, aluminum, and plastic, and aims to achieve these targets by making the industry directly responsible for the collection and recycling of all its packaging. The industry responded to these regulations by forming a private non-profit company Duales System Deutschland, which provides collection and recycling services for consumers. The DSD system sells participating companies the right to put a green dot on their product, a symbol, which guarantees that the packaging of the product is eligible for the services provided by the DSD. The program has been a huge success in terms of recycling efforts, but it has not been without problems. The costs of the program are quite high and the system of the green dot is prone to fraudulent activities (Goddard, 1995). The large increase in recycling rates, however, has stimulated various countries such as the Netherlands, France, the United States, and Japan to implement producer responsibility programs of their own. For an extensive overview of the implementation of producer responsibility programs in other countries see Palmer and Walls (2002).

This section has provided a concise overview of the literature dealing with policy instruments to control the problem of municipal solid waste management. As should be clear, there is no unique solution to policy questions regarding the municipal solid waste problem. Most of the studies agree that without the possibility of illegal disposal, the unit-based pricing scheme for collection of waste is the preferred policy instrument. If illegal disposal is a serious threat, most of the studies favor a deposit/refund system.

2.2.3 Elasticities of the demand for waste disposal services and consumer attitudes towards recycling

The success of market based policy instruments, like the ones mentioned in Section 2.2.2, depends on the elasticity of demand for waste disposal services. For example, a unit-based price for waste disposal will only affect the disposal of waste if the demand for disposal services is sensitive to the price of the waste disposal services. As municipalities have been experimenting with the introduction of recycling programs, unit-based pricing, and deposit/refund systems, a large range of empirical studies discussing the price elasticity of waste generation have been conducted. In this section, I will discuss the most important literature on this subject.

Wertz (1976) analyzed the households' responsiveness to unit-based prices. By comparing the average quantity of waste generated in San Francisco, a city with a user fee, with the average quantity generated by an average town of the United states, without a user fee, Wertz calculated a price elasticity of demand equal to –0.15.

Hong *et al.* (1993) examined the effects of volume-based pricing using a survey of 2298 households from Portland OR, USA. Hong *et al*. estimated a price elasticity of demand equal to –0.03 and an income elasticity of 0.049 suggesting that unit-based pricing only affects demand in a minimal way. They did, however, find that the demand for recycling services is influenced positively by the introduction of volume-based pricing. They also concluded that households are less likely to increase recycling if recycling requires more effort and that a larger household is not only more likely to recycle, but also to generate more waste than a smaller household.

Jenkins (1993) gathered data from 14 municipalities in the United States (including 10 municipalities that charged a unit-based price) over several years. She found an inelastic demand for waste disposal, reporting a price elasticity of –0.12. Jenkins concluded that waste generation and recycling is positively influenced by the size of the household. However, she also found the effect to be statistically insignificant.

Miranda *et al.* (1994) used data from a 21-city sample to estimate the effects of introducing a unit-based price. They found that unit-based pricing provides residents with a strong incentive to both reduce waste and recycle it. They note, however, that most municipalities implement a unit-based price in combination with an aggressive recycling program. In the one municipality that introduced unit-base pricing on its own, the experiment failed. Households turned to private waste collectors and illegal disposal increased significantly. Therefore, this municipality chose to return to the flat fee-pricing system. This evidence, although anecdotal, seems to suggest that a unit-based pricing scheme cannot be successful without a recycling program.

Morris and Holthausen (1994) use a household production model to simulate responses to different pricing systems using calibration techniques. They estimate that the elasticity of demand for waste disposal services was in the range of –0.51 and -0.6.

Reschovsky and Stone (1994) employed an econometric model to estimate the actual household responses to unit-based pricing. They used data from 3040 households from Tompkins Country, New York. Based on these data, they estimated income elasticities for the demand of collection services equal to 0.23 in case of the introduction of volume-based pricing and 0.24 in case of the introduction of weight-based pricing. These results are quite similar to the results found by Wertz (1976). Reschovsky and Stone try to determine how much waste was illegally disposed of. They found that much of the illegal dumping takes place in the form of the use of alternative dumping facilities, such as roadside dumpsters. They were unable to determine how often illegal dumping or burning occurred. They argued that households are not quite as sensitive to the increased marginal costs of waste disposal as they are to the increased marginal costs of waste reduction. Thus households will only try to reduce waste generation if the marginal costs of waste reduction do not increase too greatly. If the marginal costs of waste reduction increase too much, households will dump waste illegally to reduce the costs of waste disposal. These results suggest that households may have an aversion towards the introduction of a unit-based pricing, indicating that municipalities would be wise to combine a unit-based price with recycling programs or subsidies. Introducing a unit-based price without such a program would be unpopular and less effective.

Strathman *et al.* (1995) estimated the price elasticity of demand for solid waste disposal services using data from Portland, Oregon. They used data on the generation of waste during January 1984 to December 1991. They found an elasticity of demand of –0.45. This elasticity is quite a bit lower than elasticities cited earlier. Strathman *et al.* note that they may have overestimated the absolute elasticity as they expect that the propensity of illegal disposal may be somewhat higher in the Portland region due to the large amount of public land available in this area.

Fullerton and Kinnaman (1996) used household data that were not based on self-reported surveys. They gathered data about the weight and volume of municipal solid waste and recycling efforts of 75 households four weeks prior to, and following the introduction of a volume-based price in Charlottesville, VA. In this municipality, a recycling program had all ready been operational for about a year. They found that the quantity of solid waste generated decreased only slightly, but that the volume of the waste collected decreased all the more. The density of the municipal solid waste increased significantly, from 15 pounds per bag to just over 20 pounds per bag. They estimated that the introduction of the unit-based price resulted in ten percent less waste, four percent more illegal dumping, and 14 percent more recycling.

Callan and Thomas (1997) found that the implementation of a unit-based price would increase the portion of waste recycled by 6.6 percent. If the introduction of a unit-based price was combined with the introduction of a recycling program, the portion of waste recycled would increase by 12.1 percent.

Kinnaman and Fullerton (2000) were the first to estimate both the levels of recycling and the level of waste disposal simultaneously after the introduction of a unit-based price. They estimate that the cross price elasticity of demand for recycling is 0.220. Moreover, they not only found that an implementation of a $1 unit-based price can decrease the quantity of rest waste generated by 415 pounds per person year, but that it would only increase the quantity of recyclable waste by 30 pounds per person per year. The difference can be partly explained by increased home composting and prevention, but also points towards the increased illegal disposal of waste.

Although the calculated elasticity of the demand for waste disposal services differs quite a lot between different studies, we can conclude that the demand for waste disposal services is inelastic. The introduction of a unit-based price will result in a reduction of waste. At least part of this reduction, however, may be caused by increased illegal disposal of waste. It is difficult to give definite empirical proof of increased illegal disposal. Although survey respondents claim that illegal disposal has increased after the introduction of a unit-based price, municipalities have not reported increased costs due to illegal dumping and littering.

The empirical studies discussed above report various elasticities of demand for waste disposal services. These differences may be partly explained by differences in attitudes of the households. Several empirical studies have analyzed why consumers recycle or compost at home. In the next couple of paragraphs, a brief overview of these studies is given. For a more extended overview, see Fenech (2002)

Several studies, for example, Hornik *et al.* (1995), McDonald and Ball (1998), Callan and Thomas (1999), Bruvoll *et al.* (2000, 2002), Tucker and Speirs (2002), Ando and Gosselin (2003) and Jenkins *et al.* (2003), have shown that the opportunity cost of time is a significant determinant for recycling of materials. The more households have to do to recycle and separate waste, the less willing they are to do so. A majority of the consumers are willing to pay a private company about 20 dollars a year to take away the burden of separating waste (Bruvoll *et al.,* 2002). Jenkins *et al.* (2003) conclude that consumers are more likely to recycle materials like aluminum, or paper given that effort in recycling these materials is less than other materials such as glass, plastic, and organic waste.

Recycling behavior is influenced by socio-economic factors such as income, education, population density, single or multiple family dwellings, household size and average age of the head of the household. Most empirical studies, like Jenkins *et al.* (2003), find that income and education are positively correlated with recycling. Population density is negatively correlated with recycling and specifically with home composting of organic waste. An explanation for this correlation is the growing scarcity of suitable outdoor storage of waste as the population density increases. Age and household size have a positive correlation with recycling. Ando and Gosselin (2003) show that multi-family dwellings are less likely to recycle than single-family dwellings. They find that differences in recycling convenience and household demographics are the main reason why this occurs.

Introducing a unit-based pricing system not only increases recycling of waste, but also changes the attitude of households towards waste. Fullerton and Kinnaman (1996) demonstrate that the introduction of a volume-based pricing system in Charlottesville, USA did not so much decrease the quantity of waste generated, but instead decreased the volume of the waste generated. Households reduced the number of bags they generated by crushing the waste down in size, rather than by preventing or recycling it. Households, however, were already participating in voluntary recycling programs before the introduction of the volume-based price, thus the incremental benefit of the volume-based price was low.

Sterner and Bartelings (1999) show that the introduction of a unit-based price in Tvååker, a municipality in Sweden, led to a significant reduction of the quantity of waste collected and that the quantity of waste recycled increased. With an extensive survey of about 600 households and focusing on the motivation behind recycling, they demonstrate that whilst people are encouraged by economic incentives, this is not the only reason why they start to recycle. The amount of time and effort invested in recycling are greater than can be purely motivated by savings on their waste management bill. Halvorsen and Kipperberg (2003) support this conclusion. Berglund (2003) analyses the effect of moral motives on household recycling. He finds that moral motives significantly reduce the costs associated with recycling efforts, thus consumers are more willing to recycle even when they are not financially compensated for doing so.

2.3 The optimal mix of waste management methods

In the previous section, I have outlined the instruments policy makers have at their disposal to reduce waste generation. Even if a society reduces waste generation as cost-efficiently as possible, it is likely that a substantial quantity of waste will still have to be disposed of. In this section, I will, therefore, discuss how waste can be treated and what kind of environmental problems waste treatment can cause.

The physical and thermal properties of various solid waste types, such as calorific value, ash, moisture and bulk density, give a reasonable indication of the likely environmental impact that the disposal of waste will have. Much more uncertainty surrounds the biodegradation processes in landfill sites and pollution of both surface and groundwater around such sites (Turner, 1995).

All available disposal options, *i.e.* re-use, recycling, composting, incineration, and landfilling, will lead to environmental externalities. These may range from neighborhood nuisances, such as the noise or smell created by the presence of a disposal site, to air and water pollution, health impacts and congestion costs (see for example Daskalopoulos *et al.*, 1998 or Sonesson *et al.*, 2000).

Since most of these externalities are negative much has be done to prevent them. Most industrialized countries have adopted waste management policies designed to reduce the effects of waste disposal as much as possible. The Dutch government, like most governments in the European Union, bases its waste policy on the waste hierarchy, which ranks waste management methods in a strictly descending order. According to the waste hierarchy, prevention is the best option, followed by recycling or re-use, composting, incineration and finally landfilling. A graphic representation of the waste hierarchy is given in Figure 2-3 .

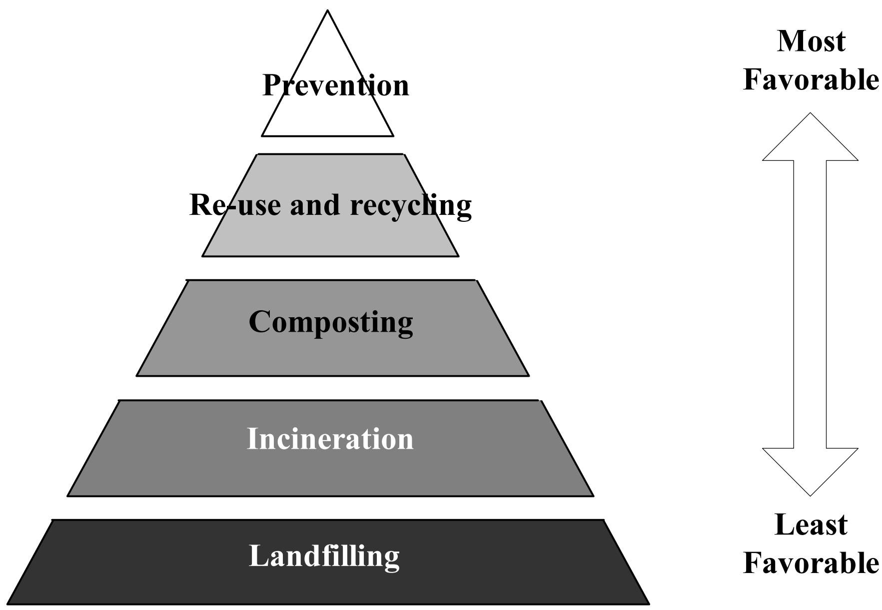

Figure 2-3 The waste hierarchy

The waste hierarchy provides clear guidelines as how to deal with waste management problems. Based on the waste hierarchy, the Dutch government has attempted to stimulate composting and incineration, instead of landfilling. In 1994, the Dutch government passed laws to force municipalities to collect organic waste and rest waste separately, thus ensuring that organic waste would be composted instead of landfilled or incinerated. In 1998, the government forbade landfilling of waste that could be incinerated and, in 2000, it instituted a substantial tax on landfilling of waste, raising the price of landfilling to about €128 per tonne, which is well above the cost of incineration (about €106 per tonne). Other countries, like Denmark and the United Kingdom, have also adopted landfill taxes to discourage landfilling (for more information, see Sedee *et al.*, 2000).

The concept of the waste hierarchy, however, appears to reflect some form of 'green intuition', with little consideration for the actual social costs and benefits of that particular policy. While the waste hierarchy serves a useful purpose, many environmental economists have argued that the hierarchy should not be viewed as fixed and that one should exercise a degree of caution before drawing conclusions about what represents the optimal waste management disposal practice (Brisson, 1997; Dijkgraaf and Vollebergh, 1997; Faaij *et al.,* 1998).

Brisson (1997) offers a clear intuitive account of why the waste hierarchy may not be the best policy in every situation. In the next section, the advantages and disadvantages of the waste hierarchy will be discussed in greater detail.

2.3.1 Financial cost problem

Using a simple optimization model, Brisson (1997) shows how the optimal waste treatment method can be determined. Suppose, for the purposes of simplification, that there are three possible ways of dealing with waste, namely recycling (including composting), incineration, and landfilling. Waste (W) is recycled (W_R), incinerated (W_I), or landfilled (W_L):

$$W = W_R + W_I + W_L \tag{2.7}$$

Figure 2-4 illustrates the optimal mix between the three waste management options. The choice between these options is left solely to the market, so the costs reflect only financial costs rather than environmental costs. The households will choose to recycle waste only if recycling provides benefits. This would mean that if recycling was left to market forces and thus considered exogenous to the cost minimization problem, recycling would take place up to the point where marginal profit of recycling (*i.e.* $-MC_R$) equals zero. The remainder of the waste (W-W_R) will either have to be incinerated or landfilled. Given that there are no institutional constraints, the policy maker will base the choice between waste treatment options solely on the net social costs of disposal and choose levels of incineration and landfilling so that the marginal costs of incineration will equal the marginal costs of landfilling.

The line MC_{L+I} in Figure 2-4 illustrates the total quantity of waste that can be landfilled and incinerated at any given marginal costs. The minimum marginal costs MC_{min} at which all the waste (equal to the quantity $W-W_R$) can be disposed of is reached when this line intersects the vertical $W-W_R$ line. At this point the marginal costs of landfill are equal to the marginal costs of incineration. A quantity of W_I will be incinerated and a quantity of W_L will be landfilled (Brisson, 1996).

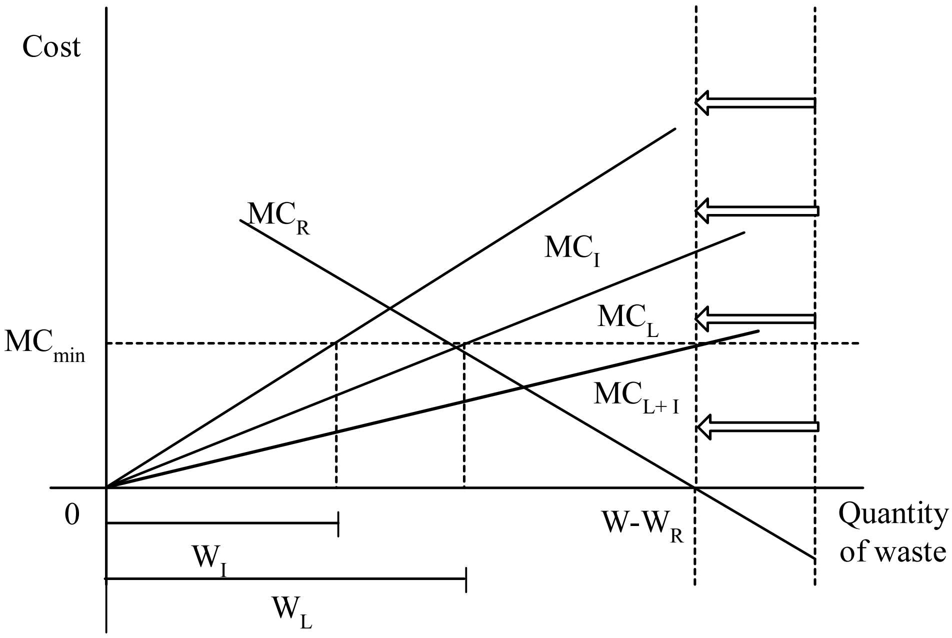

Figure 2-4 Mix of waste management options considering only the financial costs.

Source: Brisson (1996)

2.3.2 Social cost problem

Waste treatment costs, however, do not only consist of financial costs but also environmental ones. To choose between waste treatment options, these environmental costs and benefits should be taken into consideration. The problem the policy maker faces becomes one of:

$$\begin{aligned} &Min\ \ NSC(W) \\ &s.t.\ \ \ W = W_R + W_I + W_L \end{aligned} \tag{2.8}$$

Where $NSC(\overline{W})$ is net social cost of waste management of all waste materials.

The net social costs of waste management are equal to the sum of the net social costs of each waste disposal option:

$$NSC(W) = NSC(W_R) + NSC(W_I) + NSC(W_L) \tag{2.9}$$

The net social costs of each waste handling option are composed of private financial costs (*PC*), external costs (*EC*), and external benefits of both sales of recycled materials (R_r) or energy (R_{en}).

$$
\begin{aligned}
NSC(W_R) &= PC_R + EC_R - R_R \\
NSC(W_I) &= PC_I + EC_I - R_{en} \\
NSC(W_L) &= PC_L + EC_L - R_{en}
\end{aligned}
\tag{2.10}
$$

Each of the waste treatment options will lead to different costs and benefits. Landfilling results mainly in methane emissions (in the Netherlands about 40% of the total methane production) and incineration is a big contributor to, for example, CO_2 emissions and dioxin emissions. The process of recycling is also not free from environmental pollution (see, for example, Powell *et al.*, 1996 and Butler and Hooper, 2000). One big disadvantage of recycling, which should also be kept in mind, is that it is often impossible to use the material for its original purpose. Materials become polluted when thrown away together with other waste materials. The quality of recycled material will, therefore, be lower than the quality of virgin material. Instead of a full recycling process, a sequence of degradation of material quality takes place. Waste generated from high quality applications is transformed into low quality applications (Starreveld and van Ierland, 1994). For an overview of environmental emissions created by waste treatment in the Netherlands, see Chapter 3.

Waste treatment will also provide some benefits to society. In the case of recycling, it is possible to sell part of the materials recovered; in the case of incineration and landfilling, it is possible to sell recovered energy. Recovery of materials and energy will have both financial benefits and environmental benefits in the form of avoided financial and environmental costs caused by the production of virgin materials and energy. In the case of paper recycling, Nakamura (1999) reveals that these benefits may be extensive.

The optimization problem, as presented in equations 2.8 to 2.10, is illustrated graphically in Figure 2-5. In contrast to Figure 2-4 the net social costs of the three waste handling options, as represent in the total marginal cost line, include environmental costs. Furthermore, recycling is no longer exogenously given to the optimization problem, but included as an option. Akin to the previous picture, the line MC_{R+L+I} represents the total quantity of waste that can be disposed of, *i.e.* recycled, landfilled, or incinerated, given the marginal costs. The minimal marginal cost of disposal (MC_{min}) can again be determined (intersection of the *W*-line with the MC_{R+L+I} line) and the optimal quantities of recycling, incineration and landfilling can be found (Brisson, 1996).

As shown in Figure 2-5 it is not optimal to recycle all waste. This is far too expensive, even when the environmental costs caused by waste treatment are considered. The same holds for incineration as compared to landfilling. The choice between waste handling options is also not as black and white as the waste hierarchy suggests, timing may be of utmost importance if the government wants to minimize the costs of waste treatment. Ready and Ready (1995), for example, illustrate that landfilling becomes increasingly more expensive when space in the landfill is depleted. They argue that it may be optimal if waste recycling, composting, and incineration programs are delayed until the landfill is partially full. Highfill and McAsey (2001) find that both the wealth of the municipality and the landfill capacity available to the municipality are important factors in the success of recycling programs. Wealthy municipalities should recycle as much as possible; poorer municipalities, however, should optimally rely less on recycling and first exhaust the available landfill space.

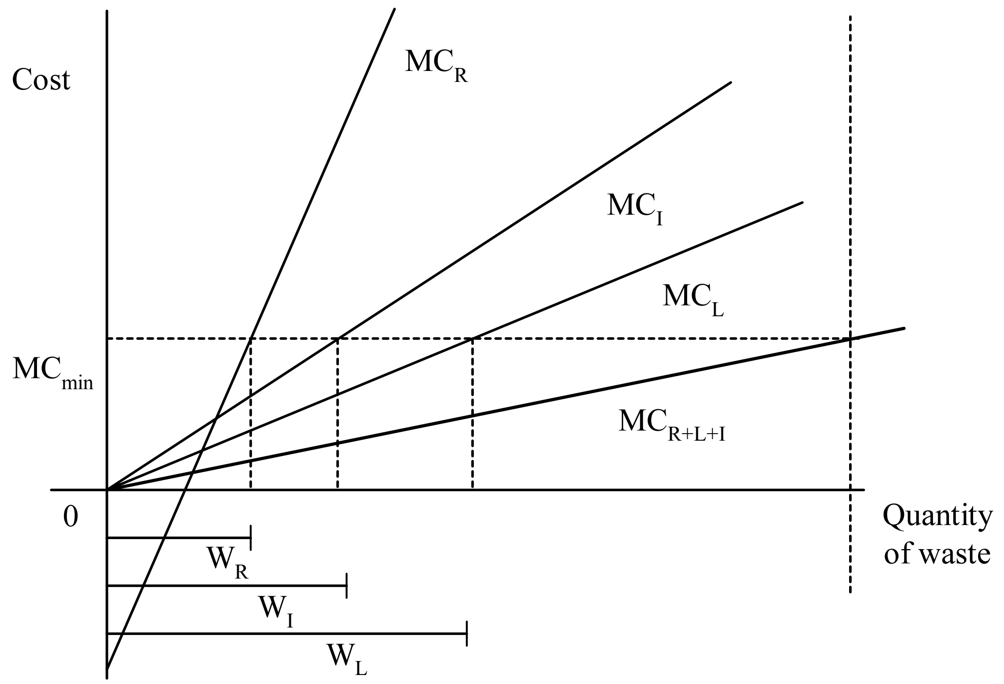

Figure 2-5 The optimal level of recycling, incineration, and landfilling under full social costs

Source: Brisson, 1996

By applying a strict regime as the waste hierarchy, the government could be stimulating inefficiently high levels of recycling and incineration, thus making society pay far too much for waste treatment. As it is extremely hard to determine the total social costs of waste treatment, it will be very difficult for the government to either determine how much waste should optimally be recycled, incinerated and landfilled or to internalize the social costs of waste treatment in the price of waste treatment. Moreover, most countries are just starting to stimulate recycling and incineration and to discourage landfilling. Thus, the waste hierarchy, although not optimal, may indeed help us to channel waste management policies in the right direction as long as we bear in mind that it is not an optimal policy tool and should not be applied mindlessly.

2.3.3 Estimating environmental costs

As mentioned above, it is important to determine the full social costs of waste treatment in order to design the optimal policy mix. One of the greatest difficulties in finding these social costs is the estimation of the environmental costs associated with waste disposal. This requires an assessment of the amount of actual pollution that occurs. Economic valuation or lifecycle analysis for waste disposal, for example, could be helpful in this regard.

Economic valuation

Environmental externalities may be valued by a cost-benefit analysis (CBA). The key principle, which underpins CBA, is very simple. The impact of the project on each affected person is identified at each point in time. The value of any gain or loss to each person is then estimated. These valuations should be based on the preferences of the affected individuals, and should ideally reflect each person's willingness to pay for an improvement or willingness to be compensated for a loss (Perman *et al*, 1996). It enables direct translation of emissions and environmental effects into costs.

One way of determining the environmental externalities of a landfill site is by using the hedonic pricing method. For example, the development of a landfill site may cause a temporary or permanent drop in real-estate prices in the neighboring area. This negative value could be taken as an indicator of the costs of visual, odor, and health effects of the landfill site (see for example Hite *et al.,* 2001). Another useful cost-benefit analysis is contingent valuation method (see also Hanley and Spash, 1993).

Lifecycle assessment

Life cycle assessment (LCA) is a tool to predict the overall environmental impact of a product or service. It is defined by the society of environmental toxicology and chemistry (SETAC) as:

> "A process to evaluate the environmental burdens associated with a product, process or activity by identifying and quantifying energy and materials used and wastes released to the environment; and to identify and evaluate opportunities to affect environmental improvements. The assessment includes the entire life cycle of the product, process or activity, encompassing extracting and processing raw materials; manufacturing, transportation and distribution; use, re-use, maintenance; recycling and final disposal."
>
> SETAC (1993) p.5

For each stage, the inputs (in terms of raw materials and energy) and outputs (in terms of emissions to air and water and as waste) are calculated, and these are aggregated over the total lifecycle. These in- and outputs are converted into their environmental impacts. The overall environmental effects of the lifecycle are then calculated by taking the sum of the environmental impacts. A comparison can thus be made between different products or services based on their environmental impacts. Rules for the first two stages of a LCA, namely defining the life cycle of a product and producing an inventory of the total in- and outputs of the system have been made and are widely accepted. Unfortunately, no rules have yet been established for the third (impact assessment) and fourth stages (evaluation). There is still much discussion on the one 'correct' way of assessing impacts and valuing these (Kirkpatrick, 1995).

In particular, the final step in a lifecycle assessment is surrounded by much controversy. The final step in the LCA is the valuation of the environmental effects. Comparing one lifecycle option with another will not normally show one option, which is 'environmentally superior', but will demonstrate the trade-offs between the two options. To simplify the decision-making process, attempts have been made to aggregate the results, ultimately to a single environmental score. This involves some weighting of the importance of different environmental problems. Not surprisingly, at the moment there are no generally accepted objectives to weight environmental problems. As a result, attempts so far have relied on (1) scoring systems from 'expert panels', which are subjective (*e.g*. Landbank, 1991); (2) comparing emissions against government targets, which are political and not global; or (3) by converting all impact into monetary units. None of these schemes have been widely accepted. It must be borne in mind that weighting environmental problems and aggregating them into a single score may simplify policy choices, but by weighting them these policies choices have implicitly already been made. Thus, broad acceptance of the outcome of the decision is uncertain (White *et al*., 1997).

In most LCA studies for waste treatment, source reduction is excluded from the model. If one is interested in minimisation of waste, it is better to use a regular LCA instead of the more detailed LCA of waste or it is possible to use material-product chains (see Kandelaars, 1998). Interesting LCA studies for waste treatment can be found in CE (1996), Powell *et al.* (1996), White *et al.* (1997), and McDougall and White (1998) .

2.4 Location problem of waste handling facilities

One aspect of the waste-handling problem has still not yet been properly addressed. As mentioned earlier, waste management becomes increasingly difficult due to a shortage of acceptable landfill and incineration sites. It is, therefore, important to give due consideration to determining where the waste should be treated.

A couple of decades ago, waste handling was mostly a local affair. Municipalities owned their own landfill sites and dumped the waste they collected at these sites. As transport of waste was too expensive or impossible, there was no choice about where waste would be treated. This situation has, however, changed significantly in the last few decades. In the Netherlands, we see that waste management has frequently become an issue of provincial or even national or international concern (see van Beukering, 2001). In line with the waste hierarchy, incineration has been stimulated instead of landfilling. Since incineration is typically one of the waste treatment options that are too expensive on a small scale, we see that waste is transported over longer distances. A province only has a couple of waste incineration plants, which handle almost all the municipal solid waste generated in that province.

Until the year 2000, laws in the Netherlands forbade the transportation of waste across province boundaries. In 2000, however, these laws were abolished. Since January 2000, municipalities and companies have been allowed to transport waste outside the boundaries of their province and they can offer their waste to any waste treatment unit. This means that more competition has been introduced into the waste market. Internationally, the boundaries for waste transport are also disappearing. The laws of the European Union already permit the export of recyclable waste. It is expected that in a couple of years the boundaries for combustible waste will also open between the different countries in the European Union. These facts give cause to investigate whether the current locations of waste treatment units are optimally situated throughout the Netherlands and the European Union.

Already there are signs that the waste market is rapidly changing. In the Netherlands and throughout the rest of Europe, a trend towards an increasing scale, concentration, and vertical integration as well as a merging towards a multi-utility structure, can be identified. The waste market has become far more privatized. Energy companies have become interested in merging with waste treatment units thus creating multi-utility companies. Waste treatment companies are interested in vertical integration; several of the largest companies are now focusing on providing all services connected to waste treatment, from collection to final disposal. International companies have also become interested in providing their services throughout the whole of Europe (WMC, 2002).

The spatial aspects of the waste treatment problem have been analyzed mostly in an optimization setting. These studies focused on finding the optimal location of a waste treatment center given that the waste treatment costs for society are minimized. Examples of these kinds of studies can be found in Opaluch *et al.* (1993), Macauley *et al.* (2002), and Ye and Yezer (1997). In Section 2.4.1, I will discuss how the waste treatment location problem can be analyzed in an optimization model. Furthermore, in Section 2.4.2, I will explore how it can also be analyzed within a spatial general equilibrium framework.

2.4.1 The spatial waste management problem: an optimization approach

There are many potential sites suitable to build a waste disposal facility. Each possible disposal location can accommodate different annual quantities of waste and due to economies of scale the larger the size of a landfill or incinerator, the lower the average annual disposal costs per tonne of waste. If no transport costs are included, the obvious least cost solution is one mammoth site handling the waste from all municipalities. The existence of transport costs, however, complicates the issue. Beyond a certain distance, the transport costs may well offset the benefits due to economies of scale, making it more efficient to build additional waste handling facilities to save on the transport costs. This problem is illustrated in Figure 2-6.

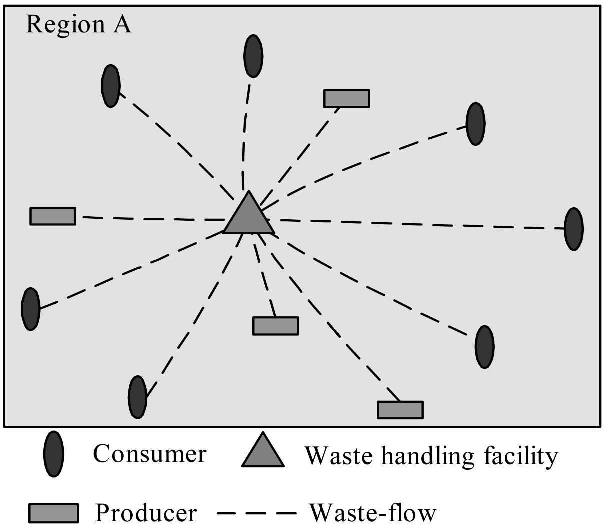

Figure 2-6 Schematic representation of location problem of waste treatment facilities.

Choosing a site is not only an economic problem that encompasses a trade-off between transport costs and economies of scale. Rather, selecting a site involves significant social trade-offs. For example, one technically suitable site may contain relatively large amounts of woodland; another potential site may contain more wetlands or farmlands. Choosing between sites implies a social preference for one type of land over another. Similarly, one technically suitable location may be situated near to a large residential area, while another site may have schools nearby or may require higher costs for site development. Thus, the optimal choice includes political, economic, and social considerations (Opaluch *et al.*, 1993).

The optimal location may also depend on the demand for that service. For example, Highfill *et al.* (1994) show that when a recycling program starts, the optimal location of a recycling center is near the landfill site. If the recycling program becomes a success, the optimal location will shift from the landfill site to a central location within the municipality.

In addition to the financial costs of waste disposal there are potential regional impacts on public good resources and local environmental costs that should also be taken into consideration. At a regional level, waste treatment facilities may produce negative effects on watersheds, aquifers, woodlands, and wetlands. Local negative impacts include traffic congestion, odor, noise, air pollution, and the community health risks posed by a disposal site.

To analyze the location problem of waste, a regional waste disposal model that minimizes social costs, subject to technological, health and safety regulations can be built. The social costs include both financial costs of waste treatment and the monetary equivalent of the environmental costs.

Symbolically the problem can be expressed as follows (Opaluch *et al.,* 1993):

$$\begin{aligned} &\underset{j}{Min}\ FC(J) + EC(J), \quad where\ J \in \varphi \\ &s.t.\ \ HS(J) \geq 0 \end{aligned} \tag{2.11}$$

Where *J* represent a vector of characteristics describing a site; *FC(J)* are the present values of financial costs of building and operating (including transport costs) a waste handling facility; *EC(J)* represents the monetary value of environmental damages; φ represents a set of all possible sites and *HS(J)* represents technological, health and safety constraints.

Mixed integer programming

The location problem of a waste handling facility involves a choice between different sites. Since there are only two choices, either a waste handling facility is built on a site or not, the problem can best be solved by a mixed integer programming (MIP) approach. The MIP approach involves a solution of a constrained optimization problem (cost minimization or profit maximizations) where the objective function is linear in the activity level (variables) and where some variables can only have integer values[2].

In Malarin and Vaughan (1997), a simple mixed integer-programming model for the waste problem is introduced. The objective is to minimize the present value of the

[2] The most common areas of application are the 'yes-or-no' decisions. With just two choices, we can represent such decisions by binary variables. Thus the jth yes-or-no decision would be represented by:

$$x_j = \begin{Bmatrix} 1, & \text{if decision j is yes} \\ 0, & \text{if decision j is no} \end{Bmatrix}$$

(Hillier and Lieberman, 1989)

total annual investment, operating and transport costs of waste treatment by determining optimal locations and sizes for the waste treatment facilities:

$$\underset{IS_{ij},GS_{kjt},VS_{ij}}{Min} \quad C = \sum_i \sum_j FC_{ij} IS_{js} + \sum_i \sum_j VC_{ij} VS_{js} + \sum_i \sum_k \sum_j TC_{ij} GS_{js} \tag{2.12}$$

Given the following four constraints:

$$\begin{array}{lll} (1) & \sum_i \sum_j GS_{kij} \geq Waste_k & , \forall \; k \\ (2) & \sum_k GS_{kij} = VS_{ij} & , \forall \; k \\ (3) & CAPACITY_i * IS_{ij} \geq VS_{ij} & , \forall \; i,j \\ (4) & \sum_i IS_{ij} \leq 1 \;,\; \{0,1\} & , \forall \; j \end{array} \tag{2.13}$$

Where:	C	=	Annual total investment, operating, and transport costs.
	FC_{ij}	=	Annualized fixed costs of construction and capital equipment for a landfill site of size *i* at site *j*
	IS_{ij}	=	Binary integer variable that allows for annual fixed costs of construction and capital equipment of landfill of size *i* at site *j*
	VC_{ij}	=	Variable costs per tonne of operating a landfill site of size *i* at site *j*
	VS_{ij}	=	Annual number of tonnes transported to landfill site of size *i* at site *j*
	TC_{kij}	=	Cost of transporting one tonne of waste from municipality k to landfill of size *i* at site *j*
	GS_{kij}	=	Annual number of tonnes of waste transported from municipality k to landfill of size *i* at site *j*
	$WASTE_k$	=	Waste generated in municipality k
	$CAPACITY_i$	=	The annual quantity of waste that can be accepted at a landfill size *i*
	i	=	Index of possible landfill sizes
	j	=	Index of different landfill sites
	k	=	Index of different municipalities

The first constraint basically states that the total quantity of waste treated in the waste treatment units is equal to the total quantity of waste generated. This means that all waste has to be disposed of and no waste can be disposed of outside the waste treatment unit, *i.e.* no illegal dumping. The second constraint specifies that the total quantity of waste sent by the municipalities to the disposal units has to be equal to the total quantity of waste received by the disposal units. The third constraint specifies that the total quantity of waste to be disposed of in the waste treatment facility cannot be larger than the total capacity of the waste treatment facility. Finally, the fourth constraint allows for only one size of waste treatment facility to be built at any one site.

One should note that although the model is linear, the economies of scale in waste treatment are not. Oorthuys (1995) demonstrates that although treatment costs per tonne of waste decline when more waste is treated, no linear relationship exists between treatment costs and size of the waste treatment unit. The economies of scale are far larger between a small and a medium sized waste treatment unit, than between a medium and a large sized waste treatment unit. Therefore, it is not likely that the model will select one mammoth site waste treatment unit as the optimal solution given that the transport costs do increase if a mammoth size waste treatment unit is selected.

An overview of other more complex optimization models to determine the optimal location of a waste treatment unit is presented in Fiorucci *et al.* (2003). A spatial optimization model concentrating more on the economies of scale in the collection sector can be found in Callan and Thomas (2001). They show that substantial costs savings, of about 5%, can be made if the collection of recyclable waste and rest waste is combined.

2.4.2 The spatial waste management problem: a general equilibrium approach

The MIP model as presented in the previous section has some major drawbacks. For instance, it takes the quantity of waste that has to be treated as an exogenous variable. This means that the model does not consider the interactions between waste generation and waste treatment. The optimal treatment of a small quantity of waste may be very different from the optimal treatment of a larger quantity of waste. Furthermore, the treatment of waste can affect the quantity of waste that is generated. If, for example, waste treatment becomes more expensive, households and industries can decide to generate less waste. As less waste is generated, it may be that the optimal location choice of waste treatment units will be affected. It is, therefore, important to include the production, consumption, and waste treatment sectors in the model. This can be done by using a general equilibrium model with spatial aspects such as transport costs and economies of scale.

To my knowledge, no research has been done to examine the interactions between consumption and waste generation on the one hand, and waste treatment, transport costs, and economies of scale, on the other. Building a general equilibrium model with spatial aspects, like transport costs and economies of scale, however, has been researched extensively. This technique has mostly been used in international trade models. An excellent discussion on how to build various forms of spatial equilibrium models can be found in Ginsburgh and Keyzer (1997). In Chapter 6 a spatial general equilibrium model for the waste sector is presented and I will illustrate how spatial aspects of the waste management problem can affect the optimal results and, more specifically, how the quality of waste affects the optimal waste treatment method.

2.5 Conclusions

Waste generation has become a serious problem to our society. Current policies are ineffective in dealing with this issue. The regular pricing-mechanism, a flat fee, is not efficient and stimulates too much waste generation. Policy makers have several policy instruments to their disposal to stimulate waste reduction. These policy tools include a tax on waste generation, a recycling subsidy, a virgin material subsidy, an advanced disposal fee, and a deposit refund system. All of these policy instruments have been extensively researched. Almost all studies have concluded that a unit-based price on the generation of waste would result in the greatest reduction of the municipal solid waste stream. If illegal disposal, however, is an option, it is possible that consumers will start to illegally dispose of their waste, thus rendering the use of a unit-based price undesirable. In such a case, most studies favor the use of a deposit-refund system; possibly combined with a small unit-based price, since of all the policy instruments available only the unit-based price can effectively stimulate waste prevention and home composting.

Empirical studies have demonstrated that the use of a policy instrument may be efficient in one municipality, but not in another. The success of the policy largely depends on the attitudes of the inhabitants of the municipality, the relative ease of illegal disposal and the availability of recycling and re-use options. Therefore, it will be almost impossible to design an efficient waste management plan for a larger region or country. It will probably be necessary to design waste management plans for individual municipalities based on the characteristics of each municipality, in the context of a regional or national waste management plan.

In the current waste market it not possible or cost-efficient to reduce waste generation by one hundred percent. It is, therefore, important to decide how the waste that is generated may be treated in the least costly way, in terms of both financial as environmental costs. The waste hierarchy provides some good indications of how we should deal with waste. The waste hierarchy, however, should not be viewed as fixed and one should exercise a degree of caution before drawing conclusions as to the preferred waste handling option.

The location of the disposal unit also deserves some attention. It is becoming increasingly difficult to find acceptable landfill and incineration sites. Choosing a site involves a variety of economic, social and political considerations. The problem can be analyzed by mixed integer programming. Mixed integer programming, however, does not take the interaction between waste generation and waste treatment into account. It is, therefore, of utmost importance that this problem be studied in a general equilibrium setting.

3 Waste flows and management in the Netherlands: data and policies

3.1 Introduction

Since most of the budget for environmental problems is spent on waste management, waste disposal can be viewed as one of the greatest environmental problems in the Netherlands. For example, each year the Dutch government spends about 11 billion Euros on dealing with various environmental issues. As is shown in Figure 3-1, about 38% of these costs are devoted to waste disposal. Waste disposal costs consist of the costs of: (1) collection of waste, (2) treatment of waste and wastewater, and (3) development and application of a large range of emission reducing technologies. In view of these costs, it may come as no surprise that the government wishes to reduce waste generation as much as possible.

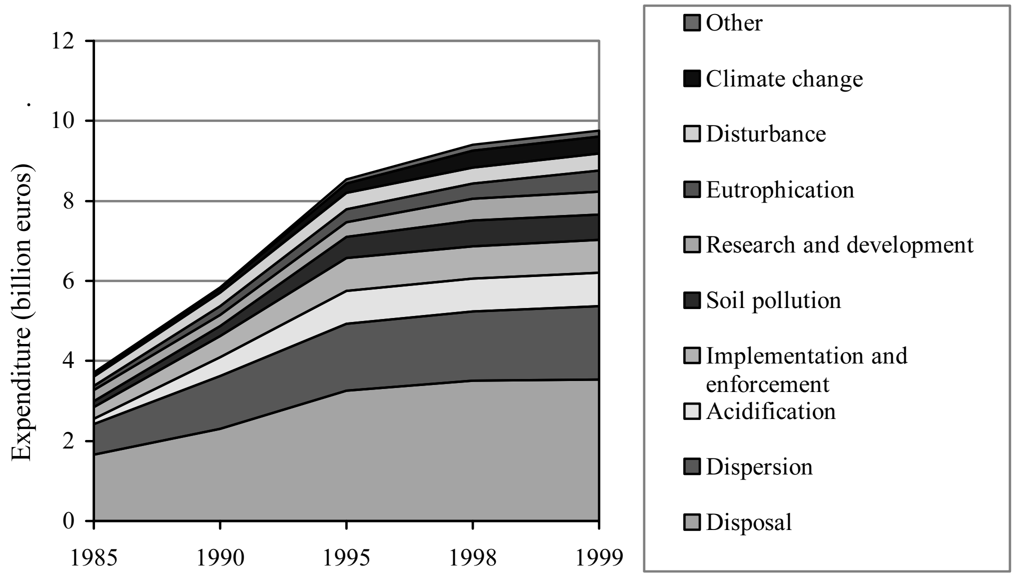

Figure 3-1 Environmental expenditures per theme

Source: RIVM (2001)

Since the 1990s, Dutch policy makers have attempted to stimulate the prevention and recycling of waste. They introduced laws to force municipalities to collect rest waste and organic waste separately. They also invested in recycling stimulation programs and passed regulations to prevent the landfilling of combustible waste. They also introduced a landfilling tax, which effectively raised the price of landfilling slightly above that of incineration. Due to these measures, the quantity of waste generated by industry declined sharply. For example, in the construction and demolition sector, the recycling rate went up to 94%. Households were less affected by these national policies. On average, industry recycles about 90% of its waste; households only recycle around 40%. The government still has a difficult task ahead of them to reduce the quantity of municipal solid waste generated in the Netherlands.

There are several options available to deal with waste, *i.e.* re-use, recycling, incineration, composting, and landfilling. Waste that is generated will either be collected separately or will be separated after collection. If possible, any useful materials, such as paper, metal and glass will be gathered and re-used or recycled. The remaining waste must be treated in the most efficient way. Organic waste may be composted; rest waste has to be incinerated or landfilled. In most discussions about waste treatment, recycling and re-use are not considered as waste treatment options, since these processes deal primarily with separating valuable materials from waste and reducing waste streams. Composting, incineration and landfilling on the other hand involve the disposal of these waste streams. In this chapter, the same distinction between waste handling options is adopted. Thus, when a reference is made to waste treatment, this will only refer to composting, incineration, and landfilling.

To compost or recycle waste, it is necessary to first sort the waste, preferably at the source. Since sorting at the source depends on the goodwill of the consumer who generates the waste, the option of mechanical separation after collection is attractive. This would solve the problem of uncooperative households. At the VAGRON facility, in Groningen, a new waste treatment method consisting of mechanical separation, anaerobic digestion and incineration of municipal solid waste is tested. This facility demonstrates that it is possible to recover about 40 to 50% of the incoming municipal solid waste for material recycling or re-use of secondary fuel. This reduces the quantity of municipal solid waste that has to be incinerated by 55%. This technique, however, is still quite expensive and in an experimental phase. Therefore, in the remainder of this thesis, the option of mechanical separation, anaerobic digestion and incineration will be excluded from the analysis. For additional information on this innovative waste treatment option see, for example, Oorthuys and Brinkmann (2000) and Oorthuys *et al.* (2002).

This chapter provides insight into the nature of waste flows in the Netherlands and the costs of waste treatment. To this end, a general overview of waste flows and waste treatment options is presented in Section 3.2 and Section 3.3 provides an overview of waste management policies. In Section 3.4, detailed information is given about the three waste treatment options: composting, incineration and landfilling. Finally Section 3.5 concludes.

3.2 A general overview of waste flows in the Netherlands

Waste generation in the Netherlands has risen sharply throughout the last couple of decades. The generation of waste tripled in the period 1960 to 1999. The growth rate of waste generation is quite similar to the growth rate of the national economy as is shown in Figure 3-2. Since the 1990s, the Dutch government has tried to change the relation between the two growth rates. As stipulated in the third national environment management plan (VROM, 1998a), the growth rate of waste generation should be 20% lower than the growth rate of the national income by the year 2010. Thus, although, waste generation will increase in the period until 2010, it should increase more slowly than the growth of the national income. In the first half of the nineties, policies seemed to be very effective, as waste generation appeared to stabilize. In the period 1994-1999, however, waste generation began to once again increase by about 2% per year (the average growth rate of the national income in this period was 3.8% per year). The total quantity of waste generated was roughly equal to 56.6 Mtonnes per year. As Chapter 2 illustrated, although the government succeeded in decoupling the generation of industrial waste and economic growth, it failed to achieve a decoupling between generation of municipal solid waste and economic growth.

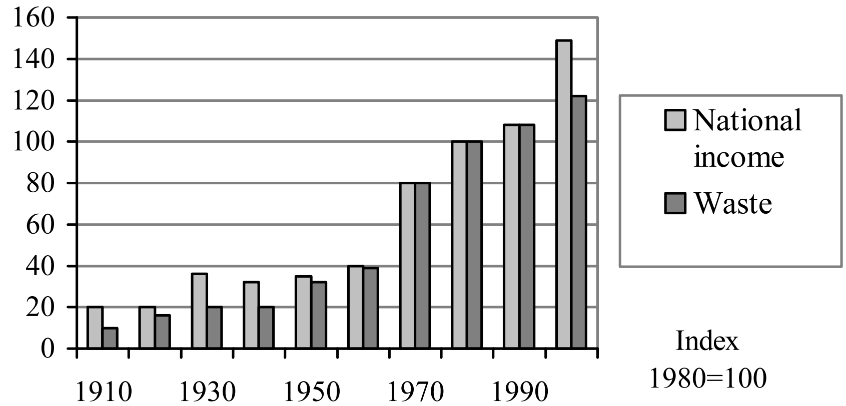

Figure 3-2 National income and waste production in the Netherlands, 1910-1999 (Index 1980=100)

Source: Adapted from VROM (1998b)

According to the Waste Management Council, the total waste stream can be divided into 6 categories, namely: (1) municipal solid waste, which is collected from households; (2) residues after sorting of municipal solid waste; (3) industrial waste; (4) construction and demolition waste; (5) contaminated soil and (6) a group of other types of waste. In Table 3-1, the quantity of waste generated per category is shown. Municipal solid waste forms the largest category, about 40%. The quantity of municipal solid waste generated has remained fairly constant over the last couple of years. Strikingly, the quantity of construction waste generated has more than halved in the past five years; the recycling percentage for this sector has increased sharply to 94%.

Table 3-1 Categories of waste treated in Mtonnes

Category	1998		1999		2000		2001		2002	
Municipal solid waste	4.9	(37%)	5.1	(37%)	5.1	(40%)	5.1	(40%)	5.0	(43%)
Residual	1.4	(11%)	1.2	(8%)	1.3	(10%)	1.4	(11%)	1.0	(9%)
Industrial waste	2.0	(15%)	2.3	(17%)	2.0	(16%)	1.9	(15%)	2.3	(20%)
Construction waste	1.2	(11%)	1.4	(10%)	1.0	(8%)	0.8	(6%)	0.5	(4%)
Contaminated soil	1.3	(10%)	1.3	(9%)	0.9	(7%)	0.5	(4%)	0.9	(8%)
Other	2.4	(18%)	2.5	(18%)	2.6	(20%)	3.0	(24%)	2.0	(17%)
Total	13.2	(100%)	13.8	(100%)	12.9	(100%)	12.7	(100%)	11.7	(100%)

Sources: WMC (2000b, 2003d)

3.2.1 The composition of the municipal solid waste stream

As shown in the previous section, municipal solid waste represents the largest category of waste generated in the Netherlands. Municipal solid waste is also quite diverse in nature. The municipal solid waste stream consists of several different waste materials as can be seen in Table 3-2[1].

Table 3-2 Collection of municipal solid waste (in Ktonnes)

	2000	2001	2002
Waste separately collected	4159	4146	4174
Organic waste	1457	1405	1416
Paper	1022	1015	993
Glass	326	335	340
Textiles	52	53	56
Small chemical waste	21	21	20
White and brown goods	43	53	59
Coarse garden waste	359	355	387
Furniture	10	14	23
Metals	76	77	70
Waste wood	225	246	262
Debris	451	437	399
Clean soil	76	90	106
Rest	41	45	43
Rest waste collected	4827	4859	4894
Total waste collected	8986	9005	9068

Source: WMC (2003a)

[1] Due to differences in categories, it is difficult to compare quantities presented in Table 3.1 and Table 3.2. Table 3.1 only presents the quantities of waste composted, incinerated, and landfilled. Table 3.2 presents both recyclable waste and waste to be composted, incinerated, and landfilled.

Part of this waste stream, like glass, paper, small chemical waste, and organic waste, is collected separately. The bulk of the municipal solid waste stream is collected as so-called rest waste, which is a waste stream that consists of mixed categories of waste materials. Given that it is costly and difficult to separate waste after collection, most of the rest waste is sent to an incinerator. The waste categories that are collected separately may be recycled or composted, depending on the waste stream in question.

About 46% of the municipal solid waste stream is collected separately and can thus be re-used, recycled or composted. This leaves about 54% of the waste stream, which has to be incinerated, since mixed waste is generally not recycled or composted. These percentages have remained fairly constant over the last couple of years. In Table 3-3 the composition of the rest waste stream is shown.

Table 3-3 The composition of municipal solid rest waste

Category	Composition in 2000 (%)	Composition in 2001 (%)	Composition in 2002 (%)
Organic waste	34	35	35
Paper	32	30	27
Plastics	13	13	13
Glass	3.9	4.2	4.2
Ferrous metals	3.6	3.9	4.5
Non-ferrous metals	0.79	0.83	0.83
Textiles	3.2	2.9	2.7
Small chemical waste	0.31	0.27	0.16
Rest	8	10	12
Total	100	100	100

Source: WMC (2003a)

Not all recyclable and organic waste is separately collected; part of it is thrown away as rest waste. The major part of the rest waste stream consists either of organic waste or paper. Recycling paper and composting organic waste is far less expensive than incineration; if households separated these waste types from rest waste then substantial waste treatment costs could be saved. In 2001, about 1255 Ktonnes of municipal solid waste was separated after collection, only about 191 Ktonnes of this waste has been turned into recycled materials. Given that the total quantity of municipal solid waste collected was equal to 9000 Ktonnes, only 2.2% of the waste stream was usefully recycled due to separation after collection.

In the Netherlands, several targets have been set for the recycling and composting of waste. By the end of 2006, households should separate: 55% organic waste, 75% paper, 90% glass, 50% textile, 90% ‘white’ and ‘brown’ goods[2], and 90% small

[2] ‘White’ and ‘brown’ goods are electrical and electronic devices. In 1999, The Netherlands passed a law that made producers of white and brown goods responsible for the life cycle of their product. They

chemical waste. These recycling and separation percentages differ a lot between various types of municipalities. Especially in the larger cities, households are less willing to separate waste and therefore the government has set less extensive goals in these municipalities. None of the large municipalities, however, have achieved results that are even close to these targets. Only the smaller municipalities are expected to reach their targets by the end of 2006 (WMC, 2003c).

3.3 Waste management policies

3.3.1 Waste management policies throughout the years

In the beginning of the 1960s, the Dutch government became concerned about environmental pollution. To control this pollution, legislation was introduced for each environmental problem, the so-called sector regulation. For the waste management problem, this led to the Hazardous Waste Law in 1976 and the Waste Material Law in 1977. Both of these laws attempted to regulate the treatment of waste.

Until the 1990s, waste management policies were mostly developed on a local and regional scale. Provinces had some coordinating tasks, but the collection and treatment of waste lay solely in the hands of the municipalities. In 1989, the government realized that to deal with the waste problem cost-effectively, waste management policies should be more centrally coordinated and developed on a provincial and national level.

In 1990, the Waste Management Council was established. This institution, which has representatives from municipalities, provinces and the national government, aims to develop an effective national waste management plan. In 1992, the waste management Council developed the ‘Ten year program for waste management’, which determined to necessary capacity for composting, incineration and landfilling for the period 1992-2002. In 1995, a second ‘Ten year program waste management’ program was developed for the years 1995-2005, which, apart from focusing on planning of necessary capacity, tackled the question of how the government could stimulate the treatment of waste according to the desired method (WMC, 1995). In 2002, the Waste Management Council published the ‘National waste management plan’, which gives guidelines on dealing with weak points in current waste management policies, the stimulation of more recycling, and further control of the environmental damage caused by waste treatment (WMC, 2003e).

are required to take back their products once they are discarded and have to ensure that the products are re-used and/or recycled in an environmentally sound manner.

In 1993, the Dutch government adopted the concept of waste hierarchy as the basis of national waste management policies. The waste hierarchy describes a strict order of preferences between different waste treatment options. Prevention is preferred above re-use; re-use above recycling; recycling above incineration and incineration above landfilling. First of all, national waste management policies aim to reduce waste flows as much as possible by promoting prevention and re-use of waste. The government tries to stimulate recycling and composting by forcing municipalities to collect organic waste and rest waste separately. Secondly, national waste management policies aim to promote incineration and composting instead of landfilling. To accomplish this, amongst other measures, a law was introduced in 1995 forbidding the landfilling of combustible waste. Another measure taken by the government in 2001 was the introduction of a landfill tax to make landfilling financially less attractive than incineration.

To reduce the environmental damage caused by waste treatment, the government passed several laws restricting the emissions of waste treatment units, for example the 'Dumping Law Soil Protection' and 'Emissions to Air Incineration Law'. Moreover, several very strict laws were implemented to regulate the quality of secondary materials produced by waste treatment units.

Although an extensive range of national waste management laws exist, European laws have become increasingly important in the waste sector. These laws will be discussed in the next section.

3.3.2 European waste management law

In recent years, the waste market has become increasingly internationally orientated. Prior to 1994, waste management was mostly a national affair. Waste could not be exported, since this would place an unfair burden on the environment of the importing country. Since 1994, however, European countries have been allowed to export recyclable waste. It is expected that within a couple of years, the borders will also be thrown open for the export of combustible waste.

Major changes can be expected in the waste market in the next couple of years, due to both the European Energy policy and the European waste policy.

1) *The European Energy policy*. The European Energy policy sets ambitious targets for increasing the use of renewable energies and especially the use of bio-fuels. By the end of 2010, 22.1% of the entire European energy use should come from renewable sources. In 1997, the percentage use of renewable energy was only 13.9%. This target is apportioned between the member states. For the Netherlands, the target is 9% by the end of 2010.

2) European waste policy. The sixth European Environmental Action program 2002-2012 formulates several activities to improve waste management at the European level. These activities are mostly aimed at increasing recycling and re-use, and reducing the environmental damages of waste treatment. Further more, the program stipulates harmonization of the waste markets of its member states[3] (WMC 2003b).

3.3.3 Municipalities and waste collection

Municipalities are responsible for the collection and treatment of municipal solid waste. They are free to determine how much they want to charge households for providing these services. Waste fees differ a lot between municipalities. The average waste fee per household has continuously risen since 1991, as shown in Figure 3-3 .

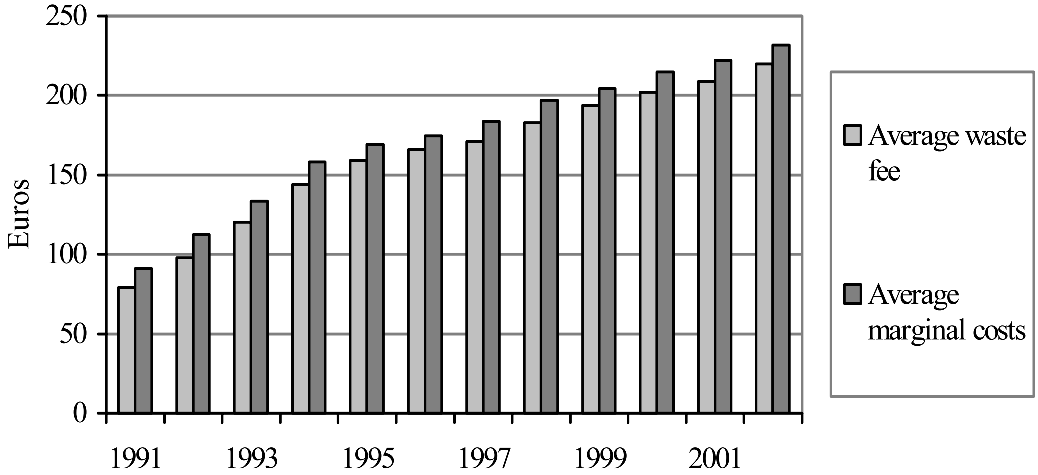

Figure 3-3 Average waste fees per household and average marginal costs of collecting and treating municipal solid waste in the Netherlands

Sources: WMC (2000a, 2002)

The average waste fee was equal to 220 Euros per household per year in 2002. Compared to 1991, this is an increase of more than 260 percent. Especially in the years between 1991 and 1995, the average waste fee increased sharply due to changes in national waste policies. In these years, the municipalities switched from landfilling waste, which was relatively cheap, to incinerating it, which was far more expensive. Furthermore, they also began to collect organic waste separately from rest waste.

[3] See also Bucklet and Goddard (2001) for additional information about European waste policies and a comparison between waste management policies in various European countries.

Since 1995, the average fee has still increased, but at a lower rate of around 5.2% per year (WMC, 2002).

The waste fee does not cover all the costs that the municipalities make by collecting and treating waste. The average cost coverage rate has steadily risen throughout the last decade from about 87% in 1991 to 94% in 2002. The cost coverage rate differs considerably between municipalities. About 70% of all municipalities have a 100% cost coverage rate. Three municipalities, namely Eesmond, Leiden and Nijmegen choose to charge no waste fee whatsoever so they have a cost coverage rate of zero.

Unit-based pricing

As mentioned earlier, municipalities are free to decide how much they want to charge households for collection of waste. Many municipalities choose to make no distinction between households based on the quantity of waste they generate. The level of the fee is independent of the quantity of waste that is actually generated. Most municipalities do differentiate the waste fee according to the number of persons living in the household. Some municipalities, however, are currently experimenting with the introduction of some sort of price differentiation based on the quantity of municipal solid waste. In 2002, about one out of four municipalities have implemented a unit-based pricing scheme for the collection of waste. There are several different types of unit-based pricing schemes, *i.e.* according to expensive bags, volume, frequency, weight, or a combination of these types. Table 3-4 shows the distribution of the different types of unit-based pricing schemes over the municipalities in 2001.

Table 3-4 Distribution of different forms of unit-based pricing in 2001

System	Percentage municipalities	Percentage households	Average number of households per municipality
Expensive bag	2.4	2.9	16,369
Expensive bag & size household	1.6	0.8	6,480
Expensive bag & frequency & volume	0.4	0.3	8,938
Volume	6.7	4.8	9,532
Volume & frequency	10.5	6.3	7,946
Weight	4.0	2.4	8,028
Weight & frequency	0.6	0.6	12,967
Other	1.0	0.8	11,397
Total	27.2	18.9	9,093
Size household	61	58	12,721
No differentiation	12	23	25,565

Source: WMC (2002)

Most municipalities, which have introduced unit-based pricing, favor a unit-based price according to either volume and/or frequency. An expensive bag system, in which households need to buy special waste disposal bags, is favored in larger municipalities, primarily due to the lower implementation costs. Of those municipalities that charge a flat fee for waste collection, a majority, *i.e.* 61%, do differentiate between household size. Only 12% of the municipalities apply no price differentiation at all.

Most of the municipalities that introduced unit-based pricing are rather small. This is understandable since unit-based pricing will be much more successful in municipalities with a relatively low share of apartment buildings. In larger municipalities, where the percentage of people living in apartment buildings is higher, unit-based pricing will be less successful given that households have less space available to compost or sort waste (Ando and Gosselin, 2003). Moreover, it will also be more difficult and costly to implement a unit-based pricing system for households living in apartment buildings than for households living in single-family houses. For example, to implement unit-based pricing for single-family houses, the municipality can choose to install weighing scales in the garbage trucks to keep track of the quantity of waste that a household generates. In apartment buildings, waste is normally not collected separately for each household. To introduce unit-based pricing, the waste collection method has to be changed completely. Some municipalities have built underground containers, which can only be accessed using a personal key, to collect waste from households living in apartment buildings. This is, of course, far more expensive to implement than simply modifying a garbage truck.

Table 3-5 Average fee for different unit-based pricing schemes

	Average fee (Euro/household/year)				Average waste management costs (Euro/household/year)			
System	1999	2000	2001	2002	1999	2000	2001	2002
Volume	200	199	215	224	209	205	220	229
Volume + frequency	179	186	228	199	185	191	227	204
Expensive bag	93	109	121	126	153	155	179	180
Expensive bag & persons	-	-	161	177	-	-	168	190
Expensive bag & volume & frequency	-	-	-	203	-	-	-	203
Weight	169	179	194	215	188	186	200	221
Weight & frequency	-	-	-	202	-	-	-	208
Rest	195	186	201	256	211	187	202	256
Average differentiated fee depending on waste offered	170	170	186	196	190	184	199	208
Size household	188	197	207	217	198	205	221	230
No differentiation	188	200	197	206	201	210	208	219

Source: WMC (2002)

On average, the fee in municipalities that introduced unit-based pricing is 12% lower than in municipalities without a unit based pricing scheme as shown in Table 3-5. The percentage of cost covered, however, is also lower. Thus households pay a lower fee for collection of waste, while the municipalities pay more. Especially in the case of price differentiation on the basis of expensive bags, the average fee covers only about half the costs.

The effects of unit-based pricing

The introduction of a unit-based pricing scheme seems to reduce the quantity of municipal solid waste generated. As shown in Table 3-6, the generation of rest waste declines by 13% in cases of differentiation based on volume, to 59%, in case of differentiation by expensive bags. Households begin to separate far more waste, indeed about 10-25% more.

Table 3-6 shows that in a system with price differentiation on the basis of volume, the total quantity of waste generated is not affected. In the other unit-based pricing systems, the total quantity of waste generated declines. The total quantity of organic waste generated requires some clarification. In the system with price differentiation based on weight, households pay a variable price for the collection of rest waste as well as organic waste. These households therefore have an incentive to reduce generation of organic waste as much of possible. In these municipalities, the quantity of organic waste that is home composted increases sharply. In the other unit-based pricing systems, consumers do not pay or pay less for the collection of organic waste, thus they have an incentive to increase their separation of organic waste from rest waste, but they have fewer or no incentives to start home composting.

Table 3-6 Differences in waste generated per household in 1997 for different unit-based pricing systems

System	Rest waste	Bulky waste	Organic waste	Paper glass textiles	Other separated waste	Total
Without unit-based pricing	0%	0%	0%	0%	0%	0%
Volume	-13%	-19%	28%	21%	10%	1%
Volume & frequency	-41%	-32%	1%	41%	34%	-15%
Expensive bag	-59%	-26%	16%	36%	17%	-22%
Weight	-51%	-29%	-25%	46%	31%	-23%

Source: KPMG (1999)

The figures presented in Table 3-6 suggest that unit-based pricing is quite effective in the promotion of waste separation and recycling. However, a few words of caution are in order. First of all, it is not exactly clear whether a straight relation between unit-based pricing and waste separation exists. Unit-based pricing so far has only been introduced in combination with efforts to raise the public awareness for the social costs of waste treatment and for the benefits of recycling. Hence it is not possible to separate the effects of unit-based pricing and aggressive recycling programs. Secondly, thus far unit-based pricing has only been introduced in small municipalities, with a relatively small number of households per square mile. It is expected that the results would be far less positive if unit-based pricing was introduced in municipalities with a higher number of households per square mile (KPMG, 1999).

3.4 Waste treatment options

In 2002, about 54.0 Mtonnes of waste was generated. Of this amount, about 78.5% was re-used or recycled, which left about 11.6 Mtonnes of waste to be composted, incinerated, or landfilled. Due to the aforementioned government regulations, the percentage of waste landfilled has declined sharply over the last couple of years and the percentage of waste incinerated has increased sharply. The quantity of waste composted, incinerated, and landfilled over the last seven years are presented in Table 3-7.

Table 3-7 Quantity of waste treated per waste treatment option in Ktonnes

	Quantity of waste (in Ktonnes)						
Treatment option	1996	1997	1998	1999	2000	2001	2002
Landfilling	8,450	7,400	7,100	7,600	6,550	6,530	5,157
Incineration	3,550	4,350	4,649	4,810	4,898	4,770	5,006
Composting	1,500	1,500	1,515	1,490	1,568	1,448	1,444
Total	13,500	13,250	13,264	13,900	13,016	12,748	11,607

Source: WMC (2003d)

The amounts of waste treated per category are shown in Table 3-8.

Table 3-8 Types of waste treated per waste treatment option (in Mtonnes)

	Composting			Incineration			Landfilling		
Waste category	2000	2001	2002	2000	2001	2002	2000	2001	2002
Municipal solid waste	1.5	1.4	1.4	2.7	2.8	3.0	0.9	0.8	0.6
Residual	-	-	-	0.7	0.7	0.7	0.6	0.7	0.3
Industrial waste	-	-	-	1.0	1.0	1.2	1.0	1.0	1.1
Construction waste	-	-	-	-	-	-	1.0	0.8	0.5
Contaminated soil	-	-	-	-	-	-	0.9	0.5	0.9
Other	-	-	-	0.5	0.2	0.1	2.1	2.7	1.9
Total	1.5	1.4	1.4	4.9	4.7	5.0	6.5	6.5	5.3

Source: WMC (2003d)

The chosen waste treatment option varies considerably between waste categories. Solid waste is mostly incinerated or composted; industrial waste, construction waste, and contaminated soil are still generally landfilled. Strikingly, almost 60% of all the waste incinerated comes from households.

3.4.1 Composting

In the next three sections, several aspects, such as capacity, financial costs, and social costs are discussed per waste treatment method. First of all, the various aspects of composting will be examined.

Waste flows and composting

In 2002, a total of 1837 Ktonnes of organic waste was composted. Of this amount, 1444 Ktonnes came from households (see Table 3-8), and 152 Ktonnes from other sources like the agricultural and forestry sector and small businesses and shops. The remaining 241 Ktonnes derived from the separation of municipal solid waste after collection. It is important to note that although this waste has been treated in a composting unit, no high quality compost could be made from this waste stream. Thus the compost coming from this source has not been sold, but instead incinerated.

In Figure 3-4, the quantity of organic waste generated by households throughout the years is shown.

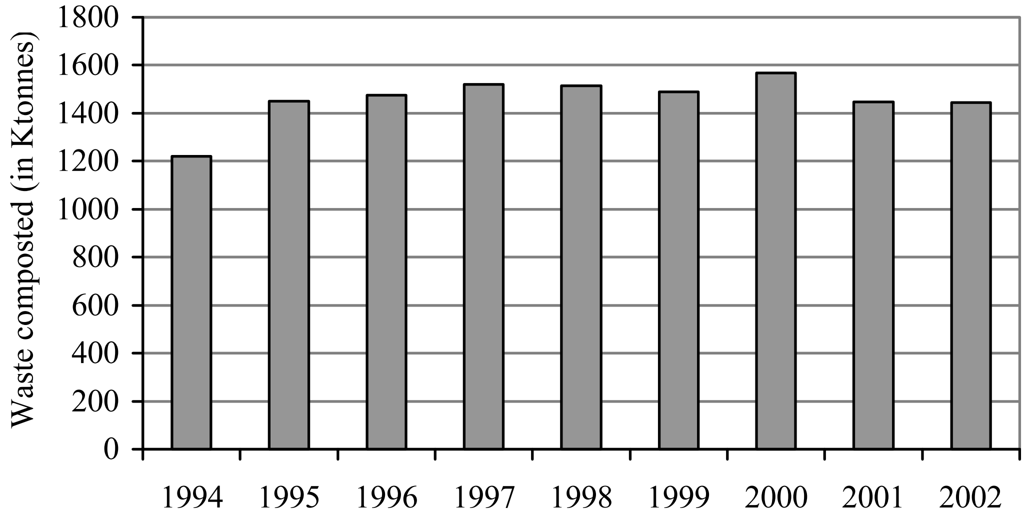

Figure 3-4 Quantity of municipal solid waste composted in the Netherlands in Ktonnes

Sources: WMC (2000b, 2003d)

In 1995, the quantity of municipal solid waste composted increased significantly due to the introduction of the Anti-dumping Law, which made it impossible to landfill waste that could be incinerated or composted. At the same time, the government stimulated more composting by forcing municipalities to collect organic waste and rest waste separately. After 1995, the quantity of waste composted stabilized at around 1.5 millions tonnes of municipal solid waste per year.

Capacity of composting units in the Netherlands

Composting units can be found in almost all provinces in the Netherlands. The average quantity of waste handled by a composting unit is about 60 Ktonnes. Table 3-9 shows how much waste is composted in each region in the Netherlands.

Table 3-9 Quantity of municipal solid waste composted per province

	Quantity of waste composted (Ktonnes)				
Province	1998	1999	2000	2001	2002
Groningen	36	37	37	38	39
Friesland	0.2	14	30	19	13
Drenthe	335	339	346	328	307
Overijjsel	67	67	69	65	66
Gelderland	212	208	222	212	221
Flevoland	37	33	30	36	28
Utrecht	-	-	-	-	-
Noord-Holland	168	160	162	150	161
Zuid-Holland	229	208	238	220	203
Zeeland	48	50	42	48	45
Noord-Brabant	247	232	235	217	212
Limburg	138	142	148	126	148
Total	1517.2	1490	1559	1459	1444

Sources: WMC (2000b, 2003d)

Note: the total quantity of waste composted does not exactly correspond with the total quantity composted as reported in Table 3-7 due to some rounding off of numbers.

Most of the composting units in the Netherlands can only treat a relatively small quantity of waste. There are only two exceptions, namely Essent Milieu Wijster Compostering in Drenthe, which treated 307 Ktonnes of waste in 2002, and VCB in Gelderland, which treated 179 Ktonnes of waste. Strikingly, most of the composting capacity is situated in the north of the Netherlands, in the provinces of Groningen, Friesland, Drenthe and Overijssel.

Financial and social costs of composting

The price of composting varies considerably between composting units. Firstly, the price depends on the size of the composting unit; larger composting units are about 23% cheaper than smaller units due to economies of scale (Oorthuys, 1995). Secondly, the price also depends on the technology used by the composting units, for example, whether the process is based on anaerobic digestion or aerobic digestion (WMC, 2003e). In 2002, the average price of composting in the Netherlands was equal to 60 Euro per tonne of waste.

The costs of composting can be divided in capital costs, labor costs, and costs for disposal of residue. Capital costs, which consist of depreciation costs and interest costs, take up most of the costs, about 60% to 70%. The remaining costs are incurred by personnel and maintenance, about 20%, and disposal of residue, about 10% to 20% (WMC, 2003e). Besides financial costs, composting also creates emissions into the air and water. In Table 3-10 the major emissions to air and water are illustrated.

Table 3-10 Emissions caused by composting

Type	Direct emissions in kg per tonne of waste
Air:	
CH_4	2.400
NH_3	0.200
N_2O	0.096
NO_X	0.016
Water:	
CZV	0.127
BZV	0.030
N	0.032
Anorg. rest	1.140
Cl	0.090
Mg	0.010

Source: WMC (2003e)

Sales of compost

In 2002, about 569 Ktonnes of compost were sold. Table 3-11 shows how much compost is sold to which sector. Most of the compost is sold to the agricultural sector, almost 50%. The quality of the compost is quite good, due to strict regulations of the government; this is the primary reason for the popularity of compost in this sector. Almost all compost that is produced is also sold.

Table 3-11 Quantity of compost sold to sectors

	Quantity of compost sold (in Ktonnes)				
Sector	1998	1999	2000	2001	2002
Agriculture	230	297	309	154	107
Gardening sector	-	68	66	52	119
Recreation	30	32	23	7	43
Private sector	13	32	5	16	29
Distributive trades	109	42	78	248	126
Municipalities	19	30	14	6	5
Rest	69	109	81	164	138
Total	470	610	576	647	567

Source: WMC (2000b, 2003d)

Since the laws concerning the quality of compost are quite strict, composting units are reluctant to accept organic waste polluted with other types of waste, such as metals and plastic. If the organic waste stream is too polluted then the composting units will refuse to compost the waste, because even if the organic waste is cleaned and the metals and plastics are removed it will still result in an inferior type of compost. IPH (1995) demonstrated that on average organic waste was only 95% pure. Unfortunately, no research has recently been conducted to determine the quality of organic waste. Especially, when unit-based pricing is introduced, the quality of the organic waste stream may possibly decline.

3.4.2 Incineration

The second waste treatment option discussed here is incineration. In the Netherlands, a relatively large percentage of waste is incinerated in comparison to other western countries. Incineration is considered to be preferable to landfilling as it first of all does not require "eternal aftercare", a disadvantage of landfill sites and secondly it produces electricity and heat as a by-product. This electricity will be partly re-used in the incineration process and partly sold to electricity providers. In some cases, the energy produced is used to heat nearby houses or companies.

Waste flows and incineration

About 5000 Ktonnes of waste were incinerated in 2002. Since the adoption of the waste hierarchy as basis of the national waste management policy, the quantity of waste incinerated has steadily increased. Figure 3-5 illustrates that the quantity of waste incinerated has nearly doubled over the last 10 years.

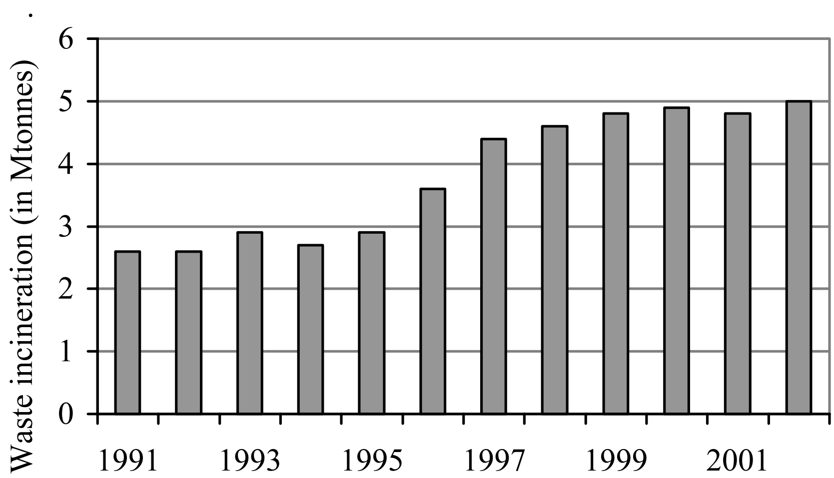

Figure 3-5 Total quantity of waste incinerated each year

Sources: WMC (2000b, 2003d)

In Table 3-12, the categories of combustible waste are shown. Most of the incinerated waste comes from households, nearly 60 percent. All rest waste collected from households should, in theory, be incinerated. As the capacity of incinerators has not always been sufficient to treat all municipal solid waste, some of this waste stream has been landfilled. The capacity of the incineration plants, however, has increased and thus, as shown in Table 3-12, the quantity of municipal solid waste incinerated has grown throughout the last five years. Industrial waste was mostly landfilled before 2000. Since 2000, when a landfill tax that raised the price of landfilling above the price of incineration was introduced, the quantity of industrial waste incinerated has increased steadily.

Table 3-12 The total quantity of incinerated waste per category (in Ktonnes)

	Quantity of waste incinerated (in Ktonnes)				
Waste category	1998	1999	2000	2001	2002
Solid waste	2614	2848	2710	2814	2987
Residue of solid waste and industrial waste after central sorting	890	722	693	676	717
Industrial waste	769	891	988	977	1203
Rest	374	363	505	307	102
Total	4647	4824	4896	4774	5009

Sources: WMC (2000b, 2003d)

Note: the total quantity of waste incinerated does not exactly correspond with the total quantity incinerated as reported in Table 3-7 due to some rounding off of numbers.

Capacity of incineration plants

There are 12 waste incineration plants in the Netherlands. Together they have a capacity to treat about 5.5 million tonnes of waste.

Figure 3-6 The location of incineration plants in the Netherlands

The location of waste incineration plants is illustrated in Figure 3-6. Most of the capacity for waste incineration can be found in the west and the north of the Netherlands. Landfilling was quite expensive in the north because the soil was unsuitable for landfilling and in the west due to a shortage of available land. In the south, both enough space was available and the soil was suitable for building landfill sites. In the south, therefore, landfilling was considered to be the optimal waste treatment option. Only during the last five years, following the government's prohibition of the landfilling of combustible waste did the south start to invest in incineration plants.

The total quantity of waste treated in each installation is shown in Table 3-13.

Table 3-13 Total quantity of waste treated in incineration plants

		Quantity of waste incinerated (in Ktonnes)				
Province	Installation	1998	1999	2000	2001	2002
Drenthe	GAVI-Wijster	413	433	441	425	422
Overijssel	AVI Twente	288	284	284	290	289
Gelderland	ARN	237	250	239	245	269
	AVIRA	297	287	315	336	339
Noord-	Huisvuilcentrale NH	450	450	448	460	464
Holland	AVI Amsterdam	790	761	801	795	827
Zuid-	AVR rijnmond	975	1040	1098	1086	1120
Holland	GEVUDO	194	171	215	212	207
	AVR Afvalverwerking Rotterdam	385	386	391	375	383
	ZAVIN	7	7	7	7	7
Noord-	SITA Roosendaal	49	55	54	50	52
Brabant	AZN	561	603	605	489	628
Total quantity of waste incinerated		4649	4810	4898	4770	5006

Sources: WMC (2000b, 2003d)

Note: the total quantity of waste incinerated does not exactly correspond with the total quantity incinerated as reported in Table 3-7 and Table 3-12 due to some rounding off of numbers.

Since the opening of the provincial borders for transport of waste in January 2000, waste incineration plants are able to accept waste from all over the country. The operational scale of the incineration plants has gone from regional to national. Moreover, several waste incineration plants (Gavi-Wijster, ARN and AVI-Twente) are planning to focus more on treatment of foreign waste (WMC, 1997).

Financial and social costs of incineration

The average cost price for incineration in 2002 was equal to 110 Euros per tonnes of waste. Figure 3-7 shows the cost price for incineration plants in 2002.

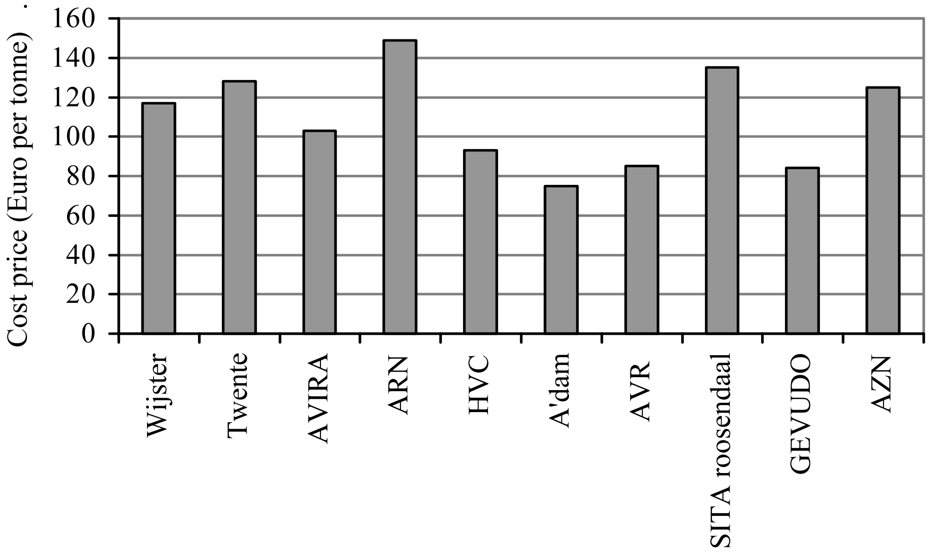

Figure 3-7 The price of waste incineration in 2002 for the incineration plants (in Euro per tonnes)
Source: WMC (2003b)

Waste incinerator ARN in Nijmegen was clearly the most expensive company, and HVC in Amsterdam the cheapest. The price of incineration depends on the size of the incineration plant. On average, a large incineration plant is about 41% less expensive than a small incineration plant due to economies of scale.

The cost price of incineration is determined by a total of capital costs, operational costs and operational benefits. Figure 3-8 illustrates the extent to which capital costs, operational cost, and benefits varied for each installation in 1996.

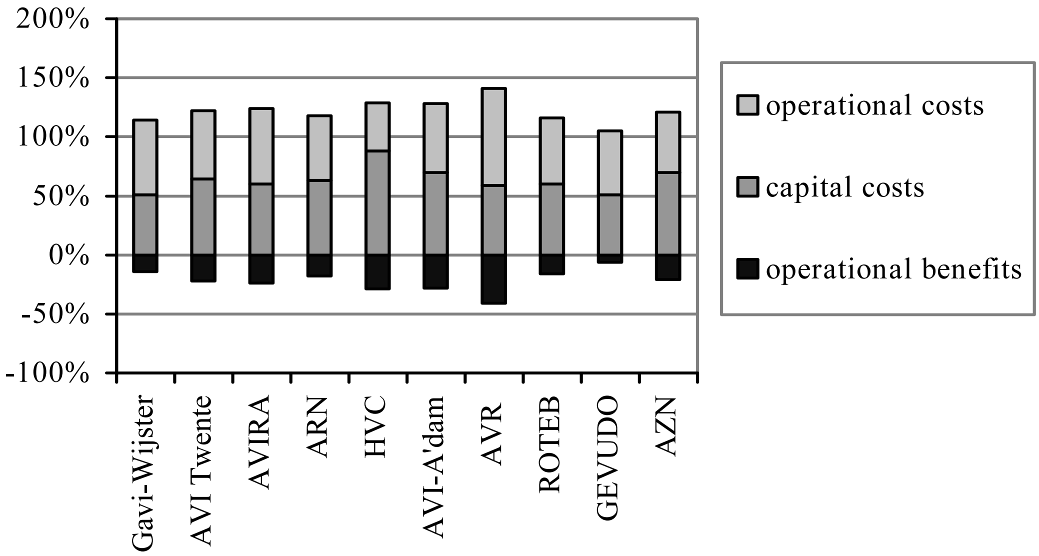

Figure 3-8 Composition of cost price of incineration of waste

Source: WMC (1997)

The greater the operational benefits, the lower the cost price will be. The capital costs consist of depreciation costs and interest costs. The operational costs are defined as the maintenance costs, costs of landfilling, the residues of incineration, energy costs, cost of personnel and other operating costs. The operational benefits consist of the sales of energy and the sales of metal and others useful residues. The large variation of benefits between the different incinerators is particularly interesting. AVR had the highest percentage of benefits, GEVUDO the lowest.

The incineration of waste creates emissions into the air and groundwater. In 1989, there was great commotion about the generation of dioxins by incinerators. Incinerators barely filtered dioxins from the smoke they emitted and this turned out to be a serious health hazard. Toady, incinerators filter almost all dioxins from emitted smoke; however, some other substances are still emitted. In Table 3-14, the most important emissions to air and water are shown.

Table 3-14 Emissions caused by incineration (per tonne of waste)

	Direct emissions per tonne of waste
Air (in kg)	
CO_2	467
CH_4	0.03
SO_2	0.20
NO_x	0.21
Dioxins (in mg)	0.00255
Water (in mg)	
Cd	5.2
Cr	4.5
Cu	5.5
Ni	2.1
Pb	1.2
Hg	0.72
Chemical waste (in kg)	21.80

Sources: WMC (2003e) and CE (1996)

Municipalities and Incinerators

In the Netherlands, municipalities and incinerators are inextricably linked. Contracts between municipalities determine the quantity of waste municipalities to be delivered to the incinerator and the price paid for waste treatment. Most of the waste treated in incinerators comes from municipalities. Figure 3-9 shows the percentages of industrial and municipal solid waste treated in incinerators in 2002. The average percentage of solid waste treated in incinerators is equal to 74 percent. Essent Wijster in Drenthe and ARN in Gelderland only treat solid waste from households. AZN in Noord-Brabant treats the largest percentage of industrial waste, 47% of all waste treated in AZN comes from industrial sources.

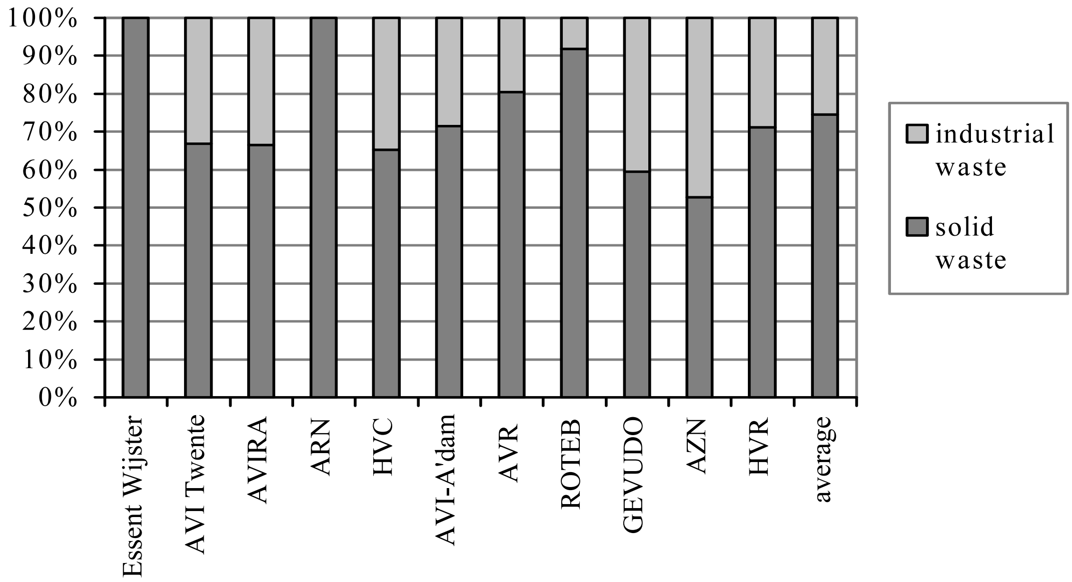

Figure 3-9 Percentage of solid and industrial waste treated in incinerators

Source: WMC (2003d)

Since almost all municipalities in the Netherlands have contracts with incinerators, the supply of municipal solid waste is more or less guaranteed. These contracts were introduced due to the necessity of promoting incineration. The starting cost of building an incineration plant were quite high and the market for incineration was rather unstable. Therefore, it was difficult to get companies interested in building an incinerator without some form of insurance. The contracts between municipalities and incinerators gave incinerators a right of existence. These contracts could be used as collateral to finance the investment costs of an incinerator. Moreover, some municipalities provided guarantees for financial obligations or possible losses. Due to these guarantees, financing the building and operation of incineration plants became less risky and the capital costs were reduced significantly (WMC, 1999).

However, the downside of the contracts and financial guarantees is that it is fairly easy for the incinerators to transfer operational risks to the municipalities. Thus the risk of operation lies partly or completely with the municipalities and therefore with the households, who pay for waste incineration through waste fees (WMC, 1999 and Dijkgraaf *et al.*, 1999).

3.4.3 Landfilling

The final waste treatment method under discussion is landfilling. Landfilling encompasses the controlled dumping of waste in specific sites and is in fact the least preferred method of waste treatment according to the waste hierarchy, as discussed in Chapter 2. Since landfilling does not fit into the concept of closed material cycles, the government does not approve of landfilling and has striven to discourage it as much as possible. One should, however, bear in mind that is not yet possible to eliminate landfilling completely as a means of dealing with waste. Both incineration and composting, for example, will produce a residue, which has to be landfilled. Moreover, the capacity of incinerators is as of yet not large enough to handle all combustible waste, thus part of this waste will inevitably have to be landfilled.

Waste flows and landfilling

Most of the waste treated in landfill sites stems from the industrial sector and the construction and demolition sector. The percentage of solid waste that is landfilled has declined sharply due to policies of the government, as can be seen in Table 3-15.

Table 3-15 Quantity of waste landfilled per waste category

	Quantity of waste in Ktonnes				
Waste category	1998	1999	2000	2001	2002
Solid waste	800	800	950	800	592
Rest waste after sorting	500	550	600	650	329
Industrial waste	1200	1450	1000	1050	1054
Cleansing department waste	150	150	250	300	96
Shredder waste	150	100	100	150	131
Construction and demolition waste	1200	1400	1000	800	461
Contaminated soil	1250	1250	900	500	854
Non-contaminated soil	200	300	250	350	103
Purification silt	350	250	150	200	21
Rest	1300	1350	1400	1800	1516
Total	7100	7600	6600	6600	5157

Sources: WMC (2000b, 2003d)

Note: the total quantity of waste landfilled does not exactly correspond with the total quantity landfilled as reported in Table 3-7 due to some rounding off of numbers.

The Anti-dumping Law does not permit the landfilling of combustible waste. Since the capacity of incineration plants was insufficient, municipalities were occasionally exempted from this regulation. In 2003, however, the government decided that the incineration capacity was sufficient to treat all municipal solid waste and thus, as of 2003, municipalities will no longer be able to be exempted from the Anti-dumping Law. (WMC, 2003e).

Capacity of landfill sites

As shown in Figure 3-10, the quantity of waste landfilled has declined sharply throughout the past decade. Figure 3-10 illustrates that the quantity of waste landfilled dropped sharply in 1995 due to the introduction of the afore mentioned Anti-dumping Law. In 1999, the quantity of waste landfilled rose slightly. This is partly due to the low costs of landfilling as compared to other treatment options.

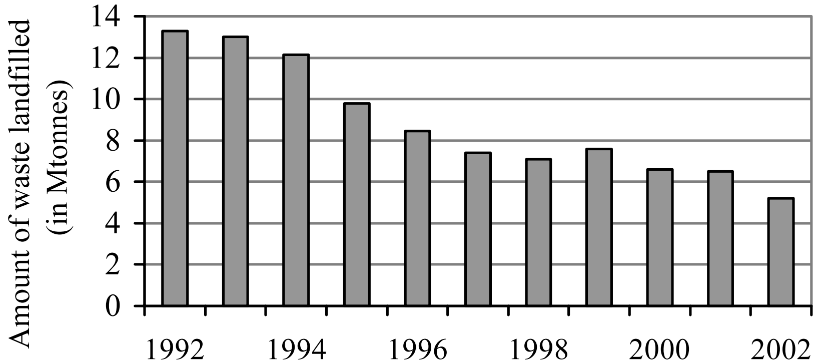

Figure 3-10 Quantity of waste landfilled in Mtonnes
Sources: WMC (2000b, 2003d)

Although in theory waste cannot be landfilled due to the Anti-dumping Law, it is still possible to get an exemption. Industries in particular are very keen on being awarded an exemption, since the price of landfilling is much lower than the price of incineration. In 2000, the government introduced a substantial tax on landfilling to raise the price of landfilling above that of incineration; thus providing industries with an incentive to incinerate and recycle waste.

Not every province landfills the same quantity of waste. Table 3-16 shows the quantity of waste landfilled in each province.

Table 3-16 The quantity of waste landfilled per province in the Netherlands

	Quantity of waste landfilled (Mtonnes)			Rest capacity (Mtonnes)			New capacity (Mtonnes)		
Province	2000	2001	2002	2000	2001	2002	2000	2001	2002
Groningen	0.23	0.31	0.26	1.5	1.3	1.2	-	-	-
Friesland	0.29	0.26	0.31	0.5	2.1	1.9	1.2	-	-
Drenthe	0.54	0.64	0.42	5.1	5.8	5.4	-	-	-
Overijjsel	0.49	0.34	0.35	5.4	5.2	5.3	3.7	3.7	3.7
Gelderland	1.1	1.19	1.1	8.8	8.6	8.1	4.6	4.6	4.6
Flevoland	0.09	0.09	0.15	0.5	0.6	0.4	4.2	4.2	4.2
Utrecht	0.18	0.18	0.15	1	3.3	2.7	2.6	-	-
Noord-Holland	1.01	0.99	0.78	6.2	5.3	4.9	3.7	3.7	3.7
Zuid-Holland	0.92	0.89	0.58	8.1	6.4	6	-	-	-
Zeeland	0.3	0.25	0.24	1	2.5	2.3	1.5	-	-
Noord-Brabant	0.94	0.99	0.54	10.8	8.6	9.4	-	-	0.7
Limburg	0.46	0.4	0.27	9.5	6.8	6.6	0.4	-	-
Total	6.55	6.53	5.15	58.4	56.5	54.2	21.9	16.2	16.9

Source: WMC (2003d)

Note: the total quantity of waste landfilled does not exactly correspond with the total quantity landfilled as reported in Table 3-8 and Table 3-15 due to some rounding off of numbers.

In 2002, there were 30 landfilling sites in use, 17 landfilling sites in the process of being closed, 3 new sites being built, and 4 sites temporarily closed. The total capacity per province is also given in Table 3-16. Strikingly, most of the landfill capacity is situated in the south of the Netherlands. As explained above, landfilling was attractive in the south due to availability of space. Only after it became evident that the government was going to discourage landfilling, did the south start concentrating on the incineration of waste. This is in contrast to the north of the Netherlands, where the soil was not suitable, and the west of the Netherlands, where space was too valuable.

Financial and social costs of landfilling

The price of landfilling has significantly increased over the last twenty years. In Table 3-17 the average prices of landfilling are provided. These average prices include the tax on landfilling. The tax on landfilling of combustible waste constitutes about 62% of the total costs of landfilling. The tax on landfilling of non-combustible waste is about 13% of the total costs. As can be seen in Table 3-17, since 2000, the average price of landfilling including the tax has been higher than the average price of incineration.

Table 3-17 Average price of landfilling and incineration (Euro per tonne)

	1985	1990	1995	1998	2000	2001	2002
Landfilling of combustible waste	10	27	78	93	110	107	128
Landfilling of non-combustible waste	10	27	48	63	60	56	58
Incineration	45	64	101	95	101	99	106

Source: RIVM (2003)

The operational costs of landfilling are divided as follows: about 15% of the costs are spent on personnel and maintenance; about 10% of the costs are spent on the aftercare tax and the remainder, about 75%, is spent on capital costs, which consist of depreciation costs and interest costs (Statistics Netherlands, 2002a). Landfill sites require eternal aftercare to prevent leakages to ground water at some point in the future. In 1996, the provinces were made responsible for taking care of the eternal aftercare for landfill sites within their borders. The provinces thus decided to tax landfill sites. The average aftercare tax is currently about 5 Euros per tonne of waste.

Landfilling can lead to emissions into both air and groundwater. It is inevitable that some emissions will occur, even if the best available techniques to prevent leakage are employed. Some biogas can be won back from the waste, thus reducing the energy costs of landfilling.

The process of landfilling works as follows: firstly, waste will be dumped in a landfilling site. Then it is covered up. For about 15 years the landfill site will be exploited in the sense that gas produced in the landfill site is captured and used to produce electricity. After 15 years, the landfill site will be closed and remaining gas will flared. Table 3-18 shows the emissions to air and water that occur during the landfilling process. Most of the emissions occur during the dumping process of waste; they are known as direct emissions. During the process of electricity generation from the landfilling gas, (called gas motor in the table) and during the process of flaring the remaining landfill gas, some emissions to NO_x will occur.

Table 3-18 Emissions and chemical waste caused by landfilling (per tonne of waste)

	Emissions to air, water and production of chemical waste per tonne waste			
Emissions	Direct	Flaring	Gas motor	Total
Air (in kg)				
CO_2	0	0	0	0
CH_4	10.44	0	0	10.44
SO_2	0	0	0	0
NO_x	0.015	0.01	0.23	0.255
Water (in mg)				
As	4.8	0	0	4.8
Cd	0.144	0	0	0.144
Cr	9.0	0	0	9.0
Cu	2.4	0	0	2.4
Ni	6.0	0	0	6.0
Pb	2.7	0	0	2.7
Hg	0.9	0	0	0.9
Chemical waste (in kg)	2.0	0	0	2.0

Source: CE (1996)

3.5 Concluding remarks

The waste market in the Netherlands is quite well documented. The Waste Management Council collects a lot of data each year. There have been several significant changes in the waste market during the last 10 years. Due to governmental regulations, far more waste is now being incinerated or composted instead of landfilled. Recycling percentages of waste have also increased. On average, about 79% of all waste generated in the Netherlands is recycled. These high recycling percentages are mostly due to the high recycling percentages in the industrial sectors. The industrial sectors recycle on average about 90%. Households do not recycle nearly as much, only about 40%.

To stimulate households to recycle more and generate less rest waste, some municipalities have introduced unit-based pricing for waste collection. The early results seem positive. If unit-based pricing is introduced, households tend to generate far less rest waste and separate far more useful materials from the rest waste stream. It is, however, difficult to determine how significant the effect of unit-based pricing is. Thus far unit-based pricing has always been introduced in combination with recycling programs. Introducing unit-based pricing is also quite expensive, so the question remains whether the initial costs of introducing the system will be compensated by the lower costs of waste collection and treatment.

In Chapters 4, 5 and 6, the data presented here will be used to analyze different aspects of the waste market. With the analysis presented in those chapters, I especially hope to clarify the costs and benefits of a unit-based pricing system and determine which municipalities should indeed introduce a unit-based pricing system.

Part II

Modeling waste management problems

4 Modeling market distortions in an applied general equilibrium framework: the case of flat fee pricing in the municipal solid waste market[1]

4.1 Introduction

Current waste management policies are inadequate to achieve a significant reduction in generation of municipal solid waste. Although governments have made great efforts to reduce waste generation, the actual quantity of waste generated has continued to rise. This is mostly due to economic growth. As shown in Chapter 2, governments have failed to achieve a decoupling between waste generation and economic growth due to presence of market distortions in the municipal solid waste market. In the Netherlands, these market distortions are (i) flat fee pricing, (ii) virgin material biased policies, and (iii) killer-contracts between municipalities and waste treatment facilities. All of these market distortions can lead to the market failure whereby waste generation is higher and recycling lower than is socially optimal, thus incurring inefficiently high waste treatment costs (Miedema, 1983).

Several studies have been conducted to analyze the effects of these market distortions and to suggest possible solutions. Wertz (1976) found that the introduction of a user charge for waste collection led to a significant reduction of waste generation. Miedema (1983) showed that a virgin material tax could reduce waste generation. Other more recent studies include Jenkins (1993), Hong *et al.* (1993), Miranda *et al.* (1994), Morris *et al.* (1994) and Sterner and Bartelings (1999). The overall conclusion of these empirical studies is that the demand for waste services is sensitive to unit-based pricing. The introduction of a unit-based price can lead to a substantial reduction in waste generation, especially if combined with programs that increase public awareness of the waste problem. However, imprudent construction of unit-based pricing may not have the desired effect and can even encourage illicit dumping, burning or other improper disposal (*e.g.* Fullerton and Kinnaman, 1995).

Although most studies agree that a flat fee pricing system is not optimal, they do not agree on optimal policy choice to minimize cost of disposal. Studies, such as

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[1] This chapter is adapted from: Bartelings, H., R.B. Dellink and E.C. van Ierland. Modeling market distortions in an applied general equilibrium framework: the case of flat fee pricing in the waste market. In: J.C.J.M. van den Bergh and M. A. Janssen (eds) *Economics of industrial ecology.* Cambridge: MIT press (forthcoming).

Miedema (1983), Jenkins (1993), Strathman *et al.* (1995) and Linderhof *et al*. (2001), have proposed the introduction of a ‘downstream’ tax, which would increase the price of disposal. Other studies, like Fullerton and Kinnaman (1995; 1996); Palmer and Walls (1997); Fullerton and Wu (1998) and Choe and Frasier (1999), favor an ‘upstream’ tax, which internalizes the costs of waste treatment in the price of the consumption good. They fear that a ‘downstream’ tax would be non-optimal due to huge implementation and enforcement costs.

In this chapter, a general equilibrium model is developed to analyze the efficiency of a ‘downstream’ tax, namely the unit-based pricing scheme, and an ‘upstream tax’, namely the advanced disposal fee (or waste tax). The general equilibrium approach makes it possible to include the entire product life cycle, from extraction, production, consumption, and collection to final disposal. Policies that attempt to reduce waste disposal will affect all of these stages. New in the analysis is the explicit role of the municipality as collector of waste. The method of solid waste collection, pricing of waste collection and the subsequent choice of waste treatment options lies solely with the municipality; the municipalities, therefore, have a significant effect on the social costs of waste treatment.

The effects of waste policy options available to the government are also analyzed in this chapter. To make a fair analysis between different policy options both implementation and enforcement costs of introducing these policy options are included in the model.

This chapter is structured as follows: Section 4.2 describes the model and provides some insights into how different policy options can be included in an applied general equilibrium model. Section 4.3 presents a stylized example based on numerical data from the Netherlands collected in 1996 and shows how different waste management policies can affect waste generation. Section 4.4 concludes and offers some policy recommendations.

4.2 Description of the model

4.2.1 Introduction

In this section, an applied general equilibrium model of the waste market is presented with the use of three sub-modules. Section 4.2.3 describes the sub-module, which includes unit-based pricing for waste collection, Section 4.2.4 explains the sub-module, which includes flat fee pricing, and Section 4.2.5 describes the sub-module, which includes an upstream tax.

As described in Chapter 1, there are several formats that can be used to build a general equilibrium model, for example the ‘Computable General Equilibrium format’, the

'Open economy format', the 'Full format', and the 'Negishi format'. Some of these formats are written in terms of excess demands, other in terms of welfare programs. Extensive information about the strengths and weaknesses of each of these formats can be found in Ginsburgh and Keyzer (1997). However, it should be stressed that a format is simply a way of presenting a model. A different format will still describe the same model and result in the same equilibrium solution.

In this chapter, a general equilibrium model is built according to the Negishi format. The advantage of this format is that it is relatively easy to incorporate externalities and non-convexities (see also Ginsburgh and Keyzer, 1997). Hence this format is particularly suitable for incorporating market distortions like flat fee pricing.

4.2.2 The subsidy-cum-tax scheme

Most municipalities have chosen to charge a fixed amount of money for waste collection, the so-called flat fee. In a flat fee-pricing scheme, the amount of money paid for waste collection is independent of the quantity of waste actually generated. The perceived price for waste collection, in economic terms the marginal perceived costs of generating waste, equals zero in such a case. If the price of a good equals zero the equilibrium demand for that good can no longer be determined through the normal demand and supply functions. In the general equilibrium framework in particular, where it is assumed that some equilibrium price will ensure that demand equals supply, the zero price poses a problem. To implement a zero price in a general equilibrium model, we thus require an indirect approach. It is possible to implement a zero perceived price by using subsidies that compensate households for the cost of waste generation.

Households pay a fixed lump-sum transfer to the government for the collection of waste, based on the flat fee. This lump-sum transfer takes away part of the households' income. Therefore, the total expenditure of the households declines. The expenditure pattern, *i.e.* the percentage of income the households spend on a certain product will, however, not be affected.

In the model presented in this chapter, private households demand waste collection services and pay an equilibrium price for these services. To introduce the zero perceived price, the government reimburses these costs to the consumers in the form of a subsidy, which equals the equilibrium price for every unit of waste collection services exactly. Thus, the perceived price of waste collection for the households equals zero. If the revenue of the lump-sum transfer is lower than the amount spent on the subsidy, the government expenditure decreases (in this case there is a net subsidy on waste generation). If the revenue of the flat fee is higher than the total costs, government expenditure increases. The idea of the subsidy-*cum*-lump-sum transfer scheme is illustrated in Figure 4-1.

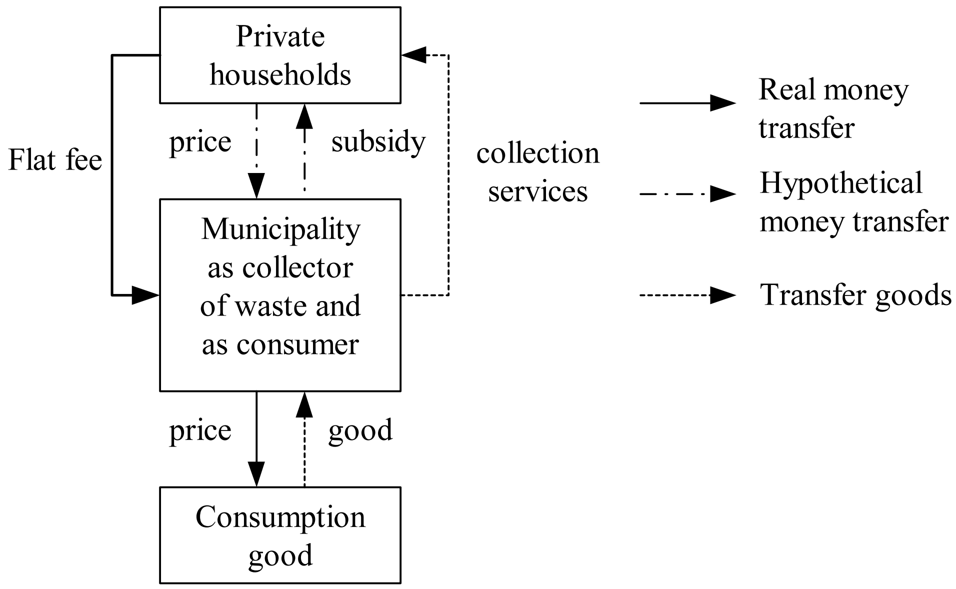

Figure 4-1 The subsidy-cum-tax scheme

Section 4.2.4 shows how the subsidy-cum-tax scheme can be implemented in a general equilibrium model.

4.2.3 Description of the model including a unit-based price for waste collection

In a simplified economy, two types of actors are distinguished: households and firms. Households consume goods and supply endowments; firms produce goods with the use of endowments and intermediate goods. Consumers are differentiated into two types: private consumers and a government consumer. Five different production sectors are distinguished, together producing eight unique goods. These sectors are: (1) an extraction sector producing virgin material; (2) a production sector producing agricultural goods, industrial goods and services; (3) a recycling sector producing recycling services; (4) a collection sector producing collection services and (5) a waste treatment sector producing incineration services and landfilling services. The hypothetical economy is shown in Figure 4-2.

Private households consume the consumer goods: agricultural goods, industrial goods, and services. The government consumes only services. Consumption of agricultural and industrial goods leads to the generation of municipal solid waste. Waste must be either recycled or collected by the municipality. We assume that collected rest waste is not separated and recycled after collection, but is instead sent immediately to an incineration plant or landfill unit. Although this puts some constraints on the model, we feel that this assumption is justified. We are primarily interested in the choice the consumer makes: the consumer can, for example, choose to separate organic waste, paper, or glass from rest waste. The consumers will have to incur costs in order to recycle these materials. Recycling will, for example, cost the consumer both time and storage space. This is modeled as if the consumer buys 'recycling services'. By buying recycling services, they generate recyclable waste; this waste is sent to a recycling unit where it is turned into recycled material.

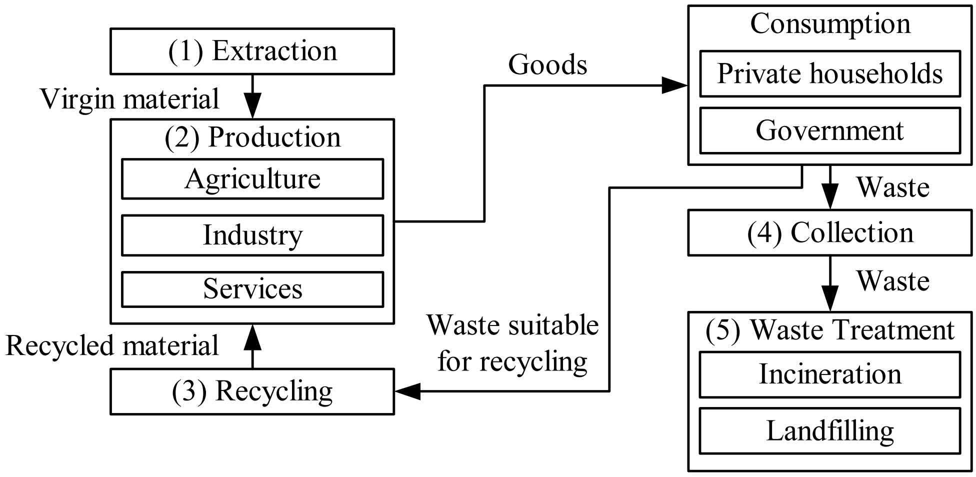

Figure 4-2 Representation of the hypothetical economy

Consumers can prevent waste by recycling more or, to a lesser extent, by substituting waste intensive goods, *i.e.* agricultural and industrial goods, for waste extensive goods, *i.e.* services. In reality, consumers have the possibility of two kinds of substitution, namely substitution within a sector and substitution between sectors. Substitution within a sector makes it possible to choose between two products that are basically the same except for waste intensity. Substituting between sectors would mean changing consumption patterns. For example, in Oostzaan, a municipality where a unit-based pricing scheme was recently introduced, households reported that they not only bought more products containing less packaging, an example of substitution within a sector, but also began to use diaper services instead of disposable diapers, an example of substitution between sectors (Linderhof *et al*., 2001). Waste prevention through substitution within a sector would add a certain degree of complexity to the model, as different products within the same sector and their associated 'waste intensity' would have to be explicitly modeled. In our opinion, this trade-off between accuracy and transparency of the model is not easy to make, but in this case we have chosen to include only the more straightforward channel of waste prevention through substitution between sectors. As a consequence, the possibility of waste prevention – and thus also the effects of introducing either a unit-based price of an upstream waste tax - may, therefore, be underestimated. This assumption, however, will not affect the comparison between the effectiveness of a unit-based price and an upstream waste tax, since the substitution possibilities will be identical in these two scenarios.

We assume that only private households generate waste. Both the government consumer and firms do not generate waste. we made this assumption (although not completely realistic) because the focus is on policies affecting the generation of municipal solid waste, and not on policies that affect the generation of industrial or government waste.

All the firms use capital and labor to produce goods or services. The extraction sector produces virgin material, which is sold to the production sector of consumer goods. The recycling sector sells recycling services to the consumer and recycled material to the production sector of consumption goods. Besides capital and labor, the production sector of consumption goods uses virgin materials and recycled materials as inputs to production. The collection sector sells collection services to private households. They use capital, labor, and waste treatment services as inputs. Finally, the waste treatment sector sells waste treatment services to the collection sector. It consists of two producers: a producer of incineration services and a producer of landfilling services.

Consumer utility function

In the Negishi format, total welfare is maximized subject to utility, balance, and production possibility constraints (Ginsburgh and Keyzer, 1997). The total welfare function is shown in equation 4.1. Total welfare (*TW*) equals the sum of weighted utilities (u_i) over consumer i (i=1,...,n) .

$$TWF(\alpha_i) = Max\sum_i \alpha_i u_i(x_i^g) \tag{4.1}$$

Consumers derive utility from the consumption of consumer goods (x_i^g) where g= agricultural goods, industrial goods and services. The utility of each consumer is weighted by a factor α_i, the so-called Negishi weights.[2]

Consumers generate waste by consuming products. Waste generation is dynamic; not all products will be transformed into waste immediately after consumption. Durable goods, for example, can continue to function properly for several years. If one looks at an infinite time scheme, every good will turn into waste. At any point in time, however, only part of the products will be transformed into waste. To include this dynamic aspect in a comparative static model, waste is determined as a fraction β^g of the consumption product[3]. Total waste generation per consumer (W_i) is equal to a

[2] These Negishi weights are determined in such a way that each consumer's budget constraint holds. This means that consumers cannot spend more money on goods and services than they receive on sales of primary inputs (capital and labor). The value of the Negishi-weights is exogenous to the model. How these Negishi weights are determined and how the equilibrium solution is found is described in appendix 4-A. See Ginsburgh and Keyzer (1997) for more information.

[3] Implicitly this means that part of the used material accumulates in a stock of durable goods. This stock is not constant, new materials enter the stock and other materials leave the stock as waste.

fixed percentage of total consumption. The fraction of waste contained in a product differs for the three types of consumption goods. Agricultural and industrial goods are relatively waste intensive and thus β will be positive for these goods; consumption of services does not generate waste and thus β is equal to zero in this case. The government only consumes services and does not generate waste, therefore, in the following equation a subset *c* is used, which encompasses only the private households.

$$W_c = \sum_g \beta^g x_c^g \tag{4.2}$$

All waste that is generated has to be dealt with. Private households can chose to either recycle the waste by demanding waste recycling services (x_i^r) or to allow the waste to be collected by demanding waste collection services (x_i^w).

Production functions

All production sectors can use two primary production factors, namely capital *(k)* and labor *(l)* and four intermediate inputs, namely virgin material (m^v), recycled material (m^r), incineration services (w^i) and landfilling services (w^l). All producers generate commodities y_j within their given production set Y_j.

$$y_j \in Y_j \tag{4.3}$$

The production set for the three consumption goods, *i.e.* agricultural goods, industrial goods, and services is given by a nested Leontief-CES production function, which depends on the input of capital, labor, virgin material, recycled material, and waste treatment services[4].

Therefore, at any given moment in time the material inflow does not have to be equal to the material outflow in the model.

[4] The notation $z=CES(x,y;\sigma)$ reflects the following function: $z = \left(x^{\frac{(\sigma-1)}{\sigma}} + y^{\frac{(\sigma-1)}{\sigma}} \right)^{\frac{\sigma}{\sigma-1}}$

If a good is produced with production factors that are completely complementary (σ→∞), a Leontief production function can be used as a special case of the CES-production function. The standard notation for a Leontief production function is: *z=min(x,y)*. A CES function can be nested. This means that, for example, the variable *x* in the equation above actually represents another function. In this chapter, several nested CES functions are used.

$$Y_j = A_j \left[min \left\{ CES(k_j, l_j; \sigma^{kl}), CES(w_j^{is}, w_j^{ls}, CES(m_j^{\ v}, m_j^{\ r}; \sigma^{vr}); \sigma^{wm}) \right\} \right] \tag{4.4}$$
$$for\ j = agriculture, industry, services$$

Where A stands for the technology level.

The production set for the producer of recycled material is given by a nested CES-function, which depends on the input of capital, labor, and recyclable waste:

$$Y_j = A_j \left[CES \left\{ CES(k_j, l_j; \sigma^{kl}), X^r; \sigma^{pr} \right\} \right] \qquad for\ \mathrm{j} = recycling\ services \tag{4.5}$$

Where X^r is the total quantity of recyclable waste generated by the private households.

The production set for the producer of collection services is indicated by a nested Leontief-CES-function, which depends on the input of capital, labor, incineration services, and landfilling services:

$$Y_j = A_j \left[min \left\{ CES(k_j, l_j; \sigma^{kl}), CES(w_j^{is}, w_j^{ls}; \sigma^{il}) \right\} \right] \quad for\ \mathrm{j} = collection\ services \tag{4.6}$$

The production sets of all other production sectors are defined by CES-functions, which only depend on the input of capital and labor.

Balance equations

As in any general equilibrium model, demand for commodities (consumed goods and primary factors) should be equal to the supply of these commodities (produced goods and endowments). This is ensured by the following balance equations.

First of all, total demand for consumption good *g* by consumer *i* and total demand for intermediate good *g* by producer *j* must not exceed the total supply *(y^g)* of good *g,* where *g* is an index of the three consumer goods: agricultural goods, industrial goods and services. The prices of the commodities can be determined from the balance equations by calculating the shadow price of the balance equation. In the following equations, this is symbolized by the '$\perp$' and a price variable p.

$$\sum_i x_i^g + \sum_j x_j^g \le y^g \qquad \perp p^g \tag{4.7}$$

Total demand of all firms *j* for the intermediate goods: "virgin material" *(m_j^v),* and "recycled material" *(m_j^r),* must not exceed total supply of these materials *(y)*. Since virgin materials and recycled materials are intermediate goods only, *i.e.* not demanded by the consumers, the only demand comes from firm *j*.

$$\sum_j m_j^v \le y^v \qquad \perp p^v \tag{4.8}$$

$$\sum_j m_j^r \le y^r \qquad \perp p^r \tag{4.9}$$

Total demand for the services: "recycling services" *(x^{rs})* and "waste collection services" *(x^w)* by consumer *c* must be equal to or less than the total supply of these services.

$$\sum_c x_c^{rs} \le y^{rs} \qquad \perp p^{rs} \tag{4.10}$$

$$\sum_c x_c^w \le y^w \qquad \perp p^w \tag{4.11}$$

Total demand for the intermediate good: "waste treatment service" *(w_j^n),* where *n* is a set of incineration and landfilling services, must be equal to or less than total supply of these waste treatment services.

$$\sum_j w_j^n \le y^n \qquad \perp p^n \tag{4.12}$$

Total demand of primary factors must be equal to or less than total supply of these factors ($\overline{K}, \overline{L}$). The total supply of capital and labor is equal to the sum of initial endowments of each consumer.

$$\sum_j k_j \le \sum_i \overline{K_i} \qquad \perp p^k \tag{4.13}$$

$$\sum_j l_j \le \sum_i \overline{L_i} \qquad \perp p^l \tag{4.14}$$

Prices for all commodities are calculated as the marginal value of the associated balance equations. The consumer obtains income by selling production factors, capital, and labor and spends his income on the three consumer goods, recycling services and waste collection services. The government only spends its income on the consumption of goods.

$$\sum_g p^g x_c^g + p^{rs} x_c^{rs} + p^w x_c^w = p^k \overline{K_c} + p^l \overline{L_c} \tag{4.15}$$

$$\sum_g p^g x_{gov}^g = p^k \overline{K_{gov}}$$

4.2.4 Description of the model including a flat fee for waste collection

To implement the subsidy-*cum*-tax scheme, as discussed in Section 4.2.2, the objective function, equation 4.1, is extended by a subsidy term[5]. This subsidy term works like a benefit on the allocation of production output. Maximum social welfare now depends on the weighted utility of consumer *i* on the one hand and on the total benefits of the subsidy (ξX^w) on the other, where X^w stands for the total quantity of waste generated and ξ stands for the subsidy wedge, which is the total amount of money spent on the subsidy per unit of waste.

$$TWF(\alpha) = \max \sum_i \alpha_i u_i(x_i^g) + \xi \; X^w \tag{4.16}$$

$$x_i^g \geq 0 \;,\; w_i \geq \; 0, \; r_i \; \geq \; 0 \;\; all\, i \;\; , \;\; y_j \;\; all\, j$$

Adding the subsidy to the social welfare function is done solely to change the perceived price of waste collection. It does not imply that introducing subsidies would positively influence social welfare of a region. The social welfare calculated by this model is not comparable with the social welfare calculated by the model presented in Section 4.2.3. The presence of the subsidy in the welfare function is for technical reasons and specific to the Negishi format of the model. If the model were written in another format, the subsidy would not have been made explicit in the welfare function.

The subsidy wedge *(*ξ*)* is defined as the difference between the equilibrium price for waste collection *(*p^w*)* and the perceived price *(*p_c^w*)*. In the present case, the perceived price of waste collection equals zero, thus the subsidy wedge is equal to the equilibrium price of waste collection.

The balance equation for waste collection services (equation 4.11) is rewritten as follows:

$$X^w \leq y^w \qquad\qquad \perp p^w \tag{4.17}$$

$$\sum_c x_c^w \leq X^w \qquad\qquad \perp p_c^w \tag{4.18}$$

In equation 4.17 the shadow price of waste collection has been calculated. This price equals the marginal production costs. In equation 4.18, the shadow price of waste

[5] See Ginsburgh and Keyzer (1997) for details on this procedure.

collection, as consumers perceive it, is calculated. This price equals the equilibrium price minus the subsidy[6].

The new budget constraint for the private households is defined as follows:

$$\sum_g p^g x_c^g + p^{rs} x_c^{rs} + p_c^w x_c^w + F_c = p^k \overline{K}_c + p^l \overline{L}_c \tag{4.19}$$

Private households spend their income on the consumption of consumer goods, recycling services and collection services (bear in mind that p_c^w is zero, so the costs of consumption of waste collection services is equal to zero) and pay a flat fee *(F)* to the government for the collection of waste.

The new budget constraint of the government is defined as follows:

$$\sum_g p^g x_{gov}^g + S = p^k \overline{K}_{gov} + \sum_c F_c \tag{4.20}$$

The government spends its income on consumer goods and the subsidy costs *(S)*. Since the government does not generate waste, it need not spend any income on the collection of waste. We assume that the government owns primary factors and earns income both from selling these primary factors and benefits of the flat fee.

The size of the subsidy costs depends on the total amount spent on the subsidy for waste collection, which is calculated as follows:

$$S = \xi \sum_i x_i^w \tag{4.21}$$

The total transfer equals the subsidy wedge *(ξ)* multiplied by the total demand for waste collection services. The subsidy wedge is calculated as follows:

$$\xi = p^w - p_c^w \tag{4.22}$$

The subsidy wedge is equal to the real price of waste collection minus the perceived price of waste collection.

[6] Note that mathematically speaking, the introduction of the total waste demand variable is irrelevant. $X^w = \sum_c x_c^w$ can be substituted in the balance equation in the equilibrium solution. The distinction of X^w, however, enables the separation of the equilibrium price for waste collection and the perceived price.

4.2.5 Description of model including an upstream tax for waste collection

In the upstream tax model, the price of waste collection and treatment is internalized in the price of the consumption good. Only agricultural and industrial goods are taxed, given that the consumption of services does not generate municipal solid waste. Introducing a tax in the Negishi format is quite similar to introducing a subsidy. First of all the social welfare function should be adjusted. The new social welfare function is defined as follows:

$$TWF(\alpha) = \max \sum_i \alpha_i u_i(x_i^g) + \xi^w X^w + \sum_g \xi^g X^g \tag{4.23}$$

$$x_i^g \geq 0\ ,\ w_i \geq\ 0,\ r_i\ \geq\ 0\ \ all\, i\ \ ,\ \ y_j\ \ all\, j$$

Where ξ^g is the tax wedge and X^g is the total demand for good *g*.

Just as in Section 4.2.4, the balance constraint for the consumption goods also has to be changed. It is important to realize that only the private households pay an upstream tax for waste collection. Neither the producers, who demand goods as intermediate deliveries, nor the government, who does not to generate waste, have to pay this tax. The new balance constraints are defined as follows:

$$X^g + \sum_j x_j^g + x_{gov}^g \leq y^g \qquad \perp p^g \tag{4.24}$$

$$\sum_c x_c^g \leq X^g \qquad \perp p_c^g \tag{4.25}$$

The equilibrium price for consumption goods can be calculated from the first balance constraint (4.24). Both the producers and the government pay this price while consuming these goods. In the second balance constraint (4.25), the price including the upstream tax is calculated. Only private households pay this price.

The budget constraint for private households is defined as:

$$\sum_g p_c^g x_c^g + p^{rs} x_c^{rs} + p_c^w x_c^w + F_c = p^k \overline{K}_c + p^l \overline{L}_c \tag{4.26}$$

The budget constraint for the government is defined as:

$$\sum_g p^g x_{gov}^g + S = p^k \overline{K}_{gov} + \sum_c F_c + T \tag{4.27}$$

Where *T* equals the total gains of the upstream tax.

4.3 A numerical example

The model discussed above is applied in a stylized example with numerical data from the Netherlands. The economic data used in the numerical example are based on national accounts for the Netherlands in 1996 (Statistics Netherlands, 1998). These data are aggregated to four sectors (agricultural goods, industrial goods, services and extraction) and two production factors (capital and labor) and supplemented with detailed data of the waste sectors (recycled material, recycling services, collection, incineration, landfilling, fee and subsidy) based on WMC (1998) and RIVM (1998). To keep the model as simple as possible, we have chosen to give the government an income dependent on capital endowments instead of an income from taxes on labor and consumer goods[7]. This has been adjusted in the social accounting matrix.

4.3.1 Parameter values used in numerical example

The social accounting matrix, displayed in Table 4-1, describes the initial equilibrium. Supply or producers' output and consumer endowments are given positive values; demand or producer inputs and consumption are given negative values[8].

Prices are normalized to unity, except the price of waste collection[9]. Private households pay 9.5 million guilders in the form of a flat fee for collection of waste. This is slightly lower than the real cost of waste collection, which equals 10 million guilders. This means that waste collection basically has two prices: the perceived price and the social price of waste collection.

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[7] Although less realistic, we feel that it is justified to make this assumption since our focus is on waste generation and recycling by private households. As we are interested in finding a first-best equilibrium solution, we abstract from the existence of distortionary taxes. Therefore, in our model in makes no difference whether the government has non-distortionary taxes as income or capital endowments.

[8] The entries in the column times the corresponding prices sums up to zero to ensure that the zero profit condition holds: value of inputs equals value of outputs. The entries in the column of each consumer times the corresponding price sums up to zero to ensure that the budget constraint holds: each consumer spends exactly his income on the consumption of goods and services. The entries in each row times the corresponding prices sums up to zero to ensure that each market clears: total demand for each commodity must equal total supply.

[9] Following standard practice, we adopted the Harberger convention in the benchmark data for all unknown prices. The Harberger convention consists of normalizing prices to unity. Quantities in the benchmark data represent expenditures, or how much of that good or factor one can buy for €1. It should be noted that an Arrow-Debreu economy only depends upon relative prices. Doubling all prices doubles both money profits and income, which results in the same equilibrium outcome.

Table 4-1 Benchmark social accounting matrix (expenditures in Billion NLG, 1996, 1 EUR=2.2 NLG)

	Agriculture	Industry	Services	Extraction	Recycled material	Recycling services	Collection	Incineration	Landfilling	Consumer	Government	price
Agriculture	40.00	-18.00	-1.00	0.00	0.00	0.00	0.00	0.00	0.00	-21.00	0.00	1.00
Industry	-12.00	219.00	-48.00	0.00	0.00	0.00	0.00	0.00	0.00	-159.00	0.00	1.00
Services	-4.00	-57.00	527.00	0.00	0.00	0.00	0.00	0.00	0.00	-446.00	-20.00	1.00
Extraction	-1.00	-31.80	-2.00	34.80	0.00	0.00	0.00	0.00	0.00	0.00	0.00	1.00
Recycled material	0.00	-0.40	0.00	0.00	0.40	0.00	0.00	0.00	0.00	0.00	0.00	1.00
Recycling	0.00	0.00	0.00	0.00	0.00	0.25	0.00	0.00	0.00	-0.25	0.00	1.00
Recycled waste	0.00	0.00	0.00	0.00	-0.25	0.25	0.00	0.00	0.00	0.00	0.00	1.00
Collection	0.00	0.00	0.00	0.00	0.00	0.00	0.95	0.00	0.00	-0.95	0.00	1.00
Incineration	0.00	0.00	0.00	0.00	0.00	0.00	-0.60	0.60	0.00	0.00	0.00	1.05
Landfilling	0.00	0.00	0.00	0.00	0.00	0.00	-0.20	0.00	0.20	0.00	0.00	1.00
Capital	-18.00	-46.80	-197.00	-27.30	-0.01	-0.10	-0.10	-0.50	-0.15	270.00	20.50	1.00
Labor	-5.00	-65.00	-279.00	-7.50	-0.05	-0.40	-0.10	-0.10	-0.05	357.20	0.00	1.00
Fee	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	-0.95	0.95	1.00
Subsidy	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	1.00	-1.00	1.00

Note: 'Fee' is the flat fee consumers pay to the government for the collection of waste; 'Subsidy' stands for the total amount of money the government gives for waste collection as a subsidy to the consumers. The price column gives the prices of all commodities.

The perceived price equals the total fee divided by the total demand for waste collection. The social price equals the total fee plus the total amount paid by the municipality for waste collection divided by total demand. We have chosen to normalize the perceived price of waste collection, which means that the social price for waste collection (which is shown in Table 4.1) is higher than unity.

All production sectors are characterized by a CES production function. As mentioned earlier, all production sectors use capital and labor. The substitution elasticity between capital and labor equals 0.8, based on the study by Draper and Manders (1996). Other substitution elasticities are presented in Table 4-2. The three production sectors of consumer goods (agriculture, industry and services) also use intermediate inputs for production. The use of primary factors and intermediate inputs is strictly complementary. Only the producer of industrial goods uses recycled material. They can fully substitute recycled for virgin materials.

Table 4-2 Substitution elasticities for the production sectors

	Agriculture	Industry	Services	Recycled Material	Collection
Sub.elas. primary & intermediate inputs (σ^{pi})	0.0	0.0	0.0	-	-
Sub.elas. materials & intermediate inputs (σ^{wm})	1.0	1.0	1.0	-	-
Sub.elas. recycled material & virgin material (σ^{vr})	-	∞	-	-	-
Sub.elas. primary factors & recycled waste (σ^{pr})	-	-	-	0.125	-
Sub.elas. landfilling & incineration (σ^{il})	-	-	-	-	0.2

The substitution parameters for the households are shown in Table 4-3. Utility of the private households depends on consumption of agricultural goods, industrial goods, and services. A substitution elasticity of unity between goods is assumed (Cobb-Douglas utility function). The government only consumes services and thus has no substitution elasticity between consumption goods.

Table 4-3 Additional parameters for households in the benchmark

	Consumer	Government
Substitution elasticity between consumer goods *(σ^g)*	1.0	
Negishi weights *(α)*	96.9	3.1

The initial Negishi weights are determined on the basis of the initial income (sales of endowments). Since the income of the private households is far greater than the income of the government, the Negishi weight of the private households is much larger.

In the base case scenario, collection and treatment of municipal solid waste costs about 1.2 Billion guilders. Consuming either agricultural goods or industrial goods generates waste. One unit of agricultural goods contains a smaller percentage of waste than industrial goods. The percentage of waste present in a unit of agricultural goods is equal to 0.46 and the percentage of waste present in a unit of industrial goods is equal to 0.69. Of the waste generated, about 20% is recycled and 80% collected for waste treatment (either landfilling or incineration). Most of the waste collected is incinerated (75%); the rest is landfilled.

4.3.2 Policy scenarios

The model specified in the previous section is used to analyze the effects of different policy options, especially on the quantity of solid waste generated and the total costs of waste treatment. In this chapter, four policy instruments are compared, namely unit-based pricing of waste collection, recycling subsidy, unit-based pricing of waste collection combined with a recycling subsidy and an upstream tax. Seven different scenario's are distinguished: (*i*) unit-based price, (*ii*) recycling subsidy (*iii*) unit-based price plus recycling subsidy (*iv*) upstream tax, (*v*) unit based price plus transaction costs, (*vi*) unit based price plus recycling subsidy plus transaction costs and (*vii*) upstream tax plus transaction costs.

The benchmark case is an exact replication of the benchmark data presented in section 4.3.1 without added policies. The seven scenarios will all be compared with the benchmark case.

In the *first* scenario, the flat fee-pricing scheme is replaced by a unit-based pricing scheme. The private households now bear the full costs of waste collection, whereas in the flat fee pricing system, consumers only had to pay 95% of the total costs. This scenario is labeled the 'unit-based price scenario'.

According to the *second* scenario, recycling is promoted by lowering the cost of producing recycling services, *i.e.* production costs for recycling are halved, and thus the benchmark price for recycling services is halved. The flat pricing scheme remains unchanged. This policy is labeled the 'recycling subsidy scenario'.

In the *third* scenario, the unit-based pricing scheme is combined with a recycling subsidy. As shown in Chapter 2, the unit-based pricing scheme is usually implemented together with policies intended to stimulate recycling. Therefore, in this scenario both policy options are implemented together.

A small upstream tax on agricultural and industrial goods is introduced in the *fourth* scenario. This tax internalizes the cost of waste collection and treatment in the price of the product. The private households do not have to pay anything for the collection of waste. This scenario is labeled the 'upstream tax scenario'.

In the *fifth* scenario, the unit-based pricing system is introduced once again. In this scenario, however, some transaction costs of introducing a unit-based price are included. These transaction cost may involve the costs of installing weighing scales in garbage trucks and costs incurred as a consequence of preventing illegal disposal. By changing the available technology (A) in the production function, we introduced transaction costs. It is here assumed that a more expensive technology has to be used, which means that less output can be generated with the same amount of input. This scenario is labeled the 'unit-based price plus transaction costs scenario'.

In the *sixth* scenario, the unit based pricing system and recycling subsidy is combined with the transaction costs involved in implementing such a policy change. Transaction costs are implemented in the same manner as in scenario five.

In the *seventh* and final scenario, the upstream tax is combined with transaction costs of introducing an upstream tax. We assume that all transaction costs will be borne by the consumers. This means that the tax will be higher than in the 'upstream tax scenario'. This scenario is labeled the 'upstream tax plus transaction costs scenario'. The different elements of each scenario are summarized in Table 4-4.

Table 4-4 Main characteristics of the policy scenarios

Scenario	Fee collection	Tax or subsidy	Transaction costs
Benchmark case	Flat fee	No	No
1. Unit-based price	Unit-based fee	No	No
2. Recycling subsidy	Flat fee	Subsidy	No
3. Unit-based price + recycling subsidy	Unit-based price	Subsidy	No
4. Upstream Tax	No fee	Tax	No
5. Unit-based price + transaction costs	Unit-based fee	No	Yes
6. Unit-based price + recycling subsidy + transaction costs	Unit-based fee	Subsidy	Yes
7. Upstream tax + transaction costs	No fee	Tax	Yes

4.3.3 Results

First scenario: Unit-based pricing scheme scenario

In the first scenario, a unit-based pricing scheme is introduced. Households pay the equilibrium price for waste collection. This means that generating more waste will result in higher collection costs. Table 4-5 shows the changes in the main variables of the model. The government no longer bears the costs of collection. This means that the relative income that can be used for the consumption of services increases. Therefore, to keep government expenditure constant, as discussed in Section 4.3.1, private households receive a positive lump-sum transfer from the government. Private households now bear the full cost of waste collection, but due to the positive lump-sum transfer there is small change in their available income.

Table 4-5 The main variables for the 'Unit-based pricing scenario' as compared to the 'Benchmark case' (expenditures in Billion NLG, 1996) and the percentage change

Variable	Benchmark case	Unit-based price	% Change
Private demand agricultural good	21.00	20.94	(-0.27%)
Private demand industrial good	159.00	158.18	(-0.52%)
Private demand services	446.00	446.89	(0.20%)
Private demand recycling services	0.25	0.26	(6.65%)
Private demand waste collection	0.95	0.93	(-2.38%)
Utility private households	309.77	309.78	(0.00%)

Households are given an incentive to prevent waste and recycle more. They substitute *agricultural and industrial goods*, which contain relative large quantities of waste, for *services,* which do not contain waste. Since the perceived price for waste collection has risen, there is some substitution between recycling and waste collection. Demand for *recycling services* increases and the demand for *collection services* decreases. Since the consumer's income is hardly affected by the policy change, the utility of the private households remains almost unchanged.

Second scenario: Recycling subsidy

In the second scenario, the production costs of recycling are reduced. This is done by introducing a new technology parameter in the production set, which makes it possible to produce the same quantity of recycling services with the use of less production factors. Table 4-6 shows the changes of the most important variables.

Table 4-6 The main variables for the 'Recycling subsidy scenario' as compared to the 'Benchmark case' (expenditures in Billion NLG, 1996) and the percentage change

	Benchmark case	Recycling subsidy	% Change
Private demand agricultural good	21.00	21.00	(0.019%)
Private demand industrial good	159.00	159.04	(0.025%)
Private demand services	446.00	446.12	(0.027%)
Private demand recycling services	0.25	0.25	(0.000%)
Private demand waste collection	0.95	0.95	(0.031%)
Utility private households	309.77	309.85	(0.026%)

The change in most variables is very small. Since the quantity of recycling is quite small according to the benchmark, the effects of lower recycling costs will also be fairly minimal. The demand for *recycling services* is unaffected by the lower price for *recycling services*. This is a logical result because if consumers have the choice between *collection and recycling* services, they will chose *collection*, which is free.

Thus, a lower price for *recycling services* does not affect the demand for these services as long as this price is larger than zero. The demand for *waste collection* rises slightly since the consumer can consume more goods since they have to spend less income on recycling.

The utility of the private consumers rises slightly, because lower recycling costs imply that a larger percentage of the income can be spent on consumer goods. The money that the government spends on the subsidy is slightly increased, as the consumers demand more *waste collection services*. Since the expenditure of the government is kept constant at the benchmark level (see Section 4.3.1), this means that the government will receive a small lump-sum transfer from the private households to compensate for the extra costs.

Third scenario: Unit-based price plus recycling subsidy scenario

In the third scenario, both the unit-based price for waste collection and the lower price for recycling services are introduced simultaneously. Consumers are given a strong price incentive to demand more *recycling services* and less *waste collection services* (see Table 4-7). Recycling increases strongly, and waste collection decreases by more than 70%. Due to the lower costs for waste treatment, consumption and utility of the private households increase slightly. Compared to the recycling subsidy scenario, utility of the private households increases more than twice as much.

Table 4-7 Changes in the main variables for the 'Unit-based price and recycling subsidy scenario' as compared to the 'Benchmark case' (expenditures in Billion NLG, 1996) and the percentage change.

	Benchmark case	Unit-based price and recycling subsidy	% Change
Private demand agricultural good	21.00	20.97	(-0.14%)
Private demand industrial good	159.00	158.46	(-0.34%)
Private demand services	446.00	446.94	(0.21%)
Private demand recycling services	0.25	0.92	(266.55%)
Private demand waste collection	0.95	0.28	(-70.56%)
Utility private households	309.77	309.96	(0.06%)

Scenario two and three demonstrate the impact of policies aimed at promoting recycling under different pricing schemes for waste collection. Under the flat fee-pricing scheme, promoting recycling is not effective. Consumption rises, waste generation rises and waste collection rises; the exact opposite of the goal of the policy change. In scenario three, however, the quantity of waste generated decreases. More waste is recycled and less waste is collected, incinerated, and landfilled.

A comparison of these scenarios reveals that in the case of a flat fee for waste collection, the market is distorted and the price of recycling has no impact on the behavior of households. The unit-based price is far more effective when combined with a recycling subsidy. This is in line with the results of actual practice; municipalities always introduce a unit-based pricing scheme in combination with policies promoting recycling.

Fourth scenario: Upstream tax scenario

An upstream tax is introduced in the fourth scenario. The tax is quite small and is only intended to cover the real cost of waste collection. Since the consumption of agricultural goods and industrial goods leads to waste generation, these two goods are taxed.

Table 4-8 The main variables for the 'Upstream tax scenario' as compared to the 'Benchmark case' (expenditures in Billion NLG, 1996) and the percentage change.

Variable	Benchmark case	Upstream tax	% Change
Private demand agricultural good	21.00	20.96	(-0.21%)
Private demand industrial good	159.00	158.34	(-0.41%)
Private demand services	446.00	446.71	(0.16%)
Private demand recycling services	0.25	0.25	(-0.04%)
Private demand waste collection	0.95	0.95	(-0.49%)
Utility private households	309.77	309.78	(0.00%)

The results of the upstream tax scenario are shown in Table 4-8. Since households pay a higher price for agricultural goods and industrial goods, the demand for these goods declines. The demand for services increases, because the price of services has not been affected. Given that fewer agricultural and industrial goods are consumed, the quantity of waste generated decreases slightly. The utility of the consumers is hardly affected by the measure. Compared with the 'unit-based price scenario', it is clear that the up-stream tax is less effective in minimizing the waste problem. Moreover, there is no substitution of recycling for collection.

Fifth scenario: Unit-based price plus transaction costs scenario

A frequent complaint about the unit-based pricing scheme is the huge transaction costs of introducing such a scheme. In most models, these costs are left out of the analysis. In the fifth scenario both the unit based pricing scheme and transaction costs of introducing such a scheme have been included. To cover these extra costs, private households will have to pay a higher fee compared to the unit-based pricing scenario.

In Table 4-9 the results of third scenario are presented. Consumption has slightly decreased due to the increase of waste disposal costs. Private households must spend more income on waste disposal and thus have less money available for consumption. Since the costs of waste collection have increased, consumers begin to recycle more waste. Compared with scenario 1, more waste is recycled, while the utility of both the private households and the government is lower.

Table 4-9 The main variables for the ‘unit-based price plus transaction costs’ as compared to the ‘Benchmark case’ (expenditures in Billion NLG, 1996) and the percentage change

Variable	Benchmark case	Unit-based price + transaction costs	% Change
Private demand agricultural good	21.00	20.93	(-0.31%)
Private demand industrial good	159.00	158.08	(-0.58%)
Private demand services	446.00	446.90	(0.20%)
Private demand recycling services	0.25	0.31	(25.45%)
Private demand waste collection	0.95	0.88	(-7.40%)
Utility private households	309.77	309.73	(-0.02%)

Implementing a unit-based pricing scheme seems inefficient based on the results presented in Table 4-9. No government should implement a policy that lowers the total welfare of the country. However, it is important to bear in mind that environmental damage is not included in the model. Collection and treatment of waste leads to environmental damage. Recycling, on the other hand, results in far less environmental damage. If the state of the environment was to be included in the social welfare function, it may well be that this policy scenario performs relatively well in terms of an increase in social welfare.

Sixth scenario: Unit-based price plus recycling subsidy plus transaction costs scenario

The unit based pricing scheme, recycling subsidy, and transaction costs of introducing such a scheme are included in the sixth scenario. The transaction costs are implemented in the same way as in the fifth scenario; consumers bear all transaction costs. The results of this scenario are shown in Table 4-10.

Table 4-10 The main variables for the ‘Unit-based price plus recycling subsidy plus transaction costs’ as compared to the ‘Benchmark case’ (expenditures in Billion NLG, 1996) and the percentage change.

Variable	Benchmark case	Unit-based price + recycling subsidy + transaction costs	% Change
Private demand agricultural good	21.00	20.97	(-0.13%)
Private demand industrial good	159.00	158.45	(-0.34%)
Private demand services	446.00	446.57	(0.21%)
Private demand recycling services	0.25	11.96	(378.38%)
Private demand waste collection	0.95	0.00	(-99.99%)
Utility private households	309.77	309.73	(0.06%)

We assumed that there are no technical restrictions on recycling of waste, therefore in theory it is possible to recycle all waste that is generated. Although this assumption is not completely realistic, the main objective of this chapter is to demonstrate the main mechanisms of the model. Due to the increased price of waste collection services and the low costs of recycling, consumers start to recycle almost all their rest waste. Since recycling is cheaper than waste collection, they are able to spend a larger part of their income on the consumption of goods, thus their utility increases. Shifting consumption from *agricultural and industrial goods* to *services* prevents some of the waste. Compared to scenario five, more agricultural goods and industrial goods and about the same quantity of services are consumed.

Seventh scenario: Upstream tax plus transaction costs scenario

In the seventh scenario, an upstream tax on consumption goods is introduced. We assume that all transactions costs of introducing such a tax will be borne by the private households. This means that transaction costs may be introduced by increasing the total tax. This tax is slightly higher than in scenario 2, to cover the transaction costs.

Table 4-11 The main variables for the ‘Upstream tax scenario plus transaction costs’ as compared to the ‘Benchmark case’ (expenditures in Billion NLG, 1996) and the percentage change.

	Benchmark case	Upstream tax + transaction costs	% Change
Private demand agricultural good	21.00	20.95	(-0.26%)
Private demand industrial good	159.00	158.24	(-0.48%)
Private demand services	446.00	446.71	(0.16%)
Private demand recycling services	0.25	0.25	(-0.04%)
Private demand waste collection	0.95	0.95	(-0.57%)
Utility private households	309.77	309.72	(-0.02%)

In Table 4-11, the results of this scenario are presented. The higher tax does not change the results too greatly. Somewhat less waste is generated. The demand for agricultural goods and industrial goods decreases slightly and the demand for services, the only good without a tax, increases. These results indicate a minor decrease in the demand for both recycling services and collection services. Due to the costs of implementing the tax, the utility of both consumers decreases.

4.3.4 Sensitivity analysis

Substitution elasticity between consumer goods

The effectiveness of the upstream tax and, to a lesser extent, the unit-based pricing scheme depends on the substitution elasticity between consumption goods. If the demand for consumption goods is more elastic it can be expected that consumers will substitute more industrial and agricultural goods for services. A sensitivity analysis is performed for the substitution elasticity between the consumption goods. The substitution elasticity is changed from low to high in a number of (equidistant) steps, resulting in a very inelastic demand to an elastic demand. The effects of parameter changes on the variables: *rest waste* and *recyclable waste* are calculated.

Figure 4-3 shows the impact of the substitution elasticity on the generation of rest waste, which is collected by the municipality.

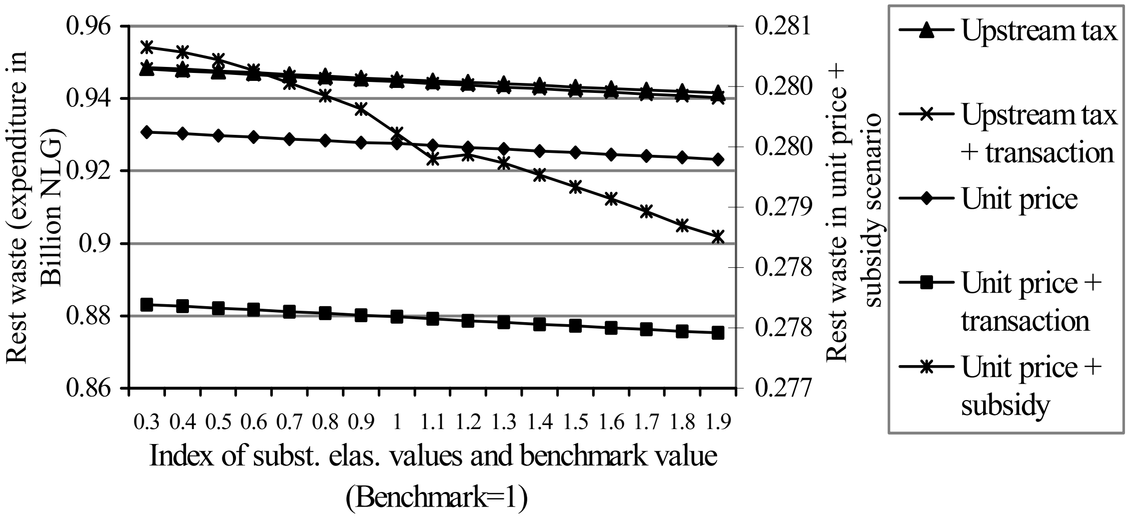

Figure 4-3 Sensitivity analysis of the substitution elasticity between different consumption goods: impacts on the generation of rest waste

Figure 4-3 does not include the scenario unit-based price plus recycling subsidy plus transaction costs as in this scenario all waste is recycled independent of the substitution elasticity between consumption goods. For each scenario, the demand for waste collection is slightly sensitive to the substitution elasticity. If the demand for consumption goods is elastic, private households will substitute agricultural and industrial goods by services and generate less waste. Figure 4-3 demonstrates that the unit-based pricing scheme is more effective in reducing waste generation than the upstream tax[10].

The demand for recycling services is barely affected by changes in the substitution elasticity, which is shown in Figure 4-4.

[10] The value of the substitution elasticity is calculated as the value of the benchmark substitution elasticity multiplied by a certain factor, where the value of the factor is shown on the x-axis.

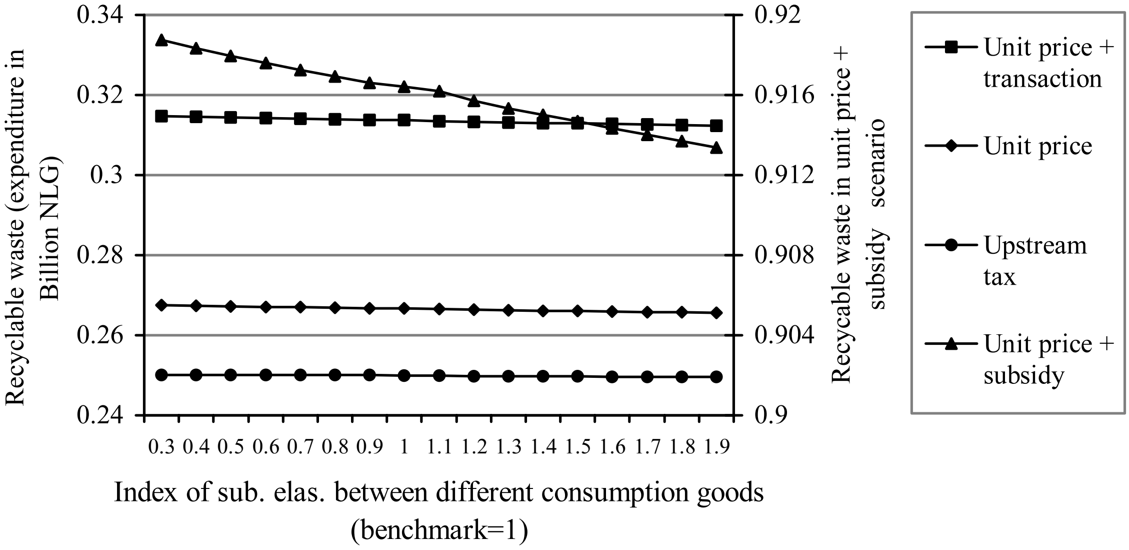

Figure 4-4 Sensitivity analysis of the substitution elasticity between different consumption goods: impacts on the generation of recyclable waste

A unit-based price provides households with an incentive to increase recycling. If less waste is generated, however, recycling will also slightly decline. An upstream tax stimulates private households to generate less waste, but does not stimulate recycling. Therefore, the demand for recycling services is not affected and remains at benchmark level for both the upstream tax scenario as the upstream tax plus transaction costs scenario. As the results are identical for these scenarios, only the results for the upstream tax scenario have been displayed in Figure 4-4.

Transaction costs

The total costs of implementing a policy will greatly determine the effectiveness of that policy. In the scenarios presented earlier, it was assumed that transaction costs would increase collection costs by 11%. The transaction costs, however, can also be far higher. To analyze how sensitive the results are, the transaction costs have been increased from benchmark level (100%) by 2.5 times as much. The results for the total quantity of waste generated, *i.e.* both recyclable waste and rest waste, are presented in Figure 4-5.

As can be seen from Figure 4-5, the upstream tax is more efficient in preventing waste than the unit-based price. Figure 4-5 demonstrates that if the transaction costs are large levying an upstream tax will prevent more waste than levying a unit-based price. The unit-based price combined with a recycling subsidy does not help the prevention of waste. The unit-based price, however, not only stimulates waste prevention but also waste recycling, as shown in Figure 4-6.

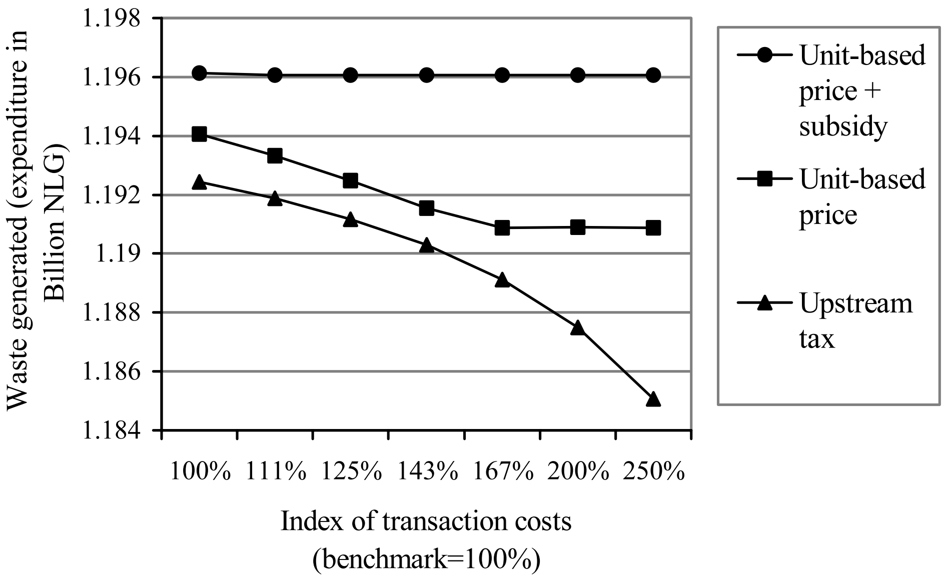

Figure 4-5 Sensitivity analysis of the transaction costs of implementing policy: impacts on the generation of total waste

The results for the individual categories: *waste recycling* and *waste collection* are shown in Figure 4-6. The larger the transaction costs become, the more waste will be recycled in the unit-based price scenario. If the costs become very large, all waste generated will be recycled.

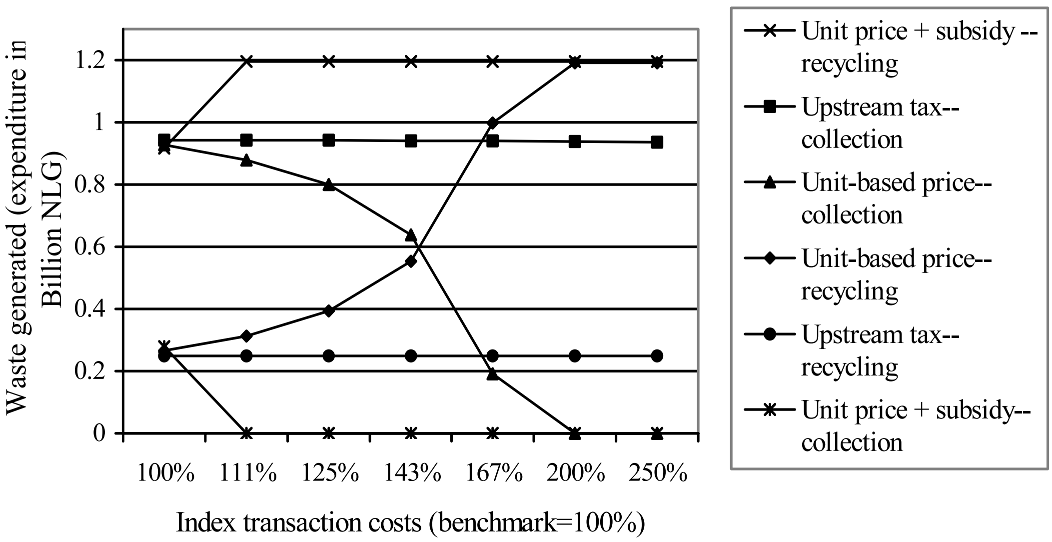

Figure 4-6 Sensitivity analysis of the transaction costs: impacts on the generation of recyclable waste and rest waste

Finally, the effects on the utility of the private households are shown in Figure 4-7. The effects on utility are nearly equal for both the unit-based price policy as the upstream tax policy if the transaction costs are small. However, if transaction costs increase, the upstream tax will have a far greater negative effect on the utility than the unit-based pricing scheme. If the transaction costs become too large in the unit-based pricing scheme, the private households will start to recycle all their waste, thereby eliminating all costs connected to the collection of waste. In the unit-based price scenario plus recycling subsidy, recycling is so cheap that all waste will be recycled and thus total consumption and utility will not be affected.

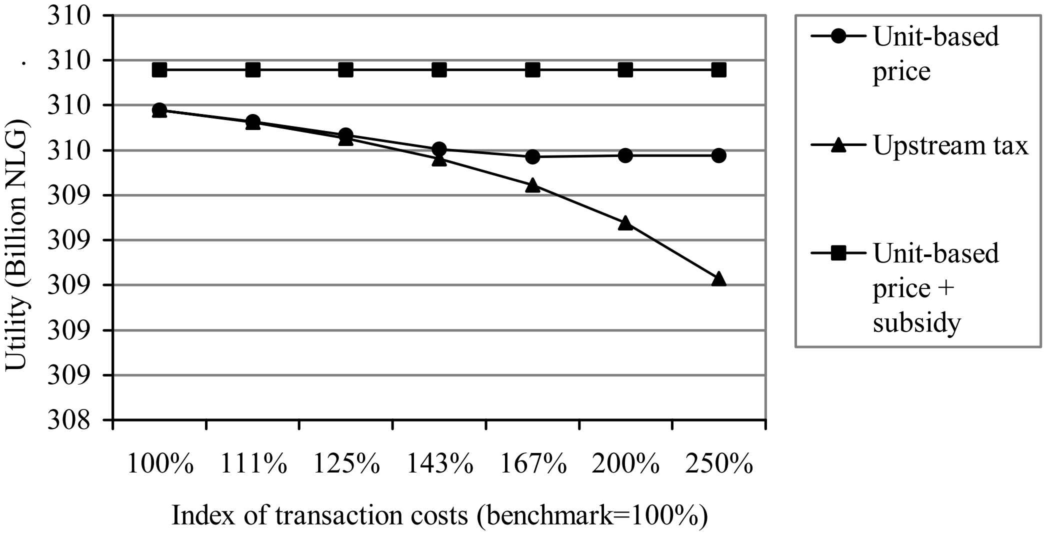

Figure 4-7 Sensitivity analysis of the transaction costs: impacts on the utility of the private households

4.4 Conclusions

In this chapter, we have demonstrated how a simple model simulating the waste market and incorporating market distortions can be built using an applied general equilibrium framework. One of the characteristics of the waste market is a flat fee-pricing scheme for waste collection. In such a pricing scheme, the marginal costs, or the price for waste collection as perceived by consumers equal zero. Special attention, therefore, has been given to modeling goods with a zero price. Introducing such a market distortion has strong effects on the results of the model. This was demonstrated by the application of the model in a numerical example.

Flat fee pricing for waste collection leads to the inefficiently high generation of waste. The effects of four waste management policies have been analyzed, *i.e.* a unit-based pricing scheme, recycling subsidy, unit-based pricing scheme combined with recycling subsidy and an upstream tax. The results show that a flat fee-pricing scheme for waste collection takes away the incentive to recycle. As long as a flat fee-pricing scheme is used, the private households do not have a price-incentive to reduce waste generation and thus households will show little tendency to increase recycling.

Making recycling more attractive by reducing the costs of recycling does not necessarily result in less waste generated. On the contrary, lower recycling costs result in the generation of more waste, due to the income effects of lower recycling costs. Only if policies promoting recycling are combined with a unit-based pricing scheme for waste collection will these policies be effective.

The results of the model clearly demonstrate that the unit-based pricing scheme is effective in providing consumers with an incentive to recycle and prevent waste, especially when combined with a recycling subsidy. In contrast to an upstream tax, the unit-based price offers consumers an incentive to both recycle and prevent waste.

The upstream tax can stimulate the private households to generate less waste if the demand for consumption goods is elastic. The upstream tax will not promote waste recycling, but only waste prevention because recycling still has a positive marginal price whereas collection has a zero marginal one. If the demand is inelastic, the policy change will have barely any effect.

Both a unit-based pricing scheme and an upstream tax will, however, also negatively affect utility and social welfare. Especially if transaction costs are considered, both utility of the private consumers and social welfare will be negatively affected. If the environmental gains of waste prevention and increased recycling are not large enough to offset the decrease in utility, neither policy option should be implemented.

Different polices have been compared in this chapter using a relatively simple example. For a more detailed assessment of waste management policies in the Netherlands, more data must be gathered. Also modeling issues, such as including environmental impacts and substitution possibilities between products within a sector, should be resolved.

In this analysis, it was assumed that the private households would pay all transaction costs of introducing a policy. In reality this may not be possible. Particularly the social costs of illegal disposal and the costs of preventing illegal disposal may render the unit-based pricing scheme less desirable than an upstream tax. Policy makers, however, should bear in mind that the upstream tax is far less efficient, especially since it does not stimulate consumers to start recycling waste.

Appendix 4-A: Solving a Negishi format

The Negishi model calculates the equilibrium through an iterative process. First the equilibrium is determined by solving the maximization model

$$TW = Max\sum_i \alpha_i u_i(x_i, r_i, w_i) \tag{E.1}$$

Subject to the balance constraint:

$$\sum_i \sum_g x_i^g + \sum_i r_i + \sum_i w_i \leq \sum_i \omega_i + \sum_j y_j, \qquad \perp p \tag{E.2}$$

The Negishi weights are initialized as follows:

$$\alpha_i = \frac{h_i}{\sum_i h_i} \tag{E.3}$$

This means that the Negishi weight of consumer *i* is determined by the initial share this consumer has in total income. If the share of consumer *i* in total income is large, the Negishi weight of that consumer is large and vice versa. It is assumed that the utility functions of both consumers are homothetic and commodity endowments are strictly proportional. Homothesticity ensures that the composition of a utility maximizing commodity is unaffected by the level of income. Due to this assumption, the social demand, *i.e.* the sum of individual demands, is proportional to the level of the total income, independent of its distribution. The competitive equilibrium prices and, therefore, the resulting allocation of resources is independent of income distribution. Thus, the problem of income distribution is assumed away (Negishi, 1972).

After the model is solved, the shadow price of each commodity is calculated. These shadow prices are used to calculate the income deficit of each consumer, *i.e.* the difference between total income and total expenditure of each consumer, labeled ‘loss’.

$$loss_i = p\omega_i - \sum_i px_i^g + pr_i + pw_i \tag{E.4}$$

If the loss for each consumer equals zero, the equilibrium solution is found. If the loss for one or more consumers is not equal to zero then the Negishi weights are adjusted as follows (Ginsburgh and Keyzer, 1997):

$$\alpha_i = \alpha_i + \beta \frac{loss_i}{\sum_i h_i} \tag{E.5}$$

For example, if a consumer has a surplus income, *i.e.* her income is larger than her expenditure, the Negishi weight will be increased. In the next iteration, the consumption of this consumer will be larger due to the larger Negishi weight. This iterative procedure results in a set of unique equilibrium Negishi weights and prices.

Appendix 4-B Definition of model indices, parameters, and variables

Indices

Label	Entries	Description
c	1	Private households
g	1...3	goods (agriculture, industry and services)
h	1,2	material (recycled and virgin)
i	1,2	Consumers (private households and government)
j	1...8	goods and services
n	1,2	waste treatment services (incineration and landfilling)
z	1...10	commodities (goods, services, capital and labor)

Parameters

Symbol	Description
α	Negishi weight
β	waste percentage
σ^{kl}	substitution elasticity between labor and capital
σ^{pi}	substitution elasticity between primary and intermediate inputs
σ^{wm}	substitution elasticity between materials and other intermediate inputs
σ^{vr}	substitution elasticity between virgin and recycled material
σ^{pw}	substitution elasticity between primary factors and waste treatment services
σ^{il}	substitution elasticity between incineration and landfilling services
ξ	subsidy wedge
A	technology parameter
F	flat fee for waste collection

$\overline{K}$	endowment of capital
$\overline{L}$	endowment of labor
LST	lump sum transfer to keep income of government constant
P	price
p_t	price including subsidy
S	transfer cost subsidy
T	gains upstream tax
Y^0	initial income

Variables

Symbol	Description
K	capital use
L	labor use
M	use material
TWF	total welfare
U	utility
W	use of waste treatment services
W	total generation of waste
X	consumption
X	total consumption
Y	production

5 Economic incentives and the quality of municipal solid waste: counterproductive effects through 'waste leakage'

5.1 Introduction

Economic literature suggests that externalities should be internalized by means of Pigovian taxation. The costs of treating waste generated by households, which can be seen as an externality of consumption, are not normally internalized in the price of waste collection. Most municipalities charge a fixed amount of money, the so-called flat fee, for the collection of waste. They tend to choose this pricing system because of its simplicity and low transaction costs. Unfortunately, this pricing system does not provide incentives to minimize waste generation. Since the price is fixed, marginal costs of waste generation equal zero. A better pricing system would be a variable price for waste collection, which is dependent on the actual amount of waste generated, the so-called unit-based price.

Recent studies have demonstrated that the introduction of a unit-based fee contributes to the solution of the solid waste problem, providing that due care is taken to prevent illegal forms of disposal, such as dumping and illegal burning (see, for example, Jenkins, 1993; Fullerton and Kinnaman, 1995, 1996; Palmer and Walls, 1997; Fullerton and Wu, 1998 and Choe and Frasier, 1999). Disposal taxes also provide an incentive for producers to make efficient choices about the degree of packaging, the weight and material input of the product, and finally its rate of recyclability (Fullerton and Wu, 1998). Municipalities throughout the world have experimented with the use of unit-based pricing schemes for waste collection. Results of these experiments can be found in, for example, Miranda *et al.* (1994), Sterner and Bartelings (1999), and Linderhof *et al*. (2001).

A unit-based pricing scheme is usually implemented for the collection of rest waste and sometimes for the collection of organic waste also. This pricing scheme is generally accompanied with policies for the separation of organic waste and the promotion of paper, glass, tin, and battery recycling. Research in the United States suggests that a unit-based price is far less successful if introduced without these recycling policies (Miranda *et al*., 1994). The results in Chapter 4 also support this.

Recent studies (*e.g.* Fullerton and Kinnaman, 1995 and Fullerton and Wu, 1998) have focused on the possibility of illegal disposal as a consequence of introducing a unit-based pricing system for waste collection. However, they failed to recognize another potential problem, namely the possibility of the 'pollution' of recyclable or organic waste. Households not only have the option of burning or illegally dumping trash, but they can also get rid of it in small amounts by putting it, for example, in organic waste bins or glass containers, both of which are collected free of charge. This kind of waste leakage can have serious effects. It will greatly increase the costs of recycling 'polluted' waste, since the recyclable waste has to be cleaned first. In the case of organic waste, the results are even worse. Heavily 'polluted' organic waste can no longer be composted. The quality of compost made from cleaned 'polluted' waste is too low. As a result, composting units do not accept this 'polluted' waste, but instead send it to an incinerator. This could eventually lead to all organic waste being incinerated or landfilled.

In the Netherlands, municipalities are obliged to collect organic waste and rest waste separately. Several large municipalities, however, have been granted an exemption from this obligation. The quality of organic waste collected in these municipalities was not good enough to be composted and thus it was not efficient to collect organic waste in large parts of these municipalities (WMC, 2003e). This suggests that waste leakage is a particular problem in larger municipalities.

Monitoring and preventing waste leakage is costly. Organic waste is usually collected in large garbage trucks where all waste is thrown together. This makes it difficult to distinguish the waste of one household from that of another. To locate the source of polluted organic waste, the quality of organic waste has to be checked during collection. This entails large transaction costs.

In this chapter, a general equilibrium model has been built to analyze the problem of waste leakage. We only focus on waste leakage effects for the organic waste stream, given that these effects are likely to present the most serious problems. The model structure, however, is such that it can easily be extended to include other waste streams, like paper and glass. Although this chapter focuses on waste leakage in a unit-based pricing system, waste leakage is potentially a problem in any system in which consumers are penalized for generating rest waste and rewarded for generating recyclable or organic waste.

Existing studies have analyzed the economic and environmental effects of policies aimed at reducing waste generation with the use of both partial and general equilibrium models[1]. Most recent studies have chosen the general equilibrium approach, *e.g.* Fullerton and Wu (1998) and Calcott and Walls (2002). One of the advantages of a general equilibrium approach over a partial equilibrium approach is that it is possible to model the entire product life cycle from production, to

[1] An overview of these studies is presented in Chapter 2.

consumption and finally to disposal. Any change in one of the stages of the life cycle can result in changes in other stages.

In this chapter, we will focus specifically on the quality of municipal solid waste. Consumers have an incentive to 'pollute' organic waste. The environmental preferences of households play an important role in deciding the quality of organic waste they want to generate. Households with little or no preference for a clean environment will have a stronger incentive to pollute organic waste than those households with a greater preference for a clean environment. This aspect will be implemented in the model by introducing two groups of consumers: 'green' consumers and 'traditional' consumers. A numerical example, based on data stylized for the Netherlands in the year 2000, will demonstrate that waste leakage can cause serious problems.

The chapter is structured as follows. Section 5.2 describes the applied general equilibrium model and shows how the problem of waste leakage can be included in such a framework. Section 5.3 presents the numerical example and demonstrates how a unit-based pricing system can inadvertently promote waste leakage. Section 5.4 concludes and offers some policy recommendations.

5.2 Modeling different waste categories

5.2.1 General introduction to the model structure

General equilibrium models can be built in several formats. In this chapter, we choose to build the model in the Negishi format, since this format is especially suited to the incorporation of price rigidities such as a zero marginal price[2]. In the Negishi format, the total welfare of an economy is maximized given constraints on the utility formation, production possibilities, and balance equations. The total welfare of the economy is specified as the weighted sum of the utilities of the individual consumers in the model. The utility of each consumer is weighted with the so-called Negishi-weight, which is determined in such a way that each consumer spends exactly its total income on the consumption of goods or savings[3].

In this section, the general model structure will shortly be discussed. The focus lies on the assumptions necessary to build a model that includes generation of three types of

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[2] Note that the choice of a format will not affect the equilibrium solution as mentioned in Chapter 1.

[3] See Chapter 4 for more information about the calculation of the Negishi weights.

waste, a flat fee pricing system and an endogenously determined labor supply[4]. To illustrate the problem of waste leakage clearly, the model has been kept as simple as possible. This makes it easy to follow the assumptions necessary to introduce waste leakage in a general equilibrium model.

The model characteristics are as follows. There are three consumers in the model: two types of private households and a government consumer. They can consume one ‘produced good’. Private households generate waste as a fixed percentage of consumption and they have to deal with this waste. They can either choose to put the waste in the waste bin or choose to separate organic waste from rest waste. The organic waste is then collected separately from the rest waste and sent to a composting unit (see Figure 5-1).

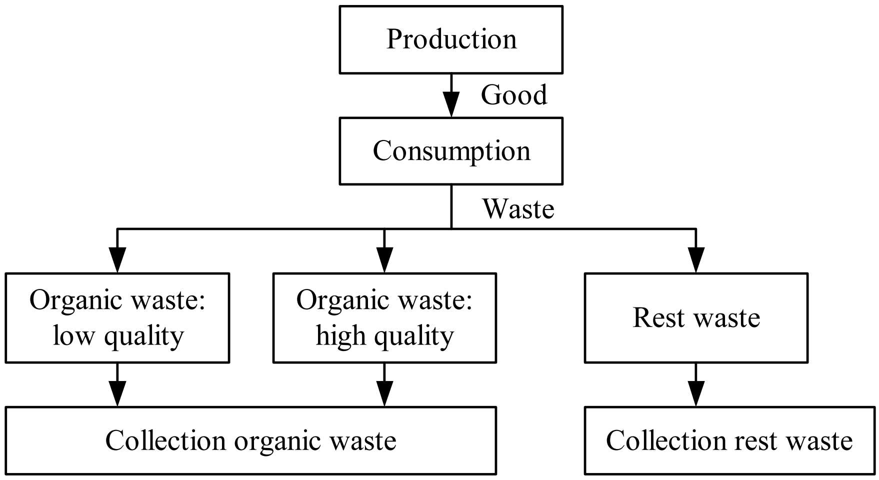

Figure 5-1 Representation of the basic model

Generating organic waste is costly for consumers because they have to invest labor in separating the organic waste from rest waste. Consumers may choose to generate low or high quality organic waste. The production of a high quality organic waste will cost more labor. This way of modeling organic waste quality simulates the situation wherein the households must incur costs to separate organic waste. They will have costs in the form of, for example, cleaning the organic waste bin or spending time on separating different waste streams. In the benchmark model, private households pay a flat fee for collection of all waste, including both organic and rest waste[5]. According to such a pricing scheme, the marginal costs of waste collection equal zero. This

[4] Labor supply equals the exogenously determined labor endowment minus the labor necessary to generate organic waste.

[5] In a flat fee-pricing scheme, consumers pay a fixed amount of money for the collection of waste, which is independent of the actual amount of waste that is produced. Therefore the marginal cost of producing one unit of waste is equal to zero.

means that the equilibrium prices for waste collection of rest waste and organic waste both equal zero. To implement this in the Negishi format, a subsidy-cum-tax scheme, as discussed in Chapter 4, is used. To recapitulate, in the subsidy-*cum*-tax scheme, consumers pay the equilibrium price for waste collection. The government, however, reimburses the consumers with exactly the same amount in the form of a subsidy, thus the price of waste disposal as perceived by the consumer equals zero. The government will finance the costs of the subsidy by demanding a flat fee or direct tax[6] from the private households for waste collection.

In the policy scenario, a unit-based pricing scheme is introduced for the collection of rest waste. This means that private households pay the equilibrium price for waste collection, which equals the marginal costs of producing these services. Consequently, both the subsidy and flat fee are abolished.

5.2.2 The model represented in equations

The model follows the general structure of an applied general equilibrium model in the Negishi format. Total welfare *(TW)* is, therefore, maximized. It depends on the weighted sum of the log of the utilities (u_i), of each consumer i with welfare weights (α_i). The welfare weights or Negishi weights are specific to the Negishi format. The values of the Negishi weights are determined in such a way that each consumer spends exactly its income on the consumption of goods[7]. (See appendices 5-A and 5-B for full model specification and notation):

$$TW = \sum_i \alpha_i \ln[u_i(x_i^g)] \tag{5.1}$$

The utility of consumer i depends solely on the consumption of the 'produced good' (x_i^g). This 'produced good' is an aggregate of all production sectors in the economy. Thus the model encompasses the whole economy, and qualifies as a general equilibrium model. Private households generate waste during consumption. For

[6] A direct tax only influences the income of the consumer and does not influence the consumption pattern.

[7] In the Negishi format, the equilibrium solution is found with the help of an iterative process. Given initial values for the Negishi-weights based on the income of a consumer, the model is solved and prices for each commodity are calculated as shadow prices. Subsequently, the budget constraint for each consumer is checked. If one or more consumers in the model spend more or less than their income, the Negishi weight for that consumer is adjusted. The model is then solved again with the adjusted Negishi-weights. The process continues until the budget constraints of all consumers hold. For more information about the Negishi-format and why the iterative process of the Negishi format results in a general equilibrium solution, the interested reader should consult Negishi (1972) or Ginsburgh and Keyzer (1997).

simplicity, the present model has deliberately been kept static, although we realize that waste generation has dynamic aspects; not all products will turn into waste immediately when they are consumed, for example, durables can continue to function properly for several years. In the comparative static model, waste generation (*W)* of consumer *c*, where *c* is a subset of *i* and contains only the two private households, is determined as a fraction β of the consumption product (x^g), where *g* stands for 'produced good' [8].

$$W_c = \beta\, x_c^g \tag{5.2}$$

The private households must deal with waste they generate by using the so-called waste collection services. They can either choose to demand collection services of rest waste *(*x^r*)* or collection services of organic waste *(*x^o*).* They can substitute demand of rest waste collection services for organic waste collection services. They can also choose between generating *low* quality organic waste *(*$x^{o,l}$*),* or *high* quality organic waste *(*$x^{o,h}$*),* as specified in the following nested CES function[9]:

$$W_c = CES(x_c^r, CES(x_c^{o,l}, x_c^{o,h}; \sigma_c^{l,h}); \sigma_c^{r,o}) \tag{5.3}$$

Where $\sigma^{l,h}$ stands for the substitution elasticity between low quality organic waste and high quality organic waste and $\sigma^{r,o}$ stands for the substitution elasticity between rest waste and organic waste.

We realize that by using a CES-substitution function the demand of waste collection services (in monetary terms) is not equal to the amount of waste generated (in Ktonnes). The amounts of waste services demand in Ktonnes are calculated on the basis of the calculated demand for waste collection services in monetary terms.

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[8] Implicitly this means that part of the used material will accumulate in the stock of durable goods. Therefore, at a given moment of time, the material inflow does not have to be equal to the material outflow in the model.

[9] The notation *z=CES(x,y;*σ*)* reflects the following function: $z = \left(x^{\frac{-(1-\sigma)}{\sigma}} + y^{\frac{-(1-\sigma)}{\sigma}} \right)^{\frac{-\sigma}{1-\sigma}}$.

As can be seen in the equation above, the substitution elasticity is the same for both variables. A CES function can also be nested. This means that, for example, the variable x in the equation above is in fact another CES-function. An advantage of the nested-CES function is that the elasticity of substitution in this case does not necessarily need to be the same for all variables in the function.

If private households decide to generate organic waste, they will expend labor (*lw)* in separating organic waste from rest waste. Producing high quality organic waste costs more labor than producing low quality organic waste.

The 'production possibility set' for the households for generating organic waste of quality $f\,(x_c^{\,f})$ is given as (f= *low, high*):

$$x_c^{o,f} \leq \mu^f \; lw_c^f \tag{5.4}$$

Where μ^f reflects the labor costs necessary to produce a unit of organic waste of quality f.

The three firms that are included in the model, *i.e.* producer of the consumption good and the producers of the two types of collection services, produce output q of good j under conditions of constant returns to scale, using as inputs capital *(k)* and labor *(l)*. For the sake of simplicity, we abstracted from the use of intermediate goods. The CES-production function for these firms is:

$$q_j = CES(k_j, l_j; \sigma_j^{k,l}) \tag{5.5}$$

Where $\sigma^{k,l}$ is the substitution elasticity between capital and labor.

Perfect competition for each producer in the model is implicitly assumed. This assumption, although restrictive, does not pose a problem in this model. Since the sector 'produced good' is an aggregated sector representing the all production sectors in the economy, it is natural to assume perfect competition for this sector. The municipalities, as collectors of waste, have no competitors. Households would have to move to be able to offer their waste to another collector. Municipalities, however, do not strive for profit on collection. Preferably, they charge households a price exactly equal to the marginal collection costs, just like the perfect competition assumption.

Consumers supply capital and labor to the firms. The capital supply (K), is exogenously determined. The labor supply (L) of consumer c, however, is calculated as the exogenous labor endowment $(\overline{L})$ minus the total amount of labor used for generating both types of organic waste (lw^f):

$$L_c = \overline{L_c} - \sum_f lw_c^f \tag{5.6}$$

Finally, the model is closed by several balance equation, which essentially state that the demand for any commodity, good, services, or production factor cannot exceed the supply of that commodity.

5.3 A numerical example

The model presented above is applied in a stylized example with numerical data from the Netherlands. The goal of this section is to demonstrate how the main mechanisms of the model operate and how these mechanisms are influenced by the assumptions inherent in the model. The economic data used in the numerical example are based on the Netherlands in the year 2000 (Statistics Netherlands, 2002b) and supplemented with data on waste collection (collection, fee and subsidy) derived from WCM (2000a, 2001, 2003d). To keep the model as simple as possible, we have chosen to give the government an income dependent on capital endowments instead of an income from taxes on labor and consumer goods[10]. This has been adjusted in the social accounting matrix.

5.3.1 Benchmark data

The accounting matrix displayed in Table 5-1 describes the initial equilibrium. The supplies of commodities, *i.e.* producers' output and consumer endowments, have positive values; demands of commodities, *i.e.* production inputs and consumption, have negative values[11].

In the benchmark data, private households pay a flat fee for the collection of both rest waste and organic waste. This fee covers about 95% of the actual cost of waste collection. Although the fee (and the cost-coverage rate) varies between different municipalities in the Netherlands, the average cost-coverage rate is around 95% (WCM, 2002).

Prices are normalized to unity according to the Harberger convention except for the prices for waste collection of rest waste and organic waste[12]. The demand for waste collection is here displayed in thousand tonnes instead of expenditures. The prices of

[10] Although less realistic, we feel that it is justified to make this assumption since our current focus is on waste generation by private households. As we are interested in finding a first-best equilibrium solution, we abstract from the existence of distortionary taxes in the same fashion as in Chapter 4.

[11] The entries in the column times the corresponding prices add up to zero to ensure that the zero profit condition holds: value of inputs equals value of outputs. The entries in the column of each consumer times the corresponding price adds up to zero to ensure that the budget constraint holds: each consumer spends exactly its income on the consumption of goods and services. The entries in each row times the corresponding prices adds up to zero to ensure that each market clears: total demand for each commodity must equal total supply.

[12] As in Chapter 4, we adopted the Harberger convention for all unknown prices. The Harberger convention consists of normalizing prices to unity. Thus quantities in the benchmark data represent expenditures, or how much of a good or factor one can buy for €1.

waste collection as shown in Table 5-1 are the average costs in million Euros of collecting a thousand tonnes of waste (WCM, 2002).

Table 5-1 Benchmark accounting matrix for the year 2000 (expenditures in million Euro and waste collection in thousand tonnes)

	Good	CS rest	CS organic	Traditional consumer	Green consumer	Gov	Colsum	Price
Good	759645	0	0	-358932	-172819	-227894	0	1
CS rest	0	3935	0	-3094	-841	0	0	0.269
CS organic	0	0	1457	-546	-911	0	0	0.269
Capital	-554112	-424	-157	220541	106186	227966	0	1
Labor	-205533	-636	-235	139323	67081	0	0	1
Fee	0	0	0	-932	-449	1380.0	0	1
Subsidy	0	0	0	980	472	-1452.0	0	1
Rowsum	0	0	0	0	0	0	0	

Note: 'Good' stands for the consumption good; 'CS rest' stands for collection services of rest waste, 'CS organic' indicates collection services of organic waste; 'Fee' is the flat fee consumers pay to the government for collection of waste, 'Subsidy' stands for the total amount of money the government gives for collection of waste as a subsidy to the consumers. The price column gives the prices of all commodities; Rowsum is sum of the entries in a column times the corresponding price, Colsum is the sum of the entries of a row times the corresponding price.

As explained in Section 5.2, two types of consumers are distinguished: a 'traditional' consumer and a 'green' consumer. Based on the MOSAIC system described in Beker (2002), the private households in the Netherlands have been divided into two consumer types[13]. According to our definition, consumers in the Netherlands may be divided into 32.5% 'green' consumers and 67.5% 'traditional' consumers.

Private households can generate three types of waste: rest waste, low quality organic waste and high quality organic waste. To generate organic waste, either high or low quality, consumers have to incur costs. They can generate rest waste for free. In this thesis, low quality organic waste and high quality organic waste are defined as follows: 1) high quality organic waste is organic waste without any pollution and thus 100% pure. We presume that low quality organic waste is somewhat polluted by organic waste and only 70% pure. Composting high quality organic waste will not

[13] The MOSAIC system divides consumers in the Netherlands into 10 groups and 41 types based on information about the neighborhood they live in and information about demographic, socio-economic and life-style factors. Based on information about the influence of social factors on the amount of waste generated (Sterner and Bartelings, 1999), these 10 groups were further aggregated into two types of consumers.

result in any residue, whereas composting of 100 tonnes of low quality organic waste will result in 30 tonnes of residue that has to be incinerated.

According to information obtained from both the Waste Management Council and the Ministry of Spatial Planning, Housing and the Environment, no recent research has been done to determine the overall quality of organic waste collected by municipalities. Composting units check the quality of organic waste beforehand and decide whether the waste delivered by the municipality can be composted or not. Their records, however, have not been released to the public. Therefore, it was impossible to determine just how much high and low quality organic waste is actually generated in the benchmark case. As an indication of the quality of organic waste, we used the amount of residue that is produced during the composting process. On average, composting of 100 tonnes of organic waste of a mixed quality results in 7 tonnes of rest waste (Beker, 2002). Given that composting of high quality organic waste results in no residue and that composting of low quality organic waste results in 30% residue, we can calculate that the overall mixture of organic waste consists of 23.3% low quality organic waste and 76.7% high quality organic waste.

According to the way in which ‘traditional’ and ‘green’ consumers are defined, we presume that ‘traditional’ consumers generate more low quality organic waste than ‘green’ consumers. Given that the overall composition of the waste stream consists of 23.3 percent low quality organic waste and 76.6 percent high quality organic waste and given that the households can be divided into 32.5% ‘green’ consumers and 67.5% ‘traditional’ consumers, the percentage of low and high quality waste generated by each consumer can be determined. Based on overall quality of the organic waste stream and the percentage of ‘green’ consumers and ‘traditional’ consumers, it can be calculated that in the benchmark case, the ‘green’ consumer generates 90% high quality organic waste and 10% low quality organic waste. The ‘traditional’ consumer generates 70% high quality organic waste and 30% low quality organic waste. The amounts of low and high quality organic waste generated by the two types of consumers are shown in Table 5-2.

The calculations on the quantity of labor necessary to generate organic waste are based on the estimation of social costs of waste handling as found in Bruvoll (1998). Bruvoll estimated the social costs of waste handling at 145 dollar per tonne of waste. The amount of time spent on waste handling per week was estimated at 30 minutes a week. This is comparable to other empirical studies like Sterner and Bartelings (1998) and Radetzki (2000)[14]. About 54% of all recyclable and organic waste collected consists of organic waste (WMC, 2003c). If one supposes that the costs are directly

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[14] See Chapter 2 for more information.

related to the amount of waste collected, the average costs of handling organic waste are equal to 78 dollars per tonne. We may thus expect that generation of low quality organic waste is less costly than generation of high quality organic waste, however the generation of both high and low quality organic waste will involve many of the same costs. For example, both high and low quality organic waste must be separated from rest waste and the organic waste bin will, in both cases, have to be cleaned. Therefore, we expect that generating high quality organic waste will be 10% more expensive than the average costs and generating low quality organic waste will be 10% less expensive than average costs. The actual labor costs in the benchmark case are presented in Table 5-2.

Table 5-2 Additional data about the generation of organic waste in the benchmark

	Traditional consumer	Green consumer
Low quality organic waste generated (thousand tonnes)	163.8	91.1
High quality organic waste generated (thousand tonnes)	382.2	820.0
Share of low quality organic waste in total amount organic waste	30%	10%
Units of labor necessary to generate 1 thousand tonnes of low quality organic waste	0.072	0.072
Units of labor necessary to generate 1 thousand tonnes of high quality organic waste	0.085	0.085
Total labor units spent on composting	44.3	76.1

Substitution elasticities for the different production sectors and the substitution possibilities between different types of waste are given in Table 5-3. The production sectors use capital and labor as inputs for production. They can substitute between the use of capital and labor. Based on Draper and Manders (1996), we choose a substitution elasticity of 0.8.

Differences in 'environmental' preferences are captured in the substitution elasticities between rest waste and organic waste and between high and low quality organic waste. The actual substitution elasticities used in the model are provided in Table 5-3.

Table 5-3 Substitution elasticities for production factors and waste categories

	Good	CS rest	CS organic	Traditional consumer	Green consumer
Substitution elasticity between capital and labor ($\sigma^{k,,l}$)	0.8	0.8	0.8		
Substitution elasticity between organic waste and rest waste ($\sigma^{r,o}$)				0.6	0.3
Substitution elasticity between low and high quality organic waste ($\sigma^{l,h}$)				0.9	0.1

Note: 'Good' stands for the consumption good; 'CS rest' stands for the collection of rest waste and 'CS organic' stands for the collection of organic waste.

Consumers have the option of substituting rest waste for organic waste and low quality organic waste for high quality organic waste. On average, the stream of rest waste contains about 32% of organic waste. Substituting rest waste for organic waste does not mean that consumers make organic waste from, for example, a tin can by using labor as an input, but that they either separate organic waste from rest waste, *i.e.* throwing vegetable waste in the organic waste bin instead of in the rest waste bin, or that they throw rest waste in the organic waste bin and thus pollute the organic waste stream. To ensure that households do not generate high quality organic waste out of thin air, upper values on the extra amounts of high quality organic waste that can be generated have been set, based on the average amount of organic waste in the stream of rest waste.

Several studies have estimated the own price elasticity of the generation of rest waste. In this chapter, the substitution elasticities between rest waste and high and low quality organic waste have been based on the price elasticity estimated by Fullerton and Kinnaman (1996). They established an own price elasticity of –0.058. Other estimates of price elasticities can be found in Hong *et al.* (1993), Wertz (1976), Jenkins (1993) and Linderhof *et al.* (2001) (see Chapter 2 for more information)[15].

Since no information is available about the quality of organic waste, it is difficult to calibrate the substitution elasticity between low and high quality organic waste. Based on our expert opinion we assert that the demand is inelastic and thus the substitution elasticity is smaller than unity. By definition, the substitution elasticity will be larger for the 'traditional' consumer than for the 'green' consumer.

5.3.2 Results

The model, as specified in Section 5.2, is used to calculate the effects of the introduction of a unit-based pricing scheme for the collection of rest waste. This means that private households will have to pay the equilibrium price, which equals the marginal costs of producing this service, for the collection of rest waste. Private households will still pay a flat fee for the collection of organic waste.

Comparing the benchmark situation, which includes the flat fee-pricing scheme for waste collection, to the unit-based price scenario, which includes the unit-based pricing scheme for waste collection gives an indication of the expected results of introducing such a policy change. In Table 5-4, the changes in the main variables are presented.

[15] The own-price elasticity (ε) of good i is equal to: $\varepsilon = -\sigma + (\sigma - 1)e_i$, where σ is the substitution elasticity and e_i is the proportion of expenses for good i.

Table 5-4 Results for the main variables in the 'flat fee scenario' and 'unit-based price scenario' (expenditure in million Euro and waste collection in Ktonnes) and the percentage change as compared to the benchmark case

	Flat fee		Unit-based fee			
	Traditional consumer	Green consumer	Traditional consumer		Green consumer	
Consumption good	358932	172819	358903	(-0.01%)	172813	(0.00%)
Collection rest waste	3093.66	841.34	2832.32	(-8.45%)	726.11	(-13.70%)
Collection organic waste	545.94	911.06	806.99	(47.82%)	1026.23	(12.64%)

As shown in Table 5-4, introducing a unit based pricing scheme for collection of rest waste has a significant effect on the demand for collection of both rest and organic waste. These results are as expected: since organic waste may be collected free of charge, households will start to substitute the more expensive rest waste with organic waste. This holds especially for the 'traditional' consumer who has a large substitution elasticity between rest waste and organic waste. The introduction of a unit-based price, however, also has the undesirable effect of increasing the quantity of low quality organic waste (Table 5-5).

Table 5-5 Results for organic waste categories for the 'flat fee' scenario and 'unit-based price' scenario in Ktonnes (in brackets % change as compared to flat fee)

	Flat fee		Unit-based fee			
	Traditional consumer	Green consumer	Traditional consumer		Green consumer	
Low quality organic waste	163.78	91.11	267.57	(63.37%)	104.08	(14.24%)
High quality organic waste	382.16	819.95	539.41	(41.15%)	922.15	(12.46%)
Share of low quality organic waste	30.00	10.00	33.16	(10.53%)	10.14	(1.40%)

Private households start to produce more *low quality* organic waste instead of *high quality* organic waste (see Table 5-5). Substitution of *rest* waste for *low quality* organic waste is especially evident among 'traditional' consumers. 'Green' consumers, who have more concern for the environment, increase both their production of *low quality* organic waste and their production of *high quality* organic waste.

Since both types of organic waste are collected together, the share of *low quality organic waste* will greatly affect the overall quality. If the amount of *low quality organic waste* is relatively large compared to the amount of *high quality organic waste,* then the overall quality of the organic waste will be low. If unit-based pricing is introduced, on average about 6.7% of organic waste collected will consist of rest waste. In the flat fee pricing system, only 5.5% of organic waste consists of rest waste.

The 'green' consumer does not pollute the organic waste stream as much as the 'traditional' consumer. The percentage of rest waste thrown away with organic waste is about 3.1%. This percentage is constant for both the flat fee pricing system and the unit-based pricing system. 'Traditional' consumers contribute more rest waste to the organic waste stream. The percentage of rest waste thrown away with organic waste increases from 9.8% to 11%. IPH (1995) shows that composting units generally will not accept such low quality organic waste as generated by the 'traditional' consumers. This means that the composting unit will reject waste, which is collected in districts with relatively many 'traditional' consumers. Waste in this case must either be incinerated or landfilled, which increases waste treatment costs. Since actual composting is not included in this model, it is impossible to predict how much the waste treatment costs will increase due to waste leakage. In Chapter 6, an analysis of these costs is made.

The total amount of waste generation is not affected by the policy change. As prevention is not included in the model, consumers can only reduce total generation of waste by consuming less. As the income of the consumers is only minimally affected by the policy change, the consumers will not reduce consumption and thus total waste generation remains constant. Table 5-5 illustrates that the share of low quality organic waste has increased. The overall share of low quality in the total organic waste stream has risen from 17% to 20%. The unit-based pricing scheme leads to waste leakage and may therefore not be suitable to provide the correct incentives to private households to minimize waste generation.

5.3.3 Sensitivity analysis

This section deals with the sensitivity of the model. The results as presented in the Section 3.3.2 depend largely on the parameters used. Three parameters in particular are difficult to measure or estimate and therefore require careful examination. These are: (*i*) the substitution elasticity between rest waste and organic waste; (*ii*) the substitution elasticity between low quality organic waste and high quality organic waste; (*iii*) the labor cost of generating low and high quality organic waste.

The following procedure has been used for the sensitivity analysis. The value of the parameter is changed in a number of (equidistant) steps from the lower to the upper value of the range of the sensitivity analysis. The effects of these parameter changes are calculated for all variables of the model. The impact of the variables, *rest waste*, *low quality organic waste* and *high quality organic waste,* are presented below.

Substitution elasticity between rest waste and organic waste

As empirical studies report several different values for the price elasticity of rest waste (see Chapter 2), in this study a sensitivity analysis was performed for the substitution elasticity between rest waste and organic waste. This substitution elasticity is lower for the 'green' consumers than for the 'traditional' consumers. We have chosen to keep the ratio of the substitution elasticities between both consumers constant. Figure 5-2 shows the results of the sensitivity analysis. All other parameters are kept constant at benchmark levels.

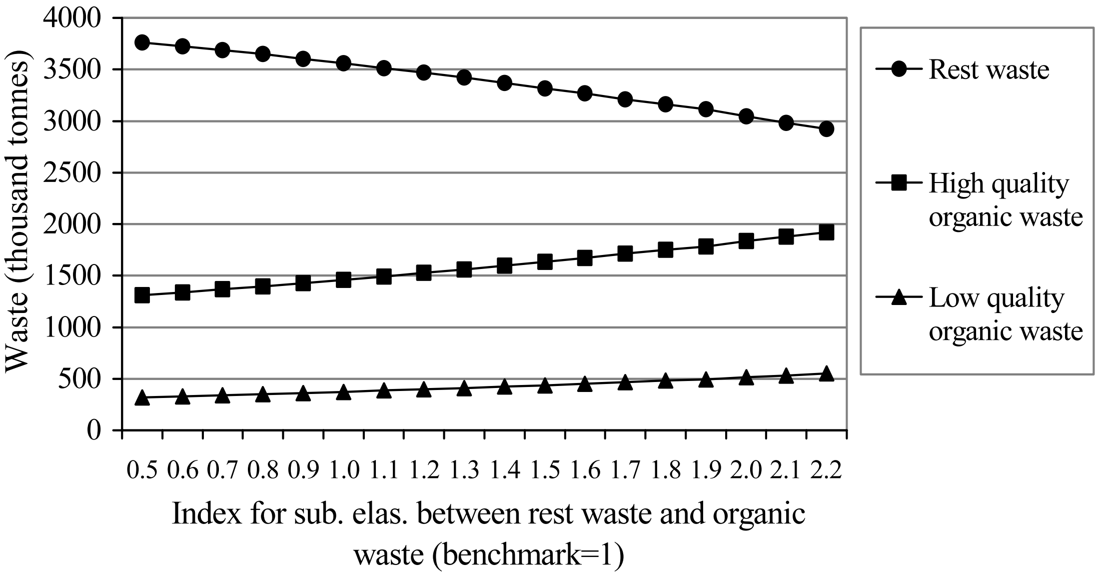

Figure 5-2 Sensitivity analysis for the substitution elasticity between rest waste and organic waste: impacts on quantities of waste

Note: The substitution elasticity is calculated as the benchmark substitution elasticity (0.6 and 0.3 respectively) multiplied by a factor δ. The value of factor δ is shown on the x-axis.

The substitution elasticity between rest waste and organic waste determines how much rest waste is generated. If the substitution elasticity is quite high then only 2900 thousand tonnes of *rest waste* will be generated. If, however, the elasticity is quite low the amount of *rest waste* increases to about 3750 thousand tonnes. Note that both the amounts of *low quality organic waste* and *high quality organic waste* grow as the substitution elasticity increases, which means that consumers are both separating more waste and discarding part of their rest waste into the organic waste bin.

Substitution elasticity between low quality and high quality organic waste

The second parameter that is examined is the substitution elasticity between low quality and high quality organic waste. As there is no data on how to calculate the value of this parameter, it is essential that a sensitivity analysis for this parameter is done. This substitution elasticity differs between the two consumers and the ratio between these two elasticities has been kept constant. The results of this sensitivity analysis are shown in Figure 5-3.

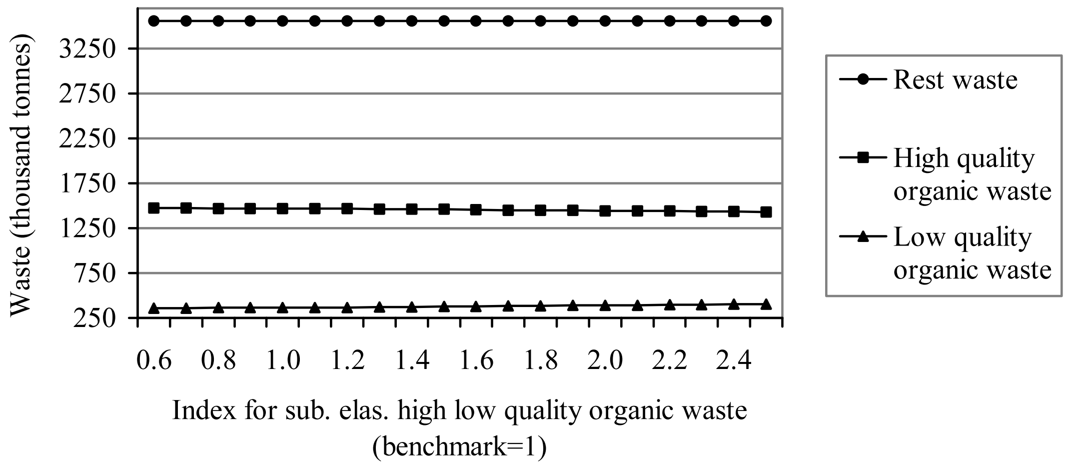

Figure 5-3 Sensitivity analysis for the substitution elasticity between low and high quality organic waste: impacts on quantities of waste

Note: The substitution elasticity is calculated as the benchmark substitution elasticity (0.9 and 0.1 respectively) multiplied by a factor δ. The value of factor δ is shown on the x-axis.

The amount of waste generated is barely sensitive to the substitution elasticity between low and high quality organic waste. When this substitution elasticity increases, a little more low quality organic waste is generated and marginally less high quality organic waste. The amount of rest waste generated is not affected at all. Obviously it is mostly the substitution elasticity between rest waste and organic waste that determines how much rest waste is transformed in high and low quality organic waste.

Labor cost organic waste

Another parameter that can affect the efficiency of the policy change is the actual labor cost of producing low quality and high quality organic waste. In the benchmark case, it is assumed that 0.072 units of labor are necessary to produce one unit of high quality waste (units in million tonnes) and 0.085 units of labor to produce one unit of low quality organic waste. This means that the labor costs parameter has a value of 13.89 for low quality organic waste and a value of 11.76 for high quality organic waste. The higher the labor costs parameter, the lower the actual labor costs. In this sensitivity analysis, the labor costs are varied from 56 percent to 333 percent of the benchmark level, this means that the value of the labor costs parameters are varied from –150% to 190%. The proportional difference between costs of generating low and high quality organic waste, however, is maintained. The results are presented in Figure 5-4.

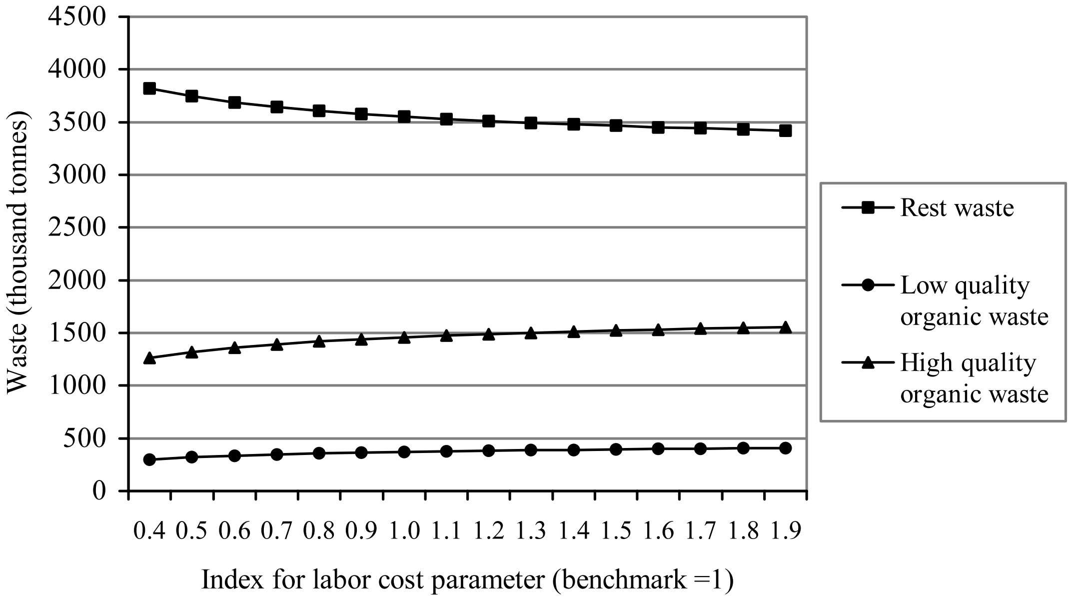

Figure 5-4 Sensitivity analysis labor cost organic waste: impacts on quantities of waste

Note: The labor cost parameter is calculated as the benchmark labor cost parameter (13.89 and 11.76 respectively) multiplied by a factor δ. The value of factor δ is shown on the x-axis.

As expected, the lower the labor costs, the higher the organic waste generation, and thus the lower the amount of rest waste generated. It is remarkable that the rate of substitution between rest waste and organic waste changes when the labor costs become higher. If the labor costs are high, a small change will result in a far larger change in the substitution rate than if the labor costs are low.

Interaction between labor costs of composting and substitution elasticity between rest waste and organic waste

Finally, the interaction between the labor cost of generating organic waste and the substitution elasticity between rest waste and organic waste was examined. In Figure 5-5, the impact on the generation of rest waste is shown. The quantity of rest waste that is generated is greatly affected by the substitution elasticity between rest waste and organic waste. The higher the substitution elasticity, a lesser the amount of rest waste is generated. The labor costs greatly affect the production of rest waste, but only if the substitution elasticity between rest waste and organic waste is fairly large. For small elasticity levels, the amount of rest waste is hardly affected. The higher the elasticity, the greater the impact of labor costs.

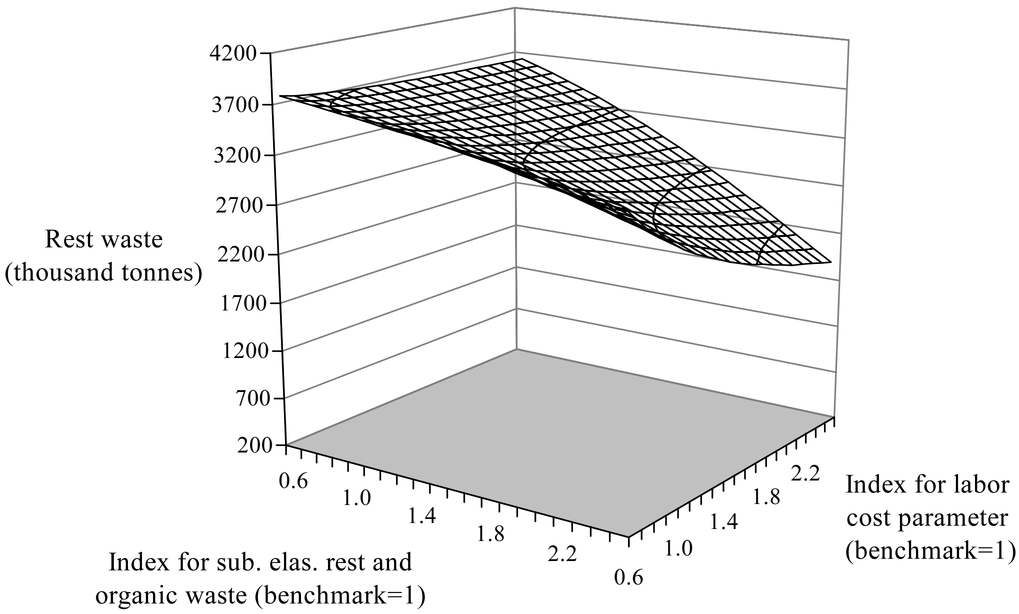

Figure 5-5 Interaction between labor cost and substitution elasticity between rest waste and organic waste: impacts on quantity of rest waste

Figure 5-6 shows the quantity of low quality organic waste generated. The effects are even more profound for the generation of low quality organic waste.

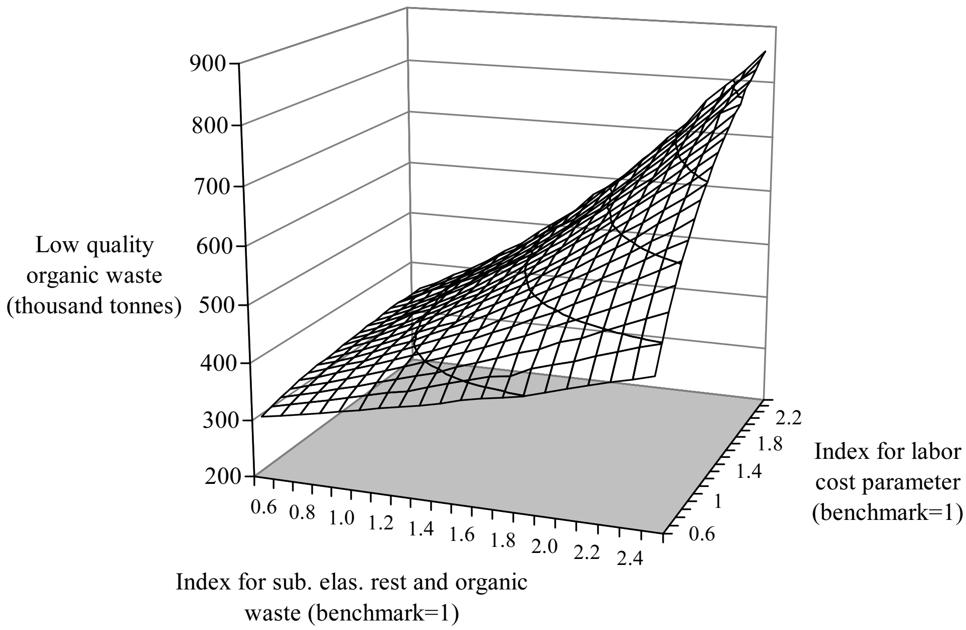

Figure 5-6 Interaction between labor cost and substitution elasticity between rest and organic waste: impacts on quantity of low quality organic waste

If we introduce higher labor costs in combination with small substitution elasticity levels, the amount of low quality organic waste generated increases by about 30% when a unit-based price is implemented. If we introduce higher labor costs combined with a large substitution elasticity, the amount of low quality organic waste increases by nearly 250%.

5.4 Discussion and conclusions

In this chapter, the effects of introducing a unit-based pricing scheme for the collection of rest waste have been analyzed. A general equilibrium model in the Negishi format was presented, with the possibility of generating three types of waste. In the model the consumers were given the possibility of substitution for these different types of waste, *i.e.* high quality organic waste, low quality organic waste and rest waste. With this model it is possible to illustrate some important aspects of the waste sector and the dangers of waste leakage. Waste leakage occurs when households throw rest waste into the organic waste or recyclable waste stream, thus polluting the organic and recyclable waste stream and making recycling or composting of this waste streams more expensive or impossible.

The results demonstrate that introducing a unit-based pricing scheme may lead to significant waste leakage. Waste leakage will greatly influence the effectiveness of the policy change. The ‘traditional’ consumers, who have less preference for a clean environment, are given strong incentives to discard part of their rest waste in the organic waste stream, thereby creating low quality organic waste, which will be more difficult to compost.

The sensitivity analysis shows that these results are quite sensitive to several parameters in the model. The labor cost of separating organic waste will especially influence how much rest waste is substituted by low quality organic waste. The substitution elasticity between organic waste and rest waste also influences the effectiveness of the policy. The substitution elasticities between low and high quality organic waste hardly have an effect on the optimal solution of the model.

This chapter only investigated how much waste leakage would occur when unit-based pricing was introduced. It did not explore how the composting costs would be affected. Therefore, it is difficult to say how the benefits of having less rest waste compare to extra composting costs due to lower quality of waste. In future research, the waste treatment sector should be included in the model in greater detail. Furthermore, it would be interesting to empirically determine several parameters concerning organic waste, such as the elasticity between low and high quality waste and the amount of low and high quality waste generated.

Waste leakage occurs when a unit-based price is introduced for the collection of rest waste. How much waste leakage occurs depends to a large extent on the type of consumers living in the municipality. Especially larger municipalities or mega-cities with a relatively large share of 'traditional', consumers can expect problems concerning waste leakage. As waste leakage leads to a decline in the quality of organic waste, potentially resulting in the situation where organic waste can no longer be composted but instead must be landfilled or incinerated, unit-based pricing will be less environmentally beneficial than might be expected. The results show that 'traditional' consumers pollute the organic waste stream to such as extent that it can no longer be composted. Since waste is collected in relatively small quantities, about 28 tonnes per garbage truck, it can be expected that districts with a higher percentage of 'traditional' consumers will generate organic waste that cannot be composted. It is therefore advisable for unit-based pricing to only be introduced in municipalities that have a large share of 'green consumers'. It is expected that substantial waste leakage will occur, particularly in the larger municipalities, since these municipalities have many districts with considerably more 'traditional' consumers.

As a unit-based pricing scheme is very efficient for lowering the amount of rest waste generated, it seems desirable that technologies, which enable us to separate rest waste from organic waste and produce high quality compost of cleaned organic waste, be invested in. Although it is already possible to mechanically separate recyclable and organic waste from rest waste (see Oorthuys and Brinkmann, 2000), the application of this technique is still in an experimental phase. It would be advisable for more to be invested in the development of such technologies. Only if this technology were available at low costs, would the implementation of unit-based pricing on a larger scale be recommended.

Appendix 5-A Specification of relevant equations

5-A.1 Model specification for a fixed fee for the collection of organic and rest waste

Welfare

Total welfare function:

$$TWF = \sum_i \alpha_i \ln(u_i) + \xi^r TD^r + \xi^c TD^o \qquad \text{(A.1)}$$

Where utility of consumers depends on the consumption of the consumer goods [16]

$$u_i \leq x_i^g \qquad \text{For } i = 1....3 \qquad \text{(A.2)}$$

In the benchmark case consumers have no price incentive to generate organic waste. As data for the Netherlands show that organic waste is generated even if municipalities charge a flat fee, organic waste generation has been included in the benchmark case. To ensure that organic waste is generated in the benchmark case, two extra consumers are introduced, who only derive benefits from the generation of organic waste. This modeling trick can be seen as if a consumer derives utility from generating organic waste due to the fact that the consumer is doing something beneficial for the environment.

$$u_i \leq CES(x_i^{o,l}, x_i^{o,h}; \sigma^{l,h}) \qquad \text{for } i = 4{,}5 \qquad \text{(A.3)}$$

Production function goods and collection services

$$q^j = CES(k^j, l^j; \sigma_{kl}^j) \text{ for each j} \qquad \text{(A.4)}$$

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[16] As mentioned in Section 5.2, to include the subsidy-cum-tax scheme in the model the total cost of the per unit subsidy (ξ) on waste collection of both rest waste ($\xi^r TD^r$) and organic waste ($\xi^o TD^o$) has to be added to the total welfare function due to technical reasons. This is done solely to change the marginal prices of waste collection. See Chapter 4 for more information on this subject.

Market clearance

Goods market balance:

$$\sum_i x_i^g \leq q^g \qquad \perp p^g \qquad \text{(A.5)}$$

By taking the marginal value of the balance constraint, the price vector p can be determined (this is symbolized by $\perp p$). The price vector is used in calculating the budget constraint and in determining the Negishi weights.

Capital market balance

$$\sum_j k^j \leq \sum_i \overline{K_i} \qquad \perp p^k \qquad \text{(A.6)}$$

Labor market balance:

$$\sum_j l^j \leq \sum_i L_i \qquad \perp p^l \qquad \text{(A.7)}$$

Where labor supply is determined by:

$$L_i = \overline{L_i} - \sum_f lw_i^f \qquad \text{for } i = 1....3 \qquad \text{(A.8)}$$

Collection rest waste market balance:

$$\sum_i x_i^r \leq TD^r \qquad \perp p_{sub}^r \qquad \text{(A.9)}$$

$$TD^r \leq q^r \qquad \perp p^r \qquad \text{(A.10)}$$

Collection organic waste market balance:

$$\sum_i x_i^{o,l} + \sum_i x_i^{o,h} \leq TD^o \qquad \perp p_{sub}^o \qquad \text{(A.11)}$$

$$TD^o \leq q^o \qquad \perp p^o \qquad \text{(A.12)}$$

Waste equations

Waste generation as percentage consumption:

$$W_c = \beta x_c^g \qquad \text{(A.13)}$$

$$W_i = CES(x_i^r, CES(x_i^{o,l}, x_i^{o,h}; \sigma_i^{l,h}); \sigma_i^{r,o}) \tag{A.14}$$

$$x_c^{o,f} \le \mu^f \;\; lw_c^f \tag{A.15}$$

Calculation Negishi weights

Budget constraint consumer *i*:

$$p^k \overline{K_i} + p^l L_i - F_i - LST_i = p^g x_i^g + p_{sub}^r x_i^r + p_{sub}^o x_i^o \quad \text{for } i\text{=1,2} \tag{A.16}$$

Budget constraint government:

$$p^k \overline{K_i} + F_i - T + \sum_i LST_i = p^g x_i^g \qquad \text{for } i\text{=3} \tag{A.17}$$

Total expenditure government is kept constant at benchmark level. If for any reason expenditure would change, then the income of the government is compensated through lump sum transfers. Where:

$$LST = Y_{gov}^o - (p^k \overline{K_i} + \sum_i F_i - T) \qquad \text{for } i = 3 \tag{A.18}$$

Total cost subsidy calculated as a lump sum transfer:

$$T = \xi^r \sum_i x_i^r + \xi^o \sum_i \sum_f x_i^{o,f} \tag{A.19}$$

5-A.2 Model specification including a unit-based pricing scheme for the collection of rest waste

In the unit-based pricing scheme model the following equation are changed:

Total welfare function (replaces equation A.1):

$$TW = \sum_i \alpha_i \ln(u_i) + \xi^o TD^o \tag{A.20}$$

Market balance constraint for collection rest waste (replaces equations A.9 and A.10) :

$$\sum x_i^r \le q^r \qquad \perp p^r \tag{A.21}$$

Budget constraint consumer *I* (replaces equation A.16):

$$p^k \overline{K_i} + p^l L_i - F_i - LST_i = p^g x_i^g + p^r x_i^r + p_{sub}^o x_i^c \qquad \text{for } i = 1,2 \tag{A.22}$$

Where fee refers to the fee for collection of organic waste only.

Budget constraint government (replaces equation A.17):

$$p^k \overline{K_i} + \sum_i F_i - T + \sum_i LST_i = p^g x_i^g \qquad \text{for } i = 3 \tag{A.23}$$

Total cost subsidy on collection of organic waste calculated as a lump sum transfer (replaces equation A.19):

$$T = \xi^{cc} \sum_i \sum_f x_i^{c,f} \tag{A.24}$$

Appendix 5-B Definition of indices, parameters, and variables

Indices

Label	Range	Description
c	1…2	traditional and green consumer
f	1...2	quality organic waste (low, high)
i	1...5	Consumers
j	1...3	goods (consumer good *g*, collection service rest waste *r*, collection service compost waste *c*)
sub	1	Subsidy
z	1...5	commodities (goods, capital and labor)

Parameters in GAMS specification

Symbol	Description
α	Negishi weight
β	waste percentage
μ	labor cost for generating organic waste

$\sigma^{k,l}$	substitution elasticity between labor and capital
$\sigma^{l,h}$	substitution elasticity between low and high quality organic waste
$\sigma^{r,o}$	substitution elasticity between rest waste and organic waste
ξ	subsidy wedge
F	flat fee for waste collection
$\overline{K}$	endowment of capital
$\overline{L}$	endowment of labor
LST	lump sum transfer to keep income of government constant
p	Price
P_c	Price including subsidy
T	Total costs subsidy waste collection
Y^0	initial income

Variables in GAMS specification

Symbol	Description
k	capital use
l	Labor use
lw	Labor used for generation of organic waste
q	production
TD	Total demand for waste collection services
TW	Total welfare
u	utility
W	Total generation of waste
x	consumption

6. Modeling economies of scale, transport costs and the location of waste treatment units in a general equilibrium framework

6.1 Introduction

The waste market is rapidly changing. In the Netherlands, one can observe a trend towards an increasing scale, concentration, and vertical integration, alongside a movement towards a multi-utility structure (WMC, 2002). The increasing scale of waste treatment units has diminished the influence of municipalities in the management of these companies. Where waste treatment was once mostly a municipal affair, it has become far more privatized. Energy companies have also become interested in merging with waste treatment units thus creating multi-utility companies. Waste treatment companies are interested in vertical integration; several of the largest companies are currently focusing on providing all services connected to waste treatment, from collection to final disposal. Moreover, international companies have become interested in providing their services throughout the whole of Europe (WMC, 2002).

Until 2000, Dutch laws prohibited the transportation of waste over provincial boundaries. Since January 2000, these laws have been revoked, which means that municipalities and companies are now allowed to transport waste outside the boundaries of their province and offer it to any waste treatment unit in the Netherlands. This means that more competition has been introduced into the waste market. Furthermore, the boundaries between countries in the European union have ceased to exist for transport of recyclable waste and are expected to disappear for combustible waste, which may also ultimately result in a reorganization of the waste market.

All of these developments have had important consequences for the waste management problem. Not only should the quantity of waste generated be minimized as cost-efficiently as possible, but it is also important that by choosing the cheapest waste treatment unit the waste treatment costs are reduced. Larger waste treatment units are able to treat waste against lower costs due to economies of scale (see, for example, Oorthuys, 1995 and WMC, 2003e). Larger units could also reduce harm to the environment at lower costs per unit of waste treatment. Naturally, this also means that waste will, in general, be transported over longer distances thus incurring both greater transport costs and more transport emissions.

Thus far the spatial aspects of the waste treatment problem have been analyzed mostly from within an optimization setting. Existing studies have focused on finding the optimal location of a waste treatment unit in order to minimize the waste treatment costs for society. Examples of these kinds of studies can be found in Opaluch *et al.* (1993), Macauley *et al*. (2002) and Ye and Yezer (1997).

These studies focus on determining the optimal waste treatment method given a certain quantity of waste. The quantity of waste generated, however, should not be taken as a given. Waste generation will to some extend depend on how waste is treated and how large the costs of waste treatment are. If waste treatment becomes more expensive, industries, for example, will start to recycle more of their waste to prevent waste disposal costs. To a lesser extent, households will also start to reduce waste generation. Households faced with a higher disposal tax will not only try to prevent waste generation by recycling and composting waste at home, but also by increasing the illegal disposal of waste. In turn, the quantity of waste generated affects the optimal choice between waste treatment options and the optimal location and size of the waste treatment units. A small quantity of waste may be treated in a local waste treatment unit; a larger waste stream or a waste stream of lower quality[1] may have to be sent to a larger waste treatment unit.

This chapter presents a model, which simulates the market for municipal solid waste. In this model, several municipalities are distinguished. Waste is generated in each municipality. This waste must be composted, incinerated, or landfilled. Depending on the price of waste treatment, transport costs, and emission restrictions, municipalities decide how and where to treat the waste.

Since we feel that the interaction between waste generation, the choice of waste treatment and choice of the optimal location of waste treatment units can best be analyzed in a general equilibrium context, we choose to build a general equilibrium model. The possibility of modeling the interaction between several markets is the main advantage of this type of model. Special attention will be paid to how the spatial aspects of the waste management problem can be included in a general equilibrium framework. The novelty in this approach is the focus on how the choice of a certain waste treatment unit is simultaneously affected by economies of scale, transport costs, quality of waste, emission restrictions, and policies aimed at reducing waste generation.

[1] The quality of the waste stream can be very important in determining the choice between waste treatment options. For example, organic waste heavily polluted with rest waste will be expensive or impossible to compost. In such a case, it may be necessary to incinerate the waste instead of composting it (see, for example, Chapter 5).

This model will be applied to a stylized setting of the waste market in a region of the Netherlands. This example will illustrate how the quality and quantity of waste generated significantly influences the choice of waste treatment methods, their size, and location. Furthermore, it will be demonstrated that the choice of optimal waste treatment methods, size, and location can significantly influence the cost-effectiveness of a policy change. A unit-based price for the collection of rest waste will be introduced and we will show that although such a policy change reduces waste generation, as recent studies such as Fullerton and Wu (1998) and Choe and Frasier (1999) have demonstrated, it will also decrease the quality of organic waste, thus increasing the composting and transport costs of waste. The model presented in this chapter is applied to the municipal solid waste market in the Netherlands, but is written in general terms and can easily be applied to the municipal solid waste market in any other industrialized country.

The structure of this chapter is as follows: Section 6.2 introduces the model specification. Section 6.3 describes the data used and presents the results of different scenarios. Section 6.4 concludes.

6.2 Modeling the spatial aspects of the municipal solid waste problem

6.2.1 General introduction to the model structure

The model presented in this section is an extended version of the model used in Chapter 5. The main aspects of the model will shortly be discussed and more attention will be paid to the new elements, namely economies of scale, transport costs, and emission rights. The structure of the model is illustrated in Figure 6-1.

We distinguish only one producer who makes a ‘produced’ good. The private households and the government consume the ‘produced’ good. Four municipalities are distinguished in the model. In each municipality, two types of consumers: a ‘traditional’ consumer and a ‘green’ consumer are modeled, just as in Chapter 5. The ‘traditional’ consumer has little preference for the environment. The ‘green’ consumer has some preference for a clean environment. Each municipality differs in the number of consumers and the share of ‘traditional’ and ‘green’ consumers.

Consumption of the ‘produced good’ results in waste. Consumers can choose to generate three types of waste, namely rest waste, low quality organic waste and high quality organic waste. The private households must invest labor in the separation of organic from rest waste. Generating high quality organic waste will cost more labor than generating low quality organic waste.

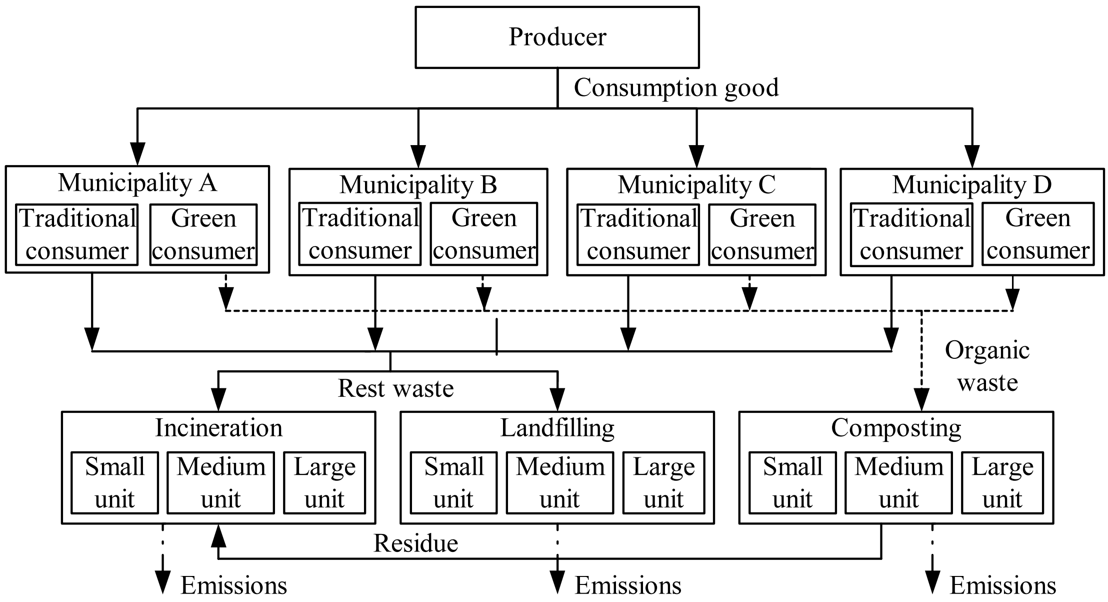

Figure 6-1 Representation of the model structure

Consumers can either pay a flat fee for the collection of waste or a unit-based price. The flat fee-pricing scheme will be implemented in the model using the subsidy-*cum*-tax scheme as described in Chapter 4. To recapitulate the idea of this scheme, consumers pay the equilibrium price for collection of waste to the municipalities. They are reimbursed with the exact amount of money in the form of a subsidy from the government. Thus the perceived price of collection for the consumers equals zero. The consumers pay a direct tax to the government to reimburse the costs of the subsidy. In such a way, the consumers do pay for waste collection, but the amount they pay is not coupled with the quantity of waste they generate, the exact definition of a flat fee.

The municipality, who is a producer of collection services, collects rest waste and sends it to an incinerator or a landfill. Low quality organic waste and high quality organic waste are both collected together and sent to a composting unit. The costs of composting depend on the quality of the organic waste stream. If the overall quality of organic waste is low (*i.e.* there is a large share of low quality organic waste in the entire organic waste stream), it will be difficult to compost the waste. Part of the waste will be rejected by the composting unit and sent to an incinerator. This will greatly increase the costs of composting.

Spatial aspects in the model: transport costs and economies of scale

In the model, the spatial aspects of the waste management problem are integrated in a general equilibrium framework. This means including transporting costs and economies of scale. In the model, municipalities collect waste and send it to a waste treatment unit. The municipality has the choice between sending waste to a small, a medium or a large sized waste treatment unit. The small unit is located close to the municipality; the medium and large waste treatment units are located further away. Thus the larger waste treatment units are more efficient in waste treatment, but they are more expensive with regard to transport costs. The key characteristics of the spatial aspects are depicted in Figure 6-2.

According to the standard specification of general equilibrium models, we assume perfect competition between production sectors. This implies that there is also perfect competition between waste treatment units. In reality this may not be the case. For example, contracts between municipalities and incinerators exist, which specify the quantity of waste municipalities must deliver to the incinerator. It goes beyond the scope of this thesis to include such imperfect competition in the model. However, in future research it would be interesting to see how monopolistic behavior of waste treatment units influence the results.

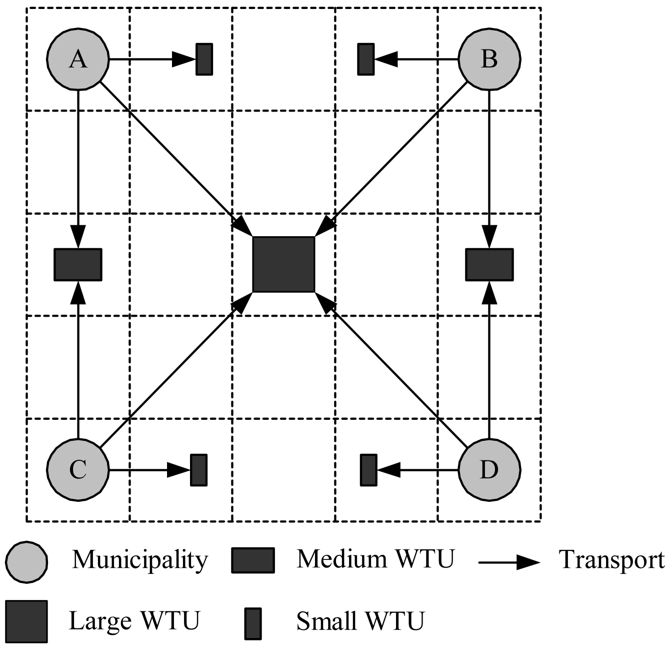

Figure 6-2 Spatial aspects in the model for four municipalities

Note: WTU stands for waste treatment unit

The three waste treatment units, namely the landfill unit, the incineration unit, and the composting unit, offer waste treatment services to the municipalities. These units use capital and labor as inputs to the production process. The composting unit, besides using capital and labor, also uses incineration services to get rid of both the residue and the rejected low quality organic waste.

Due to economies of scale in the production process, the larger waste treatment units offer waste treatment services at a lower price. In the context of our study, several locations are exogenously specified for each waste treatment unit. Given transport costs and waste treatment costs of each unit, municipalities will choose the least expensive option. Except for the differentiated marginal costs, the services provided by the different sizes of a waste treatment unit are identical.

Composting costs are also differentiated according to the quality of organic waste. To compost organic waste of a lower quality, the waste must be cleaned. If the organic waste is too badly polluted it cannot be composted. The composting unit will thus have to send the waste to an incinerator. It is, therefore, more expensive to compost organic waste of a lower quality. The costs of composting low quality organic waste will also be differentiated according to the size of the composting unit. Smaller composting units tend to charge a higher price to compost low quality organic waste than larger units.

6.2.2 The model represented in equations

As stated above, the model presented in this section is an extended version of the model presented in Chapter 5. The main differences between this model and the model in Chapter 5 are the introduction of several municipalities, waste treatment units, economies of scale, transport costs, and emissions generated by waste treatment.

Welfare, production, consumption and waste generation

The model is built like a standard general equilibrium model in the Negishi format. This means that total welfare is maximized subject to utility constraints, balance constraints and production possibility sets. The total welfare function is shown in equation 6.1. Total welfare (*TWF*) is defined as the sum of the utilities (u_i) of each consumer *i*. The utility of each consumer is weighted by a factor α_i, the so-called Negishi weight[2]. Each consumer derives utility from the consumption of the consumption good (x_i^g).

$$TWF = \sum_i \alpha_i \ln[u_i(x_i^g)] \tag{6.1}$$

[2] The value of the Negishi-weights is determined in such a way that the budget constraint for each consumer holds. This means that each consumer will spend exactly its entire income on the consumption of goods and services or savings. See Chapter 4 for more information about the determination of the Negishi-weights.

The production function of the 'produced' good is given by a CES function. The producers of the 'produced' good use only capital *(k),* labor *(l)* as inputs for production[3].

$$q_g \leq CES(k_g, l_g; \sigma_g^{k,l}) \tag{6.2}$$

Consumption of the 'produced good' results in waste generation. Waste generation has been specified as a percentage β of consumption x_i^g. In one of the scenarios presented in 6.3, we include the possibility of prevention. In this scenario, two goods are produced and consumers can alternate between these goods. The only difference between these goods is the waste content β^g. We assume that only the private households generate waste; the government does not generate waste.

$$w_c = \beta^g x_c^g \tag{6.3}$$

Where *c* is a subset of *i* including only the private households.

The private households deal with the waste they generate by demanding the so-called waste collection services provided by the municipality. They can substitute the demand for collection services of rest waste (x^r) for the demand for collection services of organic waste (x^o). Private households may also choose to generate *low* quality organic waste, $x^{o,l}$, or *high* quality organic waste, $x^{o,h}$, as specified in the following CES function:

$$W_c = CES(x_c^r, CES(x_c^{o,l}, x_c^{o,h}; \sigma_c^{l,h}); \sigma_c^{r,o}) \tag{6.4}$$

If the private households decide to generate organic waste, they will have to expend labor *(lw)* on separating organic waste from rest waste. Producing high quality organic waste involves more labor than producing low quality organic waste. The 'production possibility set' for organic waste of quality *f* is defined as: (*f*= *low, high*):

$$x_c^{o,f} \leq \mu^f \; lw_c^f \tag{6.5}$$

Where μ reflects the units of labor necessary to produce a unit of organic waste of quality *f*.

[3] The notation $z=CES(x,y;\sigma)$ reflects the following function: $z = \left(x^{\frac{(\sigma-1)}{\sigma}} + y^{\frac{(\sigma-1)}{\sigma}} \right)^{\frac{\sigma}{\sigma-1}}$

The amounts of labor expended on generating organic waste affects the labor supply. The labor supply L of each consumer i is calculated as the exogenous labor endowment, $\overline{L}$, minus the total amount of labor used for generating both types of organic waste.

$$L_c = \overline{L_c} - \sum_f lw_c^f \tag{6.6}$$

Transport costs

The municipality collects the waste generated by private households and sends it to a waste treatment unit. The municipality must pay transport costs for transporting waste to a waste treatment unit. Transport costs are modeled as if the municipality demands transport services of a transport company. Waste treatment units will charge a price to treat the waste. Each different waste treatment method, (index *m= incineration, landfilling, composting)* will charge a different price, according to the marginal waste treatment costs.

The production function for waste collection services is given by a nested Leontief-CES function. It depends on the inputs of capital, labor, transport services *(ts),* and waste treatment services *(wts)*. Municipalities can choose between different waste treatment methods, *i.e.* incineration, landfilling, composting, and different sizes of waste treatment units. Each waste treatment unit comes in three different sizes: small, medium, and large (index *s = small, medium, large*). The production of waste collection services of each municipality *j* is given as:

$$q_{cs} \leq \min(CES(k_{cs}, l_{cs}; \sigma_{cs}^{k,l}), CES(wts^{m,s}; \sigma_{cs}^{m,s}), ts_{cs}) \tag{6.7}$$

Municipalities cannot make a completely free choice between the three different waste treatment options. Organic waste will have to be transported to a composting unit. Rest waste can either be brought to an incinerator or a landfill.

The total transport costs depend on the quantity of waste transported and the distance traveled. The possible locations for each waste treatment unit are exogenously determined. The total distance from the municipality to a waste treatment unit is given in a transport matrix *T*. Thus the demand for transport services (*ts*) by municipality *j* is calculated as the sum of the transport costs to each waste treatment unit *m*:

$$ts \leq \sum_m T^m wts^m \tag{6.8}$$

The transport services are produced with the input of capital and labor. The production function for transport services is defined as follows:

$$q_{ts} \leq CES(k_{ts}, l_{ts}; \sigma_{ts}^{k,l}) \quad (6.9)$$

Emission rights

The production of waste treatment services generates emissions, a negative externality. Here the environment is treated as a resource. Firms must buy emission rights from the government to produce products, just as they need to purchase capital and labor. By introducing emission bounds, the firms will be restricted in the amount of pollution they generate. According to Ginsburgh and Keyzer (1997), it is preferable to model the environment as a resource instead of as an externality since a resource can easily be incorporated in the model. There exists a positive price for the good 'clean environment' and no extra equations are necessary to ensure that a general equilibrium solution may be found.

Economies of scale

As mentioned above, the marginal costs of waste treatment services produced by a larger unit are lower than the marginal costs of waste treatment services produced by a smaller unit, due to economies of scale. This is modeled as if the larger unit uses a more advanced technology *A,* which means that fewer input factors are necessary to produce the same quantity of services[4]. Economies of scale are introduced exogenously in the model. This means that the used technology does not depend on the quantity of waste treated in the waste treatment unit.

The production function for the different waste treatment units of type *s* and size *m* is defined as follows:

$$q_{m,s} \leq A_{m,s} CES(k_{m,s}, l_{m,s}, e_{m,s}; \sigma_{m,s}^{k,l,e}) \quad (6.10)$$

Finally, as in any general equilibrium model, balance equations have been included to determine that the demand does not exceed the supply of each commodity in the model[5].

To recapitulate, the model maximizes utility of the consumers and simultaneously determines the optimal method and location of waste treatment, the composition of

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[4] In the small waste treatment unit, *A* is equal to unity. In the medium and large waste treatment unit *A* is larger than unity, thus less production factors need to be used in the production process.

[5] See Chapter 5 for a specification of the balance equations.

municipal solid waste generated, and finally the social costs of waste treatment, which consist of financial and environmental costs.

6.3 Model application and numerical analysis

The model discussed above is applied in a stylized example with numerical data from the Randstad, a large region in the Netherlands. This example focuses on the waste market. The economic data used in the numerical example is based on national accounts for the Netherlands in 2000 (Statistics Netherlands, 2002b). This data is aggregated to one production sector and two production factors (capital and labor) and supplemented with detailed data of the waste sectors (collection, incineration, landfilling, composting) and data necessary for the subsidy-*cum*-tax scheme (fee and subsidy). The data on waste generation and waste treatment are based on WMC (2000a, 2001, 2003d).

6.3.1 The benchmark case

The social accounting matrix for total economy in the benchmark case

The social accounting matrix of the economy is presented in Table 6-1. Supply or producers' output and consumer endowments are given as positive values; demand or producer inputs and consumption are given as negative values[6].

To keep the model as simple as possible, government income is dependent on a lump-sum transfer instead of an income from taxes on labor and consumer goods[7]. This has been added to the social accounting matrix.

[6] The entries in the column times the corresponding prices sums up to zero to ensure that the zero profit condition holds: value of inputs equals value of outputs. The entries in the column of each consumer times the corresponding price adds up to zero to ensure that the budget constraint holds: each consumer spends exactly its income on the consumption of goods and services. The entries in each row times the corresponding prices adds up to zero to ensure that each market clears: total demand for each commodity must equal total supply. In Table 6-1 the rows and columns may not add up to zero exactly due to the rounding off of several numbers.

[7]As we are interested in the first best equilibrium solution, the existence of distortionary taxes has been ignored.

Table 6-1 Benchmark accounting matrix (expenditures in million Euro and quantities of waste in Ktonnes)

	Good	CS rest	CS organic	Composting	Incineration	Landfill	Transport	Traditional consumer	Green consumer	Government	Price
Good	155229	0	0	0	0	0	0	-73346	-35315	-46569	1
CS rest	0	1733	0	0	0	0	0	-1231	-502	0	0.263
CS organic	0	0	413	0	0	0	0	-217	-196	0	0.206
Composting	0	0	-413	413	0	0	0	0	0	0	0.045
Incineration	0	-58	0	-29	-87	0	0	0	0	0	0.102
Landfill	0	-1675	0	0	0	1675	0	0	0	0	0.025
Transport	0	-61	-14	0	0	0	75	0	0	0	0.36
Capital	-67264	-103	-25	-14	-8	-36	-15	45539	21926	0	1
Labor	-87965	-155	-37	-1	-1	-6	-12	59520	28658	0	1
Tax	0	-128	0	0	0	0	0	0	0	128	1
CO_2	0	0	0	0	-40478	0	-2473	0	0	0	0
NO_X	0	0	0	-7	-36	-25	-27	0	0	0	0
CH_4	0	0	0	-991	-3	-17489	0	0	0	0	0
Fee	0	0	0	0	0	0	0	-344	-166	510	1
Subsidy	0	0	0	0	0	0	0	368	172	-540	1
Transfer	0	0	0	0	0	0	0	-31369	-15103	46472	1

Note: 'Good' stands for the consumption good; 'CS rest' stands for collection services of rest waste; 'CS organic' stands for collection services of organic waste; 'Tax' stands for the tax paid for landfilling; 'CO_2', 'NO_x' and 'CH_4' stands for emissions of these gasses; 'Fee' stands for the flat fee consumers pay to the government for collection of waste; 'Subsidy' stands for the total amount of money the government gives for collection of waste as a subsidy to the consumers; 'Transfer' stands for a lump-sum transfer from the consumers to the government. The price column gives the prices of all commodities.

In total, consumers generate 1733 thousands tonnes of rest waste and 413 thousands tonnes of organic waste. We assume that all organic waste collected is composted, since this is consistent with the laws concerning organic waste in the Netherlands (WMC, 2002). Rest waste can either be incinerated or landfilled. In the benchmark case, we assume that a large quantity of waste is landfilled. Landfilling is less expensive than incineration. We have included, however, a large tax on landfilling which raises the price including the tax of landfilling to a level equal to the price of incineration. Given that the price of landfilling is equal to the price of incineration, municipalities have no preference based on price differences for either landfilling or incineration. Since 2000, such a landfilling tax has actually been introduced in the Netherlands (see Chapter 3 for more information).

Consumers pay a fixed amount of money for the collection of rest waste, the so-called flat fee. In this model, private households demand waste collection services and pay an equilibrium price for these services. To introduce a zero marginal price for waste generation, however, the government reimburses these costs to the consumers in form of a subsidy, which is exactly equal to the equilibrium price for every unit of waste collection services. Thus, the marginal price of waste generation for the households equals zero. Consumers pay a total amount of 510 million Euros for the collection of waste. On average, the fee paid by the consumers covers only 95% of the real costs (WMC, 2001). This means that the real costs of waste collection and thus the amount spent on the subsidy on waste collection equals roughly 540 million Euros.

The costs of collecting 1733 thousands tonnes of rest waste are approximately equal to 456 million Euros (WMC, 2000a). This means that the price of collecting a tonne of rest waste is equal to 263 Euros. This price is shown in Table 6-1 in the price column. The prices of organic waste collection, composting, incineration, landfilling, and transport have been calculated in a similar manner. Prices of the ‘produced good’, capital, labor, fee, subsidy, and lump-sum transfer have been normalized to one according to the Harberger convention[8].

The government derives its income from both the lump-sum transfer[9] paid by the consumers and the landfill tax. The government spends its income on the consumption of the ‘produced’ good and the subsidy costs. The value of government consumption is kept constant at its benchmark level. This means that the lump-sum

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[8] As in Chapter 4, the Harberger convention has been adopted in the benchmark data for all unknown prices.

[9] A lump-sum transfer will only affect the income level of the consumer and thus the total expenditure of that consumer. It will not result in a change of the consumption pattern of that consumer.

transfer must be variable. If, for example, the income of the government declines due to an increase of the subsidy costs, consumers will reimburse the government through an increase in the lump-sum transfer.

Composting, incineration, landfilling, and transport cause emissions of CO_2, NO_x, and CH_4 gasses. WMC (2003e) published information about the total emissions in kg per tonne of waste treated for composting, incineration, and landfilling. These data have been added to the input output matrix in Table 6-1. For a more detailed overview of the emissions caused by waste treatment, see Chapter 3. In the benchmark case, no emission control measures have been taken. Therefore, industries do not have to incur any costs of reducing emissions and thus the prices of emission rights equal zero.

Transport costs and economies of scale

The transport costs of transporting a tonne of waste to a waste treatment unit depends on the distance traveled. It is here assumed that a larger unit will be located further from the municipality than a smaller unit. In Table 6-2 the transport distances are presented. These distances are based on the average distances from a municipality to a small, medium, or large facility in the Netherlands (WMC, 2003e).

Table 6-2 Transport distances

Size of waste treatment unit	Distance from municipality (in km's)
Small	35
Medium	75
Large	150

In the benchmark data, we do not include economies of scale: the cost of treating waste in a small waste treatment unit will be identical to treating it in a larger unit. This assumption will be relaxed in scenario 3.

Municipalities

We assume that the region consists of four types of municipalities: (A) a large municipality with a relatively high percentage of traditional consumers, (B) a large municipality with a relatively high percentage of green consumers, (C) a small municipality with a relatively high percentage of traditional consumers and (D) a small municipality with a relatively high percentage of green consumers. Each of these four types of municipalities collects waste from private households living in that municipality.

It is presumed that a large municipality generates 40% of the total quantity of waste in the economy. The smaller municipality generates 10% of the total quantity of waste. Each different type of municipality has a different share of 'traditional' and 'green' consumers. ‘Traditional’ consumers are those consumers who have little preference for the environment. ‘Green’ consumers, on the other hand, have some preference for protecting the environment, these consumers are more likely to recycle and compost waste than traditional consumers. As calculated in Chapter 5, on average there are 33% green consumers and 67% traditional consumers. Table 6-3 shows the percentages of traditional and green consumers in each municipality.

Table 6-3 Differences between municipalities

Municipality	Percentage waste generated	Share traditional consumers	Share green consumer
Municipality A	40%	72%	28%
Municipality B	40%	68%	32%
Municipality C	10%	65%	35%
Municipality D	10%	50%	50%

Generation of organic waste

In accordance with the definition of Chapter 5, high quality organic waste has been defined as 100% pure organic waste and low quality organic waste as only 70% pure. This means that composting of 100 tonnes of high quality organic waste does not result in a residue and the composting of 100 tonnes of low quality organic waste will result in 30 tonnes of rest waste, which has to be incinerated. As shown in chapter 5, the overall mixture of organic waste consists of 23.3% low quality organic waste and 76.7% high quality organic waste. In the benchmark data, ‘green’ consumers generate 90% high quality organic waste and 10% low quality. ‘Traditional’ consumers generate 70% high quality organic waste and 30% low quality organic waste.

As shown in Chapter 5, the average costs of handling organic waste are equal to 78 dollars per tonne. It is 10% more expensive to generate high quality organic waste and 10% less expensive to generate low quality organic waste. The actual labor costs of generating organic waste per consumer in the benchmark case are shown in Table 6-4.

Consumers are obviously not able to create high quality organic waste from just any type of rest waste, as that would be the equivalent of transforming, for example, a tin can into organic material using only labor. An upper limit on the quantity of high quality organic waste consumers can separate from rest waste has therefore been introduced. About 32% of the rest waste stream consists of organic waste (Beker, 2002); therefore, consumers will be able to increase their generation of high quality organic waste to a maximum of 32% of their original production of rest waste.

Table 6-4 Additional inputs on the generation of organic waste in the benchmark

	Municipality A		Municipality B		Municipality C		Municipality D	
	Traditional consumer	Green consumer	Traditional consumer	Green consumer	Traditional consumer	Green consumer	Traditional consumer	Green consumer
Low quality (Ktonnes)	49.56	3.3	41.3	8.26	11.01	1.65	8.26	3.3
High quality (Ktonnes)	99.12	13.22	82.6	33.04	22.03	6.61	16.52	13.22
Share low quality	0.33	0.20	0.33	0.20	0.33	0.20	0.33	0.20
Labor low quality	5.06	0.34	4.21	0.84	1.12	0.7	0.84	0.34
Labor high quality	11.33	1.51	9.44	3.78	2.52	0.76	1.88	1.51

Substitution elasticities

All production sectors are characterized by a CES-production function. Substitution elasticities for the different production sectors and the substitution possibilities between different types of waste are given in Table 6-5. Most of the substitution elasticities presented in Table 6-5 have been calculated in Chapter 5, only the substitution elasticity between landfilling and incineration is new.

Table 6-5 Substitution elasticities for production factors and waste categories

	CS rest	Traditional consumer	Green consumer
Substitution elasticity incineration and landfilling ($\sigma^{i,l}$)	6	-	-
Substitution elasticity organic waste and rest waste ($\sigma^{o,r}$)	-	0.6	0.3
Substitution elasticity high and low quality compost ($\sigma^{h,l}$)	-	0.9	0.1

Note: 'Good' stands for the produced good; 'CS rest' stands for the collection of rest waste and 'CS organic' stands for the collection of organic waste.

The municipalities can choose between treating the waste in a landfill unit or an incinerator. In the Netherlands, municipalities do not have much choice in how the waste is treated. The landfilling of combustible waste is prohibited. Since we want to show in this chapter, however, how the choice of the preferred waste treatment option is influenced by the transport costs, quality, and quantity of waste generated, we assume that municipalities can choose to landfill their waste. The choice between landfilling an incineration will only depend on the prices of the waste treatment options. Furthermore, to ensure that municipalities can easily substitute incinerating for landfilling their waste, a large substitution elasticity between the two options has been introduced.

6.3.2 Scenarios

In several scenarios we illustrate how the quality of waste affects both the location of waste treatment units and the choice between waste treatment options. This in turn will affect both the financial and environmental costs of waste treatment. In these scenarios, some of the assumptions of the benchmark case will be relaxed and it will be demonstrated how respectively a unit-based price for the collection of waste, emission restrictions, economies of scale, differentiating composting costs and the possibility of prevention can influence the waste market and the social costs of waste treatment.

Scenario 1a Benchmark: Flat fee

Scenario 1a is an exact replication of the benchmark data as described in Section 6.3.1. In scenario 1a, consumers pay a flat fee for the collection of organic waste and rest waste. Thus they have no price incentive to lower the quantity of waste they generate. Municipalities collect the waste and choose the location of the facility to which they transport it. In this scenario, economies of scale do not influence the costs of waste treatment. This means that all different sizes of a waste treatment unit will handle waste for the same price. Thus municipalities will transport their waste to the nearest facility to minimize transport costs. Landfilling is less expensive than incineration in terms of operation costs. The government, however, has taxed landfilling so that the price of landfilling is now equal to the price of incineration. The benchmark case reflects the historical situation. Over the past few decades, landfilling has been the most popular waste treatment option. Until the mid 1990s, most of the waste was landfilled. Incineration only became popular after 1996 because the government stimulated incineration by both prohibiting and taxing the landfilling of combustible waste.

Scenario 1b Selective unit-based pricing system

In scenario 1b, a unit-based price for the collection of rest waste is introduced. This means that consumers pay the equilibrium price for the collection of rest waste, which reflects the marginal costs of producing these services. The consumers still pay a fixed amount of money for the collection of organic waste, so the marginal costs of producing organic waste remain equal to zero. It may be expected that this policy change will provide consumers with an incentive to reduce generation of rest waste in favor of generation of organic waste. Consumers will start to separate more organic waste and rest waste. To separate more waste, they will need to incur more costs in terms of labor use.

Scenario 1c Full unit-based pricing system

In scenario 1c, a unit-based price is introduced for the collection of both organic waste and rest waste. In this scenario, consumers pay the marginal costs of generating waste. Since composting is slightly less expensive than incineration or landfilling, consumers will pay somewhat less for the collection of organic waste than for the collection of rest waste. Consumers, however, do not only pay a fee for the collection of organic waste, but they also incur costs generating organic waste, as they have to invest labor in waste separation. Thus the total costs of producing low and high quality organic waste combined with the collection costs are higher than the collection costs of rest waste. Therefore, only a minimal change in the generation of organic waste and rest waste may be expected. Introducing a unit-based price on the collection of rest waste and organic waste can give the consumer an incentive to minimize their total waste generation by consuming less[10].

By comparing this scenario with scenario 1b, it can be evaluated (*i*) whether the introduction of a unit-based pricing scheme would lower the quantity of waste generated and thus the social costs of waste treatment and (*ii*) whether the selective or the full unit-based pricing scheme is more effective in terms of the quantity of rest waste prevented and in terms of the lowest social costs of waste treatment.

Scenario 2 Emission restrictions

In Scenario 2, we analyze how emission restrictions can influence the optimal method of waste treatment. To analyze this, we introduce two new elements in the model. Firstly, emission rights are included as inputs for production of the three waste treatment options: composting, incineration, and landfilling as discussed in Section 6.2.2. To treat waste, the waste treatment units will have to buy CO_2, NO_X, and CH_4 emission rights. Each waste treatment facility will generate different emissions. Incineration results in CO_2 and NO_X emissions, landfilling in NO_X and CH_4 emissions and, finally, composting creates CH_4 emissions. In relative terms, landfilling is the least environmentally friendly waste treatment option and composting the most environmentally friendly one[11].

To introduce emission restrictions in the model, two steps have been taken. In the first step the costs waste treatment units make for emissions abatement must be explicitly modeled. Waste treatment units need to control their emissions since Dutch law

[10] Note that there is no prevention in this scenario and that the possibilities for reducing waste are limited. Prevention will be investigated in a later scenario.

[11] See Chapter 3 for an overview of the emissions generated by the three waste treatment options.

specifically regulates how much waste treatment units can emit. In the benchmark data, these emission control cost were included in the capital costs. In this scenario, we wish to explicitly model these emission control cost as if the waste treatment unit buys emission rights. Thus the benchmark model has been adjusted, emission rights have been introduced, and the capital costs no longer include emission abatement costs.

As the second step, emission restrictions have been introduced. The government seeks to decrease the environmental damage of waste treatment units and therefore reduces the available emission rights by 20% for all pollutants. The results for the case of a flat fee on waste collection (scenario 2a), a selective unit-based pricing scheme (scenario 2b) and a full unit-based pricing scheme (scenario 2c) have been computed with the revised benchmark data. Comparing scenario 2a, b and c with scenario 1a, b and c respectively shows how emission restrictions will influence waste generation and waste treatment costs in the different pricing scenarios.

Scenario 3. Economies of scale

In the benchmark case we assumed that waste treatment units would charge the same price independent of the size of the waste treatment unit. In reality, a larger waste treatment facility will be able to offer waste treatment services against a lower price due to economies of scale. In this scenario, economies of scale are introduced for the three sizes of waste treatment units by changing the technology parameter in the production function. In Table 6-6 the new values of the technology parameters are shown[12]. Note that only the technology parameters for the medium and large facilities have changed; the technology parameters for the small facilities are equal to the benchmark values. In the benchmark case, all waste was transported to the small waste treatment facilities. In this scenario, municipalities can choose to transport waste to a larger facility if the extra transport costs are offset by the lower waste treatment costs due to economies of scale.

Table 6-6 Value technology parameter waste treatment

	Small	Medium	Large
Composting	1.0	1.1	1.3
Landfilling	1.0	1.1	1.3
Incineration	1.0	1.3	1.7

[12] If the value of the technology parameter *A* is larger than unity then fewer inputs are necessary to produce the same quantity of outputs.

The values of the technology parameters are calculated based on information about different sizes of waste treatment units and the prices charged by these units. Data has been used from WMC (1993, 2001, 2002).

We will show how economies of scale influence waste generation when a flat fee is charged for the collection of waste (scenario 3a), when a selective unit-based pricing system is introduced (scenario 3b) and when a full unit-based pricing system is introduced (scenario 3c). Note that the data used in this scenario is equal to the benchmark data described in Section 6.3.1. Hence scenario 3a, 3b, and 3c can be directly compared with scenario 1a, 1b, and 1c, respectively.

Scenario 4. Differentiating composting costs and economies of scale

Thus far we have not considered that a lower quality of organic waste will cost more to compost in two ways: firstly the composting process itself is more expensive because more residue is produced, which has to be incinerated and secondly organic waste either needs to be cleaned or is rejected and sent to an incinerator before composting, which also increases the costs. In this scenario, these differences in costs of composting low and high quality organic waste have been introduced combined with the economies of scale as presented in the previous scenario. To introduce the differentiated composting costs, two steps were taken.

First of all, the benchmark data set was adjusted. In the benchmark data set, it was assumed that composting of organic waste of a mixed quality would result in a residue, which would have to be incinerated. In this scenario, this assumption is modified: composting of high quality organic waste will not result in any residue whereas composting of low quality organic waste will. This residue is treated in an incinerator. As a consequence, the price of composting low quality organic waste is higher than the price of composting high quality organic waste. The price of composting a tonne of low quality organic waste in the new data set is equal to 73 Euros. The price of composting a tonne of high quality organic is equal to 38 Euros. The data is configured so that the overall costs of composting waste are identical to the benchmark data. Thus, except for the differentiated composting costs, the data set used in this scenario is equal to the benchmark data set[13].

Secondly, stricter rules on the quality of compost have been introduced. As a consequence, the composting unit will have to ensure that the quality of organic waste is good enough to produce high quality compost. Treating a lower quality of organic

[13] Note that to introduce economies of scale, the data set did not need to be adjusted, only the technolgy variable was adjusted.

waste becomes more expensive, since it will need to be cleaned before composting. Part of the organic waste will not be composted but instead sent to an incineration unit if the quality declines too greatly. This will significantly increase the costs of composting. It will be more expensive to treat low quality organic waste in small composting units. In Table 6-7, the marginal costs of composting a 1000 tonnes of respectively high and low quality organic waste are given.

Table 6-7 The marginal costs of composting low and high quality organic waste (Euro per tonne of waste)

Costs	Small	Medium	Large
Composting low quality	258	203	101
Composting high quality	38	34	29

In this scenario, a situation where a flat fee is charged for the collection of waste (scenario 4a) will once again be compared with a situation where a selective unit-based price is introduced (scenario 4b) and a situation where a full unit-based price is introduced (scenario 4c). Comparing this scenario with scenario 3a, b and c respectively will provide insight into how the increased composting costs as a result of a stricter control on compost quality will influence the results of introducing a unit-based price (either a selective or a full unit-based price).

Scenario 5. Prevention, differentiating composting costs and economies of scale

In the fifth scenario, we introduce the possibility of prevention of waste to test whether prevention will have a significant impact on the success of introducing unit-based pricing. To implement prevention in the model, we differentiate the consumption good into two products. These products differ only in the quantity of potential waste inherent to the product. Consumption of good 1 will result in 33% more waste than consumption of good 2. To keep this scenario comparable to the benchmark case, the quantity of waste generated in the flat fee scenario is equal to the quantity of waste generated in the benchmark case. A fairly large substitution elasticity is introduced between the two goods (σ=3.5). This gives the consumers the option of reducing waste generation by substituting one product for the other. The three scenarios will show how the introduction of waste prevention influences the results under the flat fee pricing scheme (scenario 5a), the selective unit-based pricing scheme (scenario 5b) and the full unit-based pricing scheme (scenario 5c).

The various scenarios and their main characteristics are summarized in Table 6-8.

Table 6-8 The main characteristics of the scenarios

	Emission restrictions	Economies of scale	Differentiating composting costs	Prevention
Scenario 1				
1A) Benchmark Flat fee	-	-	-	-
1B) Selective unit-based price	-	-	-	-
1C) Full unit-based price	-	-	-	-
Scenario 2 Emission restrictions				
2A) Flat fee	+	-	-	-
2B) Selective unit-based price	+	-	-	-
2C) Full unit-based price	+	-	-	-
Scenario 3 Economies of scale				
3A) Flat fee	-	+	-	-
3B) Selective unit-based price	-	+	-	-
3C) Full unit-based price	-	+	-	-
Scenario 4 Differentiating composting costs				
4A) Flat fee	-	+	+	-
4B) Selective unit-based price	-	+	+	-
4C) Full unit-based price	-	+	+	-
Scenario 5 Prevention				
5A) Flat fee	-	+	+	+
5B) Selective unit-based price	-	+	+	+
5C) Full unit-based price	-	+	+	+

\- = Not incorporated

\+ = Incorporated

6.3.3 Results

Results scenario 1. Benchmark

In the first scenario, the effects of introducing a unit-based price for the collection of rest waste (scenario 1b) and a unit based price for the collection of both rest waste and organic waste (scenario 1c) have been analyzed. The results of this scenario are shown in Table 6-9.

If a selective unit-based price is introduced (scenario 1b), the consumers start to generate less rest waste. They do this by generating more organic waste. Table 6-9 shows that the total production of organic waste has increased (compare benchmark with scenario 1b). Consumers generate both more low quality and more high quality organic waste. The increase in production of high quality organic waste implies a success of the policy change. Consumers start to separate more organic waste from rest waste. Besides generating more high quality organic waste, consumers also begin to generate more low quality organic waste. The increase in low quality organic waste is substantially higher than the increase in high quality organic waste. This implies that consumers are disposing of part of their rest waste in the organic waste bin. In particular, the traditional consumers increase their production of low quality organic waste. Since both low and high quality organic waste are collected together the overall quality of organic waste stream decreases.

Table 6-9 Results for the main variables under 'scenario 1 benchmark (in Ktonnes) and the percentage change as compared to scenario 1a

Variable	Scenario1a: Benchmark case: Flat fee	Scenario 1b Selective unit-based price		Scenario 1c Full unit-based price	
Consumption good $^{a)}$	1552	1552	(0%)	1552	(0%)
Collection rest waste	1733	1639	(-5.4%)	1733	(0%)
Collection organic waste	413	507	(22.7%)	413	(0.1%)
Low quality organic waste	85	117	(38.1%)	87	(2.9%)
High quality organic waste	328	390	(18.7%)	326	(-0.7%)
Transport $^{b)}$	75	75	(0%)	75	(0%)
Composting	413	507	(22.7%)	413	(0.1%)
Small unit	413	507	-	413	-
Medium unit	-	-	-	-	-
Large unit	-	-	-	-	-
Incineration	87	90	(4.0%)	87	(0.2%)
Small unit	87	90	-	87	-
Medium unit	-	-	-	-	-
Large unit	-	-	-	-	-
Landfilling	1675	1584	(-5.4%)	1675	(0%)
Small unit	1675	1584	-	1675	-
Medium unit	-	-	-	-	-
Large unit	-	-	-	-	-

Note: a) Expenditure consumption in million Euros
b) Transport in tonnes per 1000 km

All waste is treated in small waste treatment units, since no economies of scale are included in this scenario. Given that less rest waste is generated, less waste has to be incinerated and landfilled. Table 6-9 shows how the quantity of waste landfilled declines as expected. The quantity of waste incinerated, however, increases because the residue of the composting process is sent to an incinerator.

The four municipalities differ in the distribution of traditional and green consumers. It can therefore be expected that the quality of organic waste collected in each of the municipalities will differ. Table 6-10 shows the quality of organic waste collected in each of the four municipalities. In all municipalities, the quality of organic waste decreases. The absolute quality of organic waste is lowest in the two cities (municipality A and B). In scenario 1b, the percentage change in the quality of the organic waste stream is comparable for municipality A, B and C. For these municipalities, the difference in the shares of 'green' consumers is not that large. Thus, although municipality A, the least environmentally friendly municipality, generates the lowest quality of organic waste, the percentage difference between the three municipalities is comparable. Only in municipality D, which has the lowest percentage of 'traditional' consumers and thus does not pollute the organic waste stream as much, is the percentage increase in low quality organic waste lower than in the other municipalities.

Table 6-10 Changes in the share of low quality organic waste in total organic waste stream for each municipality compared to scenario 1a

	Scenario 1a Benchmark case	Scenario 1b Selective unit-based price		Scenario 1c Full unit-based price	
Municipality A	0.216	0.243	(13%)	0.222	(3%)
Municipality B	0.206	0.232	(13%)	0.212	(3%)
Municipality C	0.199	0.225	(13%)	0.205	(3%)
Municipality D	0.170	0.190	(12%)	0.174	(2%)

In scenario 1c, there is hardly any substitution of the generation of organic waste for the generation of rest waste. Due to the increased perceived price of waste collection (from zero to the marginal costs of collection), one can see that the consumers minimize their costs by disposing of some rest waste in the organic waste bin.

Results scenario 2. Emission restrictions

In scenario 2, emission restrictions are introduced for composting, incineration, and landfilling facilities. These facilities have to buy emission rights and, since the available emission rights are restricted, the price of emission rights increases. The goal of the government is to reduce all emissions by 20%. The waste treatment facilities do have the option of substituting capital for emission rights to simulate the possibility of emission reductions. The results for this scenario are shown in Table 6-11. If the results for the selective unit-base pricing system (scenario 2b) are compared with the results of scenario 1b, one sees that consumers generate a little more organic waste and a little less rest waste. Landfilling and incineration is more expensive due to emission restrictions. Generating rest waste becomes, therefore, somewhat more expensive due to the introduction of emission restrictions and the consumers adjust their behavior accordingly.

Table 6-11 demonstrates that if emission restrictions are introduced, municipalities will choose to incinerate more waste and landfill less. Landfilling is more polluting than incineration and therefore incineration becomes more attractive. Although in relative terms composting is the most environmentally friendly waste treatment option, Table 6-11 shows that the quantity of waste composted does not change if the consumers are charged a flat fee for the collection of organic waste (scenario 2a and 2c). To compost more waste, consumers have to generate more organic waste. Since consumers have no direct price incentive to change their waste generation pattern, they do not choose to generate more organic waste even if this is the least expensive in terms of production and environmental costs.

Table 6-11 Results for the main variables under 'emission restriction scenario' (in Ktonnes) and the percentage change as compared to scenario 1a

Variable	Scenario 2a Flat fee		Scenario 2b Selective unit-based price		Scenario 2c Full unit-based price	
Consumption good [a)]	1552	(0.0%)	1552	(0.0%)	1552	(0.0%)
Collection rest waste	1733	(0.0%)	1638	(-5.5%)	1733	(0.0%)
Collection organic waste	413	(0.0%)	508	(23.0%)	413	(0.0%)
Low quality organic waste	85	(0.0%)	117	(38.4%)	87	(3.1%)
High quality organic waste	328	(0.0%)	393	(19.6%)	326	(-0.7%)
Transport [b)]	75	(0.0%)	75	(0.0%)	75	(0.0%)
Composting	413	(0.0%)	508	(23.0%)	413	(0.0%)
Small unit	413	-	508	-	413	-
Medium unit	-	-	-	-	-	-
Large unit	-	-	-	-	-	-
Incineration	91	(4.7%)	91	(5.2%)	91	(4.7%)
Small unit	91	-	91	-	91	-
Medium unit	-	-	-	-	-	-
Large unit	-	-	-	-	-	-
Landfilling	1671	(-0.2%)	1582	(-5.6%)	1671	(-0.2%)
Small unit	1671	-	1582	-	1671	-
Medium unit	-	-	-	-	-	-
Large unit	-	-	-	-	-	-

Note: [a)] Expenditure consumption in million Euros
[b)] Transport in tonnes per 1000 km

Results scenario 3. Economies of scale

In the third scenario, economies of scale have been introduced for the three different sizes of waste treatment facilities. The results of this scenario are presented in Table 6-12. Due to economies of scale, in scenario 3a, 3b, and 3c respectively, municipalities opted to transport their waste to a medium sized incinerator instead of a small one. The economies of scale are smaller for composting units and landfill facilities and therefore municipalities continue to use the small sized units for these waste treatment options to avoid higher transport costs. The transport costs to a large incinerator are too high to be offset by lower incineration costs, so the large incinerators are not used. Because incineration in a medium sized facility is less expensive than landfilling municipalities choose to incinerate more and landfill less. Since they transport waste over a longer distance, the transport costs increase. Part of the waste is still treated in a small waste incinerator. This is the quantity of waste that is left as a residue of the composting process[14]. All rest waste that is generated by the private households is sent to the medium sized waste incinerator.

[14] It is assumed that composting units only incinerate waste in a small incinerator located near the composting unit, thus the waste does not require transportation.

Table 6-12 Results for the main variables under the 'economies of scale scenario' (in Ktonnes) and the percentage change as compared to scenario 1a

Variable	Scenario 3a Flat fee		Scenario 3b Selective unit-based price		Scenario 3c Full unit-based price	
Consumption good [a)]	1552	(0.0%)	1552	(0.0%)	1552	(0.0%)
Collection rest waste	1733	(0.0%)	1639	(-5.4%)	1733	(0.0%)
Collection organic waste	413	(0.0%)	507	(22.7%)	413	(0.1%)
Low quality organic waste	85	(0.0%)	117	(38.0%)	87	(2.9%)
High quality organic waste	328	(0.0%)	390	(18.7%)	326	(-0.7%)
Transport [b)]	79	(5.0%)	79	(4.8%)	79	(5.0%)
Composting	413	(0.0%)	507	(22.7%)	413	(0.1%)
Small unit	413	-	507	-	413	-
Medium unit	-	-	-	-	-	-
Large unit	-	-	-	-	-	-
Incineration	122	(40.3%)	124	(42.6%)	122	(40.9%)
Small unit	29	-	36	-	29	-
Medium unit	93	-	88	-	93	-
Large unit	-	-	-	-	-	-
Landfilling	1642	(-2.0%)	1552	(-7.3%)	1641	(-2.0%)
Small unit	1642	-	1552		1641	-
Medium unit	-	-	-		-	-
Large unit	-	-	-		-	-

Note: [a)] Expenditure consumption in million Euros
[b)] Transport in tonnes per 1000 km

Results scenario 4. Differentiating composting costs

In scenario 4, different composting costs for low and high quality organic waste have been introduced. The lower the quality of organic waste, the greater the costs that have to be incurred in order to actually compost the waste. The results for this scenario are shown in Table 6-13. Introducing differentiated composting costs has little effect on the actual waste streams in the flat fee pricing system (scenario 4a). However, as Table 6-13 illustrates, it does have a strong effect in both the selective (scenario 4b) and the full unit-based pricing system (scenario 4c). In the selective unit-based pricing system, the two larger municipalities (A and B) start to transport their waste to the large sized composting unit. The quality of the organic waste stream has declined so much that it is cheaper to transport the waste to a large facility than to a small facility. In the two smaller municipalities (C and D), the share of 'green' consumers is larger than in the bigger municipalities and thus the quality of the organic waste stream does not decline as much. This means that the cost of composting the waste in a small unit does not increase enough for it to become attractive to compost the waste in a large unit.

In the full unit-based pricing system, consumers actually start to generate less low quality organic waste and more high quality organic waste. These results are inherent to the assumptions of the model. It was assumed that consumers would pay the marginal costs of waste treatment. Since the marginal costs of treating low quality organic waste are higher than the marginal costs of treating high quality organic waste, even if the households need to spend more labor to generate organic waste, the private households have a direct price incentive to reduce the quantity of low quality organic waste. In reality, it is unrealistic to assume that the consumers would pay the exact marginal costs of waste treatment in a full unit-based pricing system. As explained in Section 6.1, municipalities do not check the quality of organic waste during collection. Thus it is impossible to charge consumers a higher price for the collection of low quality organic waste and therefore these results are too optimistic.

Table 6-13 Results for the main variables under the 'differentiating composting costs scenario' (in Ktonnes) and the percentage change as compared to scenario 1a

Variable	Scenario 4a Flat fee		Scenario 4b Selective unit-based price		Scenario 4c Full unit-based price	
Consumption good [a)]	1552	(0.0%)	1552	(0.0%)	1552	(0.0%)
Collection rest waste	1733	(0.0%)	1639	(-5.5%)	1735	(0.1%)
Collection organic waste	413	(0.0%)	507	(22.7%)	410	(-0.6%)
Low quality organic waste	85	(0.0%)	117	(38.1%)	76	(-10.2%)
High quality organic waste	328	(0.0%)	390	(18.7%)	334	(1.9%)
Transport [b)]	79	(5.0%)	125	(66.0%)	79	(5.0%)
Composting	413	(0.0%)	507	(22.7%)	410	(-0.6%)
Small unit	413	-	106	-	410	-
Medium unit	-	-	-	-	-	-
Large unit	-	-	400	-	-	-
Incineration	122	(40.5%)	128	(47.7%)	119	(37.5%)
Small unit	29	-	40	-	26	-
Medium unit	93	-	88	-	93	-
Large unit	-	-	-	-	-	-
Landfilling	1641	(-2.0%)	1552	(-7.4%)	1643	(-2.0%)
Small unit	1641	-	1552		1643	-
Medium unit	-	-	-		-	-
Large unit	-	-	-		-	-

Note: [a)] Expenditure consumption in million Euros
[b)] Transport in tonnes per 1000 km

Results scenario 5. Prevention

In the final scenario, the model has been adjusted to incorporate the possibility of prevention. Consumers can minimize generation of waste by consuming a product, which generates less waste. The results of this scenario are presented in Table 6-14.

As shown in Table 6-14, the introduction of prevention has little effect on the results. Consumers only adjust their consumption pattern in a minor fashion. When either a selective (scenario 5b) or a full unit-based pricing system (scenario 5c) is introduced, consumer start to consume a little bit more of good 2, the consumption of which generates less waste and a little less of good 1.

Table 6-14 Results for main variable under the ‘prevention scenario’ (in Ktonnes) and the percentage change as compared to the adjusted benchmark case

Variable	Scenario 5a Flat fee		Scenario 5b Selective unit-based price		Scenario 5c Full unit-based price	
Consumption good 1 [a]	1008	(0.0%)	1007	(-0.2%)	1007	(-0.3%)
Consumption good 2 [a]	543	(0.0%)	545	(0.4%)	546	(0.4%)
Collection rest waste	1733	(0.0%)	1637	(-5.5%)	1733	(0.1%)
Collection organic waste	413	(0.0%)	506	(22.6%)	410	(-0.6%)
Low quality organic waste	85	(0.0%)	117	(38.0%)	76	(-10.2%)
High quality organic waste	328	(0.0%)	389	(18.6%)	334	(1.8%)
Transport [b]	79	(5.0%)	125	(65.8%)	79	(5.0%)
Composting	413	(0.0%)	506	(22.6%)	410	(-0.6%)
Small unit	413	-	106	-	410	-
Medium unit	-	-	-	-	-	-
Large unit	-	-	400	-	-	-
Incineration	122	(40.6%)	128	(46.9%)	119	(37.1%)
Small unit	29	-	41	-	26	-
Medium unit	93	-	88	-	93	-
Large unit	-	-	-	-	-	-
Landfilling	1641	(-2.0%)	1552	(-7.4%)	1642	(-2.0%)
Small unit	1641	-	1552		1642	-
Medium unit	-	-	-		-	-
Large unit	-	-	-		-	-

Note: [a] Expenditure consumption in million Euros
[b] Transport in tonnes per 1000 km

The price incentive by increasing the marginal price of waste generation, however, is too small to have a large impact on the consumption patterns of private households. These results are not completely unexpected, practical experience has shown that although unit-based pricing stimulates more waste separation, it does not or hardly stimulates prevention (Linderhof *et al.*, 2001). The price incentive is simply too small to change consumption patterns.

6.3.4 Comparing the different pricing mechanisms

Based on the fifth scenario, the most complex scenario, it is possible to make an extensive comparison between the three different pricing systems. As shown in Section 6.3.3, the total quantity of waste generated is hardly affected by the introduction of either a selective unit based pricing system or a full unit-based pricing system. The composition of the waste stream, however, did differ between the three different pricing systems. This is illustrated in Figure 6-3.

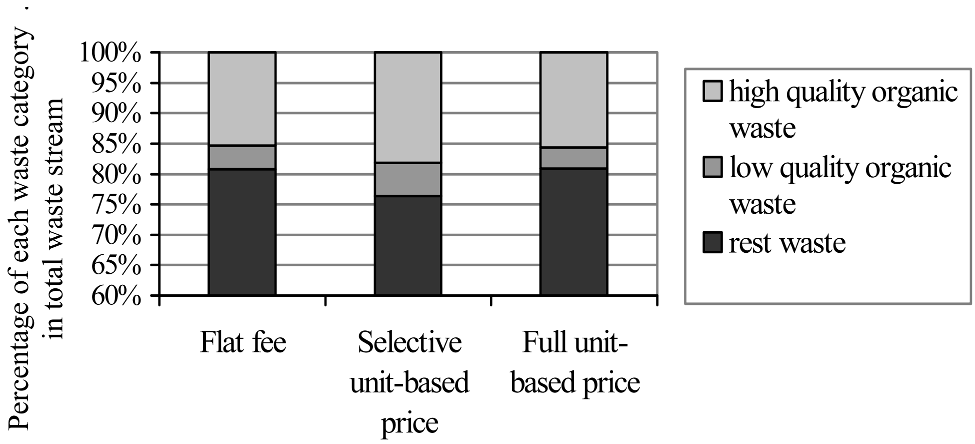

Figure 6-3 The composition of the waste stream for the three different pricing mechanisms

A selective unit-based pricing system causes the most significant substitution of organic waste for rest waste. A full unit-based pricing system has hardly any effect on the quantity of rest waste and organic waste generated. In the selective unit-based pricing system, however, there is also a substantial increase in the quantity of low quality organic waste. Since consumers do not need to fear any penalties, they start to pollute the organic waste stream with rest waste. This waste leakage effect greatly increases the cost of composting organic waste. In a full unit-based pricing system, there is barely any incentive for the consumers to start polluting waste; waste leakage is, therefore, not such a big problem in the full unit-based pricing system.

The cost of waste treatment differed significantly between the different pricing systems. This is shown in Figure 6-4.

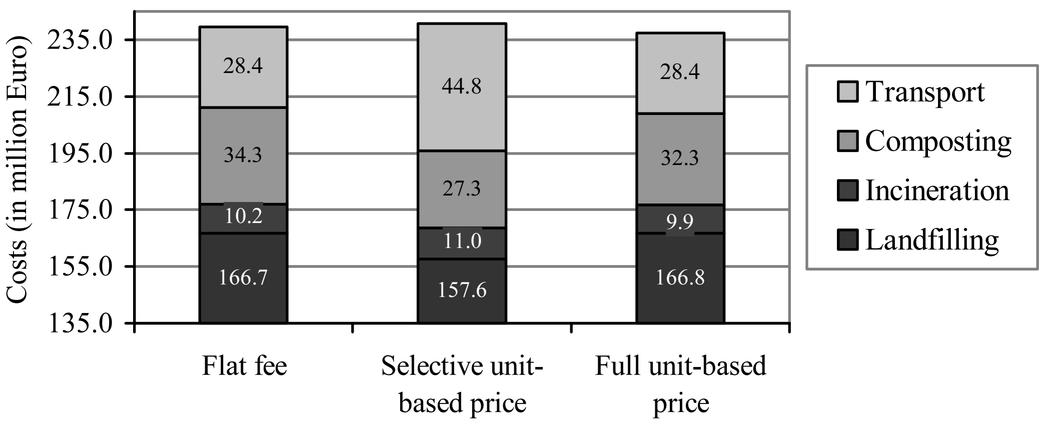

Figure 6-4 The total costs of waste treatment for the three different pricing mechanisms

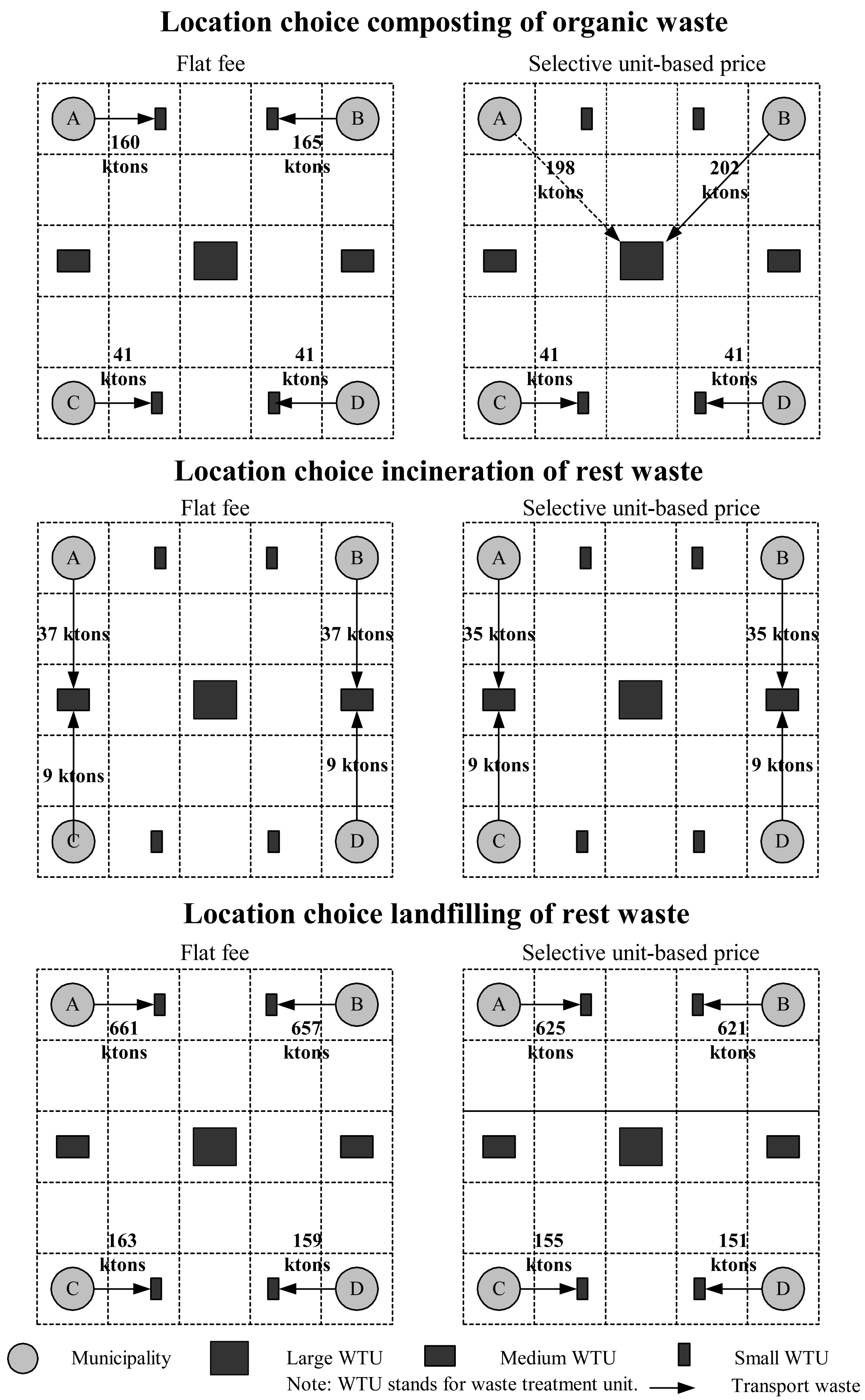

Figure 6-5 Transport of organic and rest waste for each municipality

In a flat fee pricing system, all organic waste is treated in a small composting unit. Households generate mostly high quality organic waste, but also some low quality organic waste. Composting of low quality organic waste is expensive, especially if it is treated in a small composting unit. Although it is quite expensive for the municipalities to compost low quality organic waste in a small composting unit, the increased costs of composting are not so high that they would offset the costs of transporting the waste to a large composting unit. If a selective unit-based pricing system is introduced the situation is quite different. In the larger municipalities, the quality of the organic waste stream declines so much that these municipalities choose to transport the waste to a large composting unit. This results in sharply increasing transport costs and decreasing composting costs, as shown in Figure 6-4. An illustration of the transport flows of waste in a flat fee pricing system and a selective unit based pricing system is given in Figure 6-5 as shown on the previous page.

The distribution of costs in the full unit-based pricing system was quite similar to the distribution of costs in the flat fee pricing system. The overall costs spent on waste treatment were highest in the selective unit-based pricing system, due to waste leakage effect.

Finally, it is also interesting to compare the emissions of CO_2, NO_X, and CH_4 in the different pricing systems. These emissions are shown in Figure 6-6.

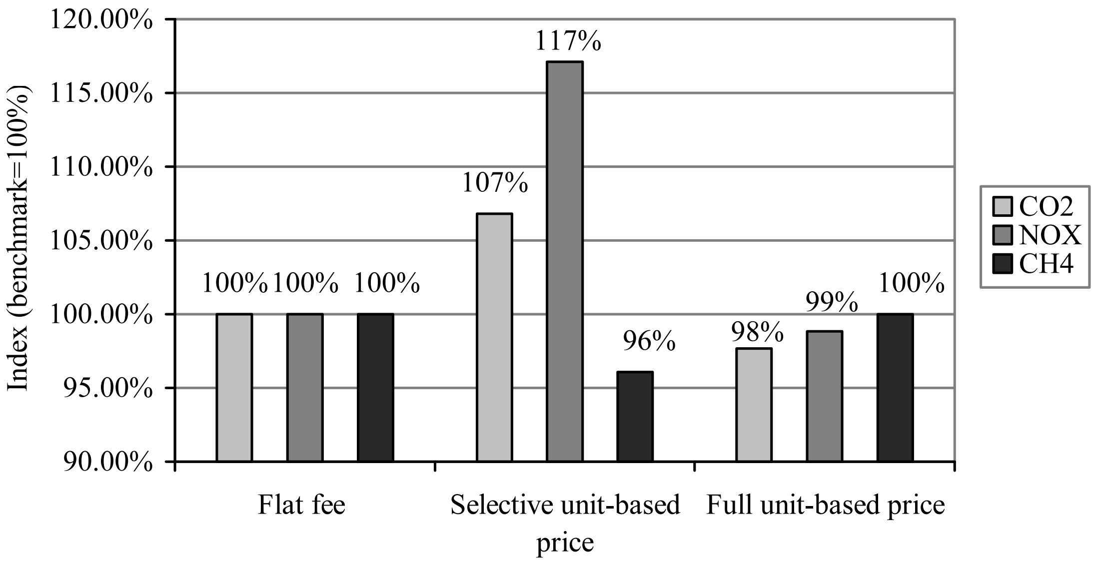

Figure 6-6 Normalized CO_2, NO_X, and CH_4 emissions caused by waste treatment and transport as compared to flat fee scenario (flat fee =100%)

In the selective unit-based pricing system, both CO_2-emissions and NO_X-emissions increase. This is caused by two factors. Firstly, the transport of waste increases sharply in the selective unit-based pricing system. As transport lead to both CO_2 and

NO_X emissions, these emissions increase. Secondly, in the selective unit-based pricing system, more low quality organic waste is generated. Composting low quality organic waste creates a residue, which is incinerated. Incineration of this residue results in extra CO_2-emissions and NO_X-emissions. A positive effect of the selective unit-based pricing system is that it reduces the quantity of waste that is landfilled, thus generating substantially fewer CH_4 emissions.

The emissions in the full unit-based pricing system are slightly lower than the emissions in the flat fee pricing system. The full unit-based pricing system is slightly better for the environment than the flat fee-pricing scheme. Based on the results presented in Figure 6-6, one may jump to the conclusion that a selective unit-based pricing scheme is worse for the environment than either of the two other pricing schemes; however this would depend on an economic valuation of the damage caused by CO_2, NO_X, and CH_4 emissions. It is, however, clear that in our model the selective unit-based pricing scheme aggravates the problem of acidification.

6.4 Discussion and conclusions

In this chapter, we have demonstrated how the spatial aspects of the waste management problem can be included in a general equilibrium framework. A general equilibrium model is a suitable tool for modeling the interactions between the choices that the consumers make about the quality and quantity of waste they generate, the choices that the municipalities make about how and where the waste should be treated and the costs of waste treatment.

In a stylized example with numerical data based on the Randstad in 2000, we have shown how the optimal waste treatment choice and the choice for the optimal location of the waste treatment unit can be influenced by a policy change for the consumers. The results show that if a unit-based price for the collection of rest waste was implemented, the consumers would start to generate less rest waste and more organic waste. This is a positive effect. More waste will be composted, which will result in lower waste treatment costs and less environmental damage. There is, however, also a negative effect. The quality of organic waste would be seriously affected. The 'traditional' consumers in particular would start to dispose of rest waste in the organic waste bin, thus polluting the organic waste stream. Since the quality of the organic waste stream greatly declines, it consequently becomes far more expensive to compost the waste. If the quality of organic waste is too poor, part of the waste cannot be composted. Instead it must be incinerated. This means that the environmental gains will be much smaller than can be expected based on the reduction in waste generation without considering waste leakage. Although less rest waste is incinerated or landfilled, more organic waste has to be composted, thus leading to more CO_2 emissions.

The results of the numerical example show that in the larger municipalities the quality of the organic waste stream declines so much that waste will have to be treated in a large composting facility instead of a small one, thus incurring more transport costs. In this example, it would not be cost-effective to introduce a unit-based price for the collection of rest waste in larger municipalities since the increased costs of composting due to the decrease in the quality of organic waste are not offset by the benefits of the lower generation of rest waste. Only in small municipalities, with a relatively large number of green consumers, will unit-based pricing for the collection of rest waste be interesting.

The introduction of unit-based pricing on the collection of rest waste as well as on the collection of organic waste creates fewer problems with regard to waste leakage. This policy option, however, does not stimulate the consumer to separate its waste. Therefore, there are hardly any benefits of introducing such a pricing scheme, given the restrictions of the model. If we consider that introducing such a pricing scheme will probably involve high transaction costs, it would not be advisable for municipalities to consider this policy option. It is important to bear in mind that only the option of substituting rest waste for organic waste has been analyzed. Particularly in smaller municipalities, consumers will have the option of reducing their organic waste stream by composting it at home. The full unit-based pricing scheme may well provide an incentive for more waste to be composted at home, thus reducing the organic and rest waste streams. Municipalities with a relatively large share of consumers that are not able to engage in home composting, however, do not need to consider this kind of pricing scheme.

Naturally, there are a number of uncertainties in this analysis. Several extensions, such as, for example, home composting, recycling, detailed modeling of prevention or more categories of waste, could be added to the model to make it more realistic. This analysis, however, demonstrates that the interactions between the quality of waste, the method of waste treatment, transport costs, the presence of economies of scale and environmental damages can have significant implications for the success of a policy change and therefore should be considered when deciding on which policy should be implemented in order to minimize waste generation.

Appendix 6-A.1 Results main variables for each scenario (in 1000 tonnes)

Variable	Scenario 1. Benchmark			Scenario 2. Emission restrictions			Scenario 3. Economies of scale			Scenario 4. Differentiating composting			Scenario 5. Prevention		
	Flat fee	Selective unit-based price	Full unit-based price	Flat fee	Selective unit-based price	Full unit-based price	Flat fee	Selective unit-based price	Full unit-based price	Flat fee	Selective unit-based price	Full unit-based price	Flat fee	Selective unit-based price	Full unit-based price
Consumption good 1[a)]	1552.3	1552.1	1552.3	1552.3	1552.1	1552.3	1552.3	1552.1	1552.3	1552.1	1551.9	1552.1	1008.9	1006.7	1006.5
Consumption good 2[a)]	-	-	-	-	-	-	-	-	-	-	-	-	543.3	545.2	545.7
Collection rest waste	1733.0	1638.9	1732.8	1733.2	1637.7	1732.7	1733.0	1639.0	1732.7	1732.5	1638.6	1735.1	1732.5	1637.3	1733.2
Collection organic	413.0	506.9	413.2	412.8	508.0	413.3	413.0	506.7	413.2	413.0	506.6	410.4	413.0	506.2	410.4
Low quality	84.8	117.1	87.3	84.7	117.4	87.4	84.8	117.0	87.3	84.8	117.0	76.1	84.8	117.0	76.1
High quality	328.3	389.8	326.0	328.1	392.6	325.9	328.2	389.7	326.0	328.2	389.6	334.3	328.2	389.2	334.3
Transport	75.1	75.1	75.1	75.1	75.1	75.1	78.9	78.7	78.9	78.9	124.7	78.9	78.9	124.6	78.8
Composting	413.0	506.9	413.2	412.8	508.0	413.3	413.0	506.7	413.2	413.0	506.6	410.4	413.0	506.2	410.4
Small unit	413.0	506.9	413.2	412.8	508.0	413.3	413.0	506.7	413.2	413.0	106.3	410.4	413.0	106.2	410.4
Medium unit	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Large unit	-	-	-	-	-	-	-	-	-	-	400.3	-	-	400.0	-
Incineration	86.7	90.2	86.9	90.7	91.2	90.5	121.6	123.6	122.2	121.8	128.0	119.2	121.9	128.0	118.9
Small unit	86.7	90.2	86.9	90.7	91.2	90.5	28.9	35.5	28.9	28.9	39.9	25.8	28.9	40.5	26.3
Medium unit	-	-	-	-	-	-	92.7	88.1	93.2	92.9	88.1	93.4	93.0	87.5	92.6
Large unit	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Landfilling	1675.2	1584.2	1674.8	1671.4	1582.1	1671.2	1641.8	1552.4	1641.0	1641.2	1552.0	1643.4	1641.1	1552.0	1641.8
Small unit	1675.2	1584.2	1674.8	1671.4	1582.1	1671.2	1641.8	1552.4	1641.0	1641.2	1552.0	1643.4	1641.1	1552.0	1641.8
Medium unit	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Large unit	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-

Note: [a)] Expenditure consumption in million euros

Appendix 6-A.2 Total costs and emissions waste treatment (costs in million euros and emissions in tons.)

Variable	Scenario 1. Benchmark Flat fee	Scenario 1. Benchmark Selective unit-based price	Scenario 1. Benchmark Full unit-based price	Scenario 2. Emission restrictions Flat fee	Scenario 2. Emission restrictions Selective unit-based price	Scenario 2. Emission restrictions Full unit-based price	Scenario 3. Economies of scale Flat fee	Scenario 3. Economies of scale Selective unit-based price	Scenario 3. Economies of scale Full unit-based price	Scenario 4. Differentiating composting Flat fee	Scenario 4. Differentiating composting Selective unit-based price	Scenario 4. Differentiating composting Full unit-based price	Scenario 5. Prevention Flat fee	Scenario 5. Prevention Selective unit-based price	Scenario 5. Prevention Full unit-based price
Total costs waste	224.6	219.9	224.6	241.9	240.1	241.8	223.9	219.3	223.9	239.6	240.8	237.5	239.6	240.7	237.3
Costs composting	18.6	22.8	18.6	20.8	26.3	20.8	18.6	22.8	18.6	34.3	27.3	32.3	34.3	27.3	32.3
Costs incineration	8.8	9.2	8.8	10.3	11	10.3	10.2	10.5	10.2	10.2	10.9	9.9	10.2	11	9.9
Costs landfilling	170.2	160.9	170.1	183.7	175.8	183.7	166.8	157.7	166.7	166.7	157.6	166.9	166.7	157.6	166.8
Costs transport	27	27	27	27	27	27	28.4	28.3	28.4	28.4	44.9	28.4	28.4	44.8	28.4
Total emissions															
CO2	42951	44586	43036.6	34905.7	34905.7	34855.7	59367.9	60300	59642.8	59460.7	63881.5	58269.2	59516.2	63569.5	58124.4
NOX	94.1	95.7	94.2	80.6	80.6	80.7	109.3	110.2	109.5	109.3	128.4	108.3	109.4	128.1	108.1
CH4	18483	17758.3	18479.6	14786.6	14786.6	14786.6	18135.3	17426.4	18127.8	18129	17422.3	18146	18127.8	17414	18128.6

Appendix 6-A.3 Results main variables as percentages compared to the benchmark case

Variable	Scenario 1. Benchmark Flat fee	Scenario 1. Benchmark Selective unit-based price	Scenario 1. Benchmark Full unit-based price	Scenario 2. Emission restrictions Flat fee	Scenario 2. Emission restrictions Selective unit-based price	Scenario 2. Emission restrictions Full unit-based price	Scenario 3. Economies of scale Flat fee	Scenario 3. Economies of scale Selective unit-based price	Scenario 3. Economies of scale Full unit-based price	Scenario 4. Differentiating composting Flat fee	Scenario 4. Differentiating composting Selective unit-based price	Scenario 4. Differentiating composting Full unit-based price	Scenario 5. Prevention Flat fee	Scenario 5. Prevention Selective unit-based price	Scenario 5. Prevention Full unit-based price
Consumption good 1	1552.0	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%[a)]	-0.2%[a)]	-0.3%[a)]
Consumption good 2	-	-	-	-	-	-	-	-	-	-	-	-	0.0%[a)]	0.4%[a)]	0.4%[a)]
Collection rest waste	1733.0	-5.4%	0.0%	0.0%	-5.5%	0.0%	0.0%	-5.4%	0.0%	0.0%	-5.5%	0.1%	0.0%	-5.5%	0.0%
Collection organic	413.0	22.7%	0.1%	0.0%	23.0%	0.0%	0.0%	22.7%	0.1%	0.0%	22.7%	-0.6%	0.0%	22.6%	-0.6%
Low quality	84.8	38.1%	2.9%	0.0%	38.4%	3.1%	0.0%	38.0%	2.9%	0.0%	38.1%	-10.2%	0.0%	38.0%	-10.2%
High quality	328.3	18.7%	-0.7%	0.0%	19.6%	-0.7%	0.0%	18.7%	-0.7%	0.0%	18.7%	1.9%	0.0%	18.6%	1.8%
Transport	75.1	0.0%	0.0%	0.0%	0.0%	0.0%	5.0%	4.8%	5.0%	5.0%	66.0%	5.0%	5.0%	65.8%	4.9%
Composting	413.0	22.7%	0.1%	0.0%	23.0%	0.0%	0.0%	22.7%	0.1%	0.0%	22.7%	0.0%	0.0%	22.6%	-0.6%
Incineration	86.7	4.0%	0.2%	4.7%	5.2%	4.7%	40.3%	42.6%	40.9%	40.5%	47.7%	37.5%	40.6%	46.9%	37.1%
Landfilling	1675.2	-5.4%	0.0%	-0.2%	-5.6%	-0.2%	-2.0%	-7.3%	-2.0%	-2.0%	-7.4%	-2.0%	-2.0%	-7.4%	-2.0%

[a)] percentage change as compared to the adjusted benchmark case. See section 6.3.2 for more information.

Appendix 6-A.4 Cost and emissions waste treatment as percentages compared to the benchmark case

	Scenario 1. Benchmark			Scenario 2. Emission restrictions			Scenario 3. Economies of scale			Scenario 4. Differentiating composting			Scenario 5. Prevention		
Variable	Flat fee	Selective unit-based price	Full unit-based price	Flat fee	Selective unit-based price	Full unit-based price	Flat fee	Selective unit-based price	Full unit-based price	Flat fee	Selective unit-based price	Full unit-based price	Flat fee	Selective unit-based price	Full unit-based price
Total costs waste	224.6	-2.1%	0.0%	3.9%	3.1%	3.9%	-0.3%	-2.3%	-0.3%	6.7%	7.2%	5.8%	6.7%	7.2%	5.7%
Costs composting	18.6	22.7%	0.0%	10.6%	39.8%	10.6%	0.0%	22.7%	0.0%	84.4%	47.0%	73.6%	84.4%	46.8%	73.6%
Costs incineration	8.8	4.0%	0.2%	8.4%	15.7%	8.4%	15.6%	19.1%	16.0%	15.8%	24.2%	12.6%	15.8%	24.3%	12.5%
Costs landfilling	170.2	-5.4%	0.0%	3.4%	-1.0%	3.4%	-2.0%	-7.3%	-2.0%	-2.0%	-7.4%	-1.9%	-2.0%	-7.4%	-2.0%
Costs transport	27	0.0%	0.0%	0.0%	0.0%	0.0%	5.1%	4.8%	5.1%	5.0%	66.0%	5.0%	5.0%	65.9%	4.9%
Total emissions															
CO2	42951	3.8%	0.2%	-18.7%	-18.7%	-18.8%	38.2%	40.4%	38.9%	38.4%	48.7%	35.7%	38.6%	48.0%	35.3%
NOX	94.1	1.7%	0.1%	-14.3%	-14.3%	-14.3%	16.1%	17.1%	16.4%	16.2%	36.4%	15.1%	16.2%	36.1%	14.9%
CH4	18483	-3.9%	0.0%	-20.0%	-20.0%	-20.0%	-1.9%	-5.7%	-1.9%	-1.9%	-5.7%	-1.8%	-1.9%	-5.8%	-1.9%

Appendix 6-B Definition of indices, parameters, and variables

Indices

Label	Range	Description
c	1…2	traditional and green consumer
g	2	consumer goods
f	1...2	quality organic waste (low, high)
i	1...5	consumers
j	1...4	municipality
m	1..3	size waste treatment unit (small, medium, large)
s	1..3	type waste treatment units (composting, incineration, landfilling)

Parameters in GAMS specification

Symbol	Description
α	Negishi weight
β	waste percentage
μ	labor cost for generating organic waste
$\sigma^{k,l,}$	substitution elasticity between labor and capital
$\sigma^{k,l,,e}$	substitution elasticity between labor capital and emission rights
$\sigma^{l,h}$	substitution elasticity between low and high quality organic waste
$\sigma^{r,o}$	substitution elasticity between rest waste and organic waste
$\sigma^{m,s}$	substitution elasticity between landfilling and incineration
ξ	subsidy wedge
A	technology parameter waste treatment units
F	flat fee for waste collection

$\overline{K}$	endowment of capital
$\overline{L}$	endowment of labor
LST	lump sum transfer to keep income of government constant
p	price
P_c	price including subsidy
T	total costs subsidy waste collection
Y^0	initial income

Variables in GAMS specification

Symbol	Description
e	emission rights
k	capital use
l	labor use
q	production
TWF	total welfare
u	utility
ts	transport services
w	generation of waste
wts	waste treatment services
x	consumption

Part III

Conclusions and recommendations

7 Summary, conclusions and recommendations

7.1 Introduction

Each year the Netherlands spends approximately 3.5 billion Euros on the treatment of waste. Waste treatment not only costs our society a lot of money, but it also creates environment problems. For example, the landfilling of waste generates about 40% of total methane emissions in the Netherlands. The Dutch government is, therefore, very eager to reduce waste generation as much as possible. Annually, households and industry generate about 12 Mtonnes of waste. Although this is still a considerable amount of waste, the quantity of waste generated today is already 40% lower than the quantity generated in 1990. In particular, the industrial sector and the construction and demolition sector have started to recycle far more waste as compared to 1990. These sectors recycle about 90% of all their waste, which naturally results in far less waste incinerated or landfilled. Private households do not recycle nearly as much. On average, only about 40% of all municipal solid waste is recycled. The amount of solid waste generated increases every year due to economic growth; though thus far it has not been possible to decouple economic growth and the generation of municipal solid waste.

Although the Dutch government has tried to stimulate waste separation and recycling since the beginning of the 1990s, the results are not promising. Since 1995, for example, municipalities have been obliged to collect organic waste separately from rest waste. Despite the initial increase of organic waste collected in 1995, since that time the amount of organic waste generated has hardly increased at all. Annually, about 12% of all municipal solid waste is collected as organic waste and composted. This, however, does not necessarily mean that all organic waste is collected separately. On average, about 34% of the waste collected as rest waste is actually organic waste. If households separated all organic waste from rest waste, a significant saving on the expenses of waste treatment, about 170 million Euros, could be made. The incineration of rest waste is namely twice as expensive as the composting of organic waste.

The failure to increase recycling and waste separation has been caused by a number of distortions in the municipal solid waste market. It is important to analyze how these market distortions can be resolved. Only if these distortions are resolved, will it be possible to reduce waste generation, and thus the social costs of waste treatment also.

For this reason, this thesis has aimed to analyze the municipal solid waste market in the Netherlands. The main objectives, as formulated in Chapter 1, were: (1) To analyze how the incentive structure of the consumers, emission restrictions, interrelations between the municipal solid waste sector and the rest of the economy and the spatial aspects of the waste problem influence the optimal municipal solid waste management plan. (2) To asses whether a flat fee-pricing system, a unit-based pricing system for the collection of rest waste, a unit-based pricing system for the collection of organic and rest waste, or a recycling subsidy is the preferable policy option to minimize the social costs of municipal solid waste treatment. (3) To gain insight into how to develop a more efficient municipal solid waste management plan, which solves inefficiencies caused by market distortions present in the municipal solid waste market.

On the basis of these objectives, five research questions were formulated in Chapter 1. In this chapter, the most important results of the thesis will be presented by answering these research questions.

7.2 The economic and environmental topics concerning the municipal solid waste management problem

Research question 1:

What are the most important environmental and economic topics with regard to the municipal solid waste management problem?

Besides financial costs, waste treatment also creates many environment problems. As policy makers are concerned about the state of the environment, they have adopted the concept of the waste hierarchy as a basis for waste policies. The waste hierarchy is a method to prioritize different waste handling options. According to the waste hierarchy, the prevention of waste has the absolute preference, followed by recycling and re-use, composting, incineration and, as the last option, landfilling.

Since landfilling is the least preferred waste treatment method, the government has tried to prevent the landfilling of waste as much as possible. For this reason, two policy measures were introduced, namely a ban on landfilling and a landfilling tax. As a result, landfilling of waste decreased dramatically. In 2002, for example, 5157 Ktonnes of waste were landfilled; compared to 1992 this meant a decrease of more than 60%. Increasingly less waste has thus been landfilled and increasingly more waste recycled, composted, or incinerated. In this way, the government has tried to control waste flows in an environmentally sound manner and minimize social waste treatment costs. Although it may seem that the landfilling tax has been rendered obsolete by the ban on landfilling, the landfilling tax is still necessary to control the waste flows. Producers were often granted an exemption from the ban on landfilling, because the incineration capacity in the Netherlands was not large enough to incinerate all combustible waste. Producers were very keen on getting such an exemption, since landfilling was far cheaper than incineration or recycling. The introduction of a landfilling tax, which raised the price of landfilling above the price of incineration, provided the necessary price-incentive to the production sectors to recycle or incinerate their waste as much as possible.

Although it serves a useful purpose, the waste hierarchy should not be considered the only rule that may determine the optimal waste treatment option. It is important to bear in mind that, in some cases, a strict application of the waste hierarchy can do our society more harm than good. Each waste handling option, *i.e.* recycling, re-use, composting, incineration, or landfilling, may lead to environmental damage. Since each waste treatment unit is unique and operates quite differently to the others, is it important to realize that one waste treatment unit will process waste in a more environmental friendly way than another. Large waste treatment units, for example, have more capital to invest in emission reduction measures than smaller installations. For this reason, is it important to include economies of scale in the analysis in order to determine the socially most efficiently waste management plan.

The minimization of social waste treatment costs is not the only solution to the waste management problem, it is also important to minimize the amount of waste generated. The production sectors have a strong financial incentive to decrease their generation of waste due to a significant landfilling tax. The treatment of waste has become expensive and thus the production sectors have increasingly invested more in advanced recycling technologies.

The municipal solid waste problem is much more difficult to control. In most cases, households pay a flat fee for the collection of waste. This flat fee is not related to the amount of waste that is generated and collected. Thus, an increase in the cost of waste treatment has no or only a negligible effect on the generation of municipal solid waste. In addition, the municipal solid waste stream is very diverse. Municipal solid waste consists of several different waste materials. This makes it difficult to recycle waste to a large extent.

Due to the presence of market distortions in the municipal solid waste market, waste treatment costs are higher than desirable. Based on recent literature as described in Chapter 2, three market distortions can be distinguished: (1) the flat fee (2) indirect subsidizations of the use of raw materials and (3) ‘killer contracts’ between waste treatment facilities and municipalities that specify the quantity of waste the municipalities must deliver to the facilities and the price they will have to pay to dispose of this waste.

Chapter 2 provides an overview of the recent literature, which analyses possible solutions to these market distortions. From the literature, it may be concluded that particularly the flat fee has significant consequences for the total generation of municipal solid waste. Therefore, the flat fee must be replaced by another pricing system. A possible replacement of the flat fee is a unit-based price for waste collection. The unit-based price will decrease waste generation. However, given that unit-based pricing can stimulate illegal disposal, it is important that the prevention of illegal dumping of waste is given special attention.

Recent studies, such as Palmer and Walls (1997), Fullerton and Kinnaman (1998), and Walls and Palmer (2000), have neglected several important aspects of the municipal solid waste market, thus their analyses of the effects of introducing unit-based pricing for waste collection are incomplete. First of all, the phenomenon 'waste leakage' has been left out of the analysis. Consumers determine the quality of waste they generate. They can choose to generate rest waste that is 'polluted' with, for instance, organic waste, glass, or paper. This has no direct consequence for the treatment costs of waste as experienced by the individual consumer, as organic waste, or paper can be incinerated without any problems. It is, however, lamentable that paper, and organic waste are not recycled or composted, since recycling and composting are much cheaper and generate less environment pollution than incineration. It is also possible that consumers pollute glass, paper, or organic waste with rest waste. This has more serious consequences. Polluted glass and paper must first be cleaned before it can be recycled. Heavily polluted organic waste will be rejected by the composting unit, which means that this polluted organic waste must be incinerated. A composting unit cannot compost polluted organic waste, because the quality of compost produced from polluted organic waste is not good enough to sell[1]. The introduction of a unit-based price for waste collection may increase this kind of undesirable pollution of waste and it is therefore crucial that the possibility of 'waste leakage' be included in the analysis.

Secondly, an analysis of the spatial aspects of the waste problem within a general equilibrium framework is missing in the literature. The costs of waste treatment are significantly influenced by the location of the waste treatment unit, the transport costs and economies of scale of the waste treatment units in question. These costs also depend on the quantity as well on the quality of the waste generated. A change in the composition of the waste stream, due to the introduction of a new policy measure, can have significant consequences for both the optimal location of the waste treatment facility and waste treatment costs. The waste treatment costs may in turn significantly influence the amount of municipal solid waste generated. It is important that these

[1] Even if polluted organic waste is cleaned, the amount of heavy metals in the organic waste will be so high that it is impossible to produce high quality compost from the waste.

interactions between the spatial aspects of waste treatment and waste generation be included in the analysis.

To summarize, the economic and environment issues in relation to the waste management problem can be divided into two areas of interest. Firstly, the problem of how waste should be treated so that the costs to society are as low as possible must be solved. Waste treatment generates both financial and environmental costs. Since the environmental costs are not internalized in the price of waste treatment and thus the social waste treatment costs are higher than optimal, the Dutch government has adopted the concept of waste hierarchy as the basis for an optimal waste management plan. Scientific research has warned that a strict regime, such as the waste hierarchy, can lead to more waste treatment costs than necessary. The existing literature demonstrates that it is more appropriate for the choice between waste treatment options to be determined on an individual case basis. Secondly, the problem of how to minimize waste generation should be answered. In the Netherlands, consumers generate most of the waste to be treated, *i.e.* organic waste and rest waste. The present structure of the municipal solid waste market leads to an inefficient high quantity of waste. Policy tools, such as a differentiated price for waste collection, a recycling subsidy, or an advanced disposal fee, which internalizes the costs of waste treatment in the price of a product, could be used to decrease waste generation.

In this thesis, these waste management issues have been linked. The optimal waste treatment method and the minimization of waste treatment costs are thus coupled. The quality and quantity of municipal solid waste influence the treatment of the waste. For example, high quality organic waste may be composted, but low quality organic waste can only be incinerated or landfilled. Moreover, the treatment costs can influence the amount of municipal solid waste generated. In an undistorted market, for example, higher treatment costs result in the generation of less waste.

7.3 The problems concerning the flat fee for waste collection

Research question 2:

How does the market distortion caused by the flat fee-pricing system influence municipal solid waste generation and how can these negative effects be sufficiently reduced?

Most municipalities in the Netherlands charge a fixed amount of money, the so-called flat fee for collection of waste. Although this amount is determined on the basis of the household size, the amount is independent from the actual quantity of waste generated. Households have no financial incentive to decrease waste generation. Empirical studies show that the introduction of a unit-based price for waste collection, the so-called unit-based price, can have a strong influence on the quantity of waste generated. In those municipalities that introduced a unit-based price, the quantity of rest waste declined by about 20 to 30% (KPMG, 1999). However, the extent to which this reduction may be attributed to the introduction of a unit-based price is unclear. Thus far, the introduction of a unit-based price in the Netherlands has always been combined with programs to stimulate both recycling and prevention of waste. Therefore, recycling programs may be responsible for part of the observed reduction. Theoretical studies warn that price-differentiation may lead to the illegal dumping of waste, but up until now there has been little empirical proof to support this assertion.

Municipalities may choose between different forms of price differentiation. The possibilities are price differentiation on the basis of volume, weight, or frequency of collection or a combination of these forms. It is also possible to sell special waste bags, in which the consumers have dispose of their waste; this system is called price differentiation on the basis of an ‘expensive bag’. Each of these unit-based pricing schemes has advantages and disadvantages, but the greatest advantage of each of these schemes is that they decrease the generation of municipal solid waste. In this thesis, I have shown, with the use of a general equilibrium model, that the flat fee indeed results in inefficiently high waste generation, just as empirical studies have shown.

Chapter 4 deals with a comparison between four policy instruments: 1) the introduction of a recycling subsidy combined with a flat fee, 2) the introduction of a unit-based price for waste collection, 3) the introduction of a recycling subsidy combined with an unit-based price and 4) an advanced disposal fee on the price of a consumption good.

The results show that a flat fee distorts the municipal solid waste market. The introduction of an incentive to increase recycling, by means of a recycling subsidy, has no significant effect, since consumers are given no incentive to reduce their generation of rest waste. Although recycling becomes cheaper due to the recycling subsidy, the collection of rest waste is free of charge, which therefore means that consumers do not invest in efforts to increase recycling. The introduction of unit-based pricing in the model has more effect on waste generation and recycling. The quantity of rest waste generated decreases by about 2% and recycling increases by about 6%. If the introduction of unit-based pricing is combined with the introduction of a recycling subsidy, the effects are even more impressive. Rest waste decreases by about 70% and recycling increases by about 266%. It should be noted that no technical upper boundary is placed on the quantity of waste that can be recycled, in reality this is not possible, and therefore the model may overestimate the effect of the policy instrument. The advanced disposal fee on the price of consumption goods also decreases waste generation, but to a lesser extent. Waste generation decreases by

0.5% when an advanced disposal fee, which covers the actual waste treatment costs, is introduced. The advanced disposal fee, in contrast to a unit-based price, stimulates waste prevention but not recycling. Thus this policy instrument only has a limited impact on waste generation.

Considering the results of this model, the introduction of unit-based pricing is the most effective method for decreasing waste generation, especially if combined with a recycling subsidy. The introduction of an advanced disposal fee is much less efficient. Although it stimulates prevention, the reduction of rest waste generated is rather small. The advanced disposal fee is not as effective as the unit-based pricing scheme because: firstly, the advanced disposal fee is too low to change consumption patterns significantly[2] and secondly, the waste collection fee does not stimulate recycling since consumers no longer have to pay a fee for the actual collection of waste. They only pay a fee for collection and treatment of waste while buying the product, thus they do not have a price-incentive to recycle.

The model, presented in Chapter 4, does not include the possible evasive behavior of consumers. Consumers can easily reduce the quantity of rest waste they generate by disposing of it in the organic waste bin. Moreover, the link between generation of waste and treatment of waste, which was discussed in Section 7.2, is not included in this model. The following two research questions will deal with the question of how these two issues influence the effectiveness of unit-based pricing.

7.4 The problems of waste leakage

Research question 3:

How great a problem is waste leakage and how is waste leakage influenced by household attitudes?

Municipalities can introduce unit-based pricing for all waste streams or only for a single waste stream. In practice, unit-based pricing has been introduced for the collection of rest waste and/or organic waste. Recyclable waste, such as glass and paper, is collected free of charge. Whether one should introduce unit-based pricing for the collection of organic waste is not an easy question to answer. If a municipality wants to stimulate separation of waste, it is undesirable to charge an equal price for the collection of organic and rest waste. As the separation of waste requires extra

[2] The advanced disposal fee internalizes the costs of waste treatment in the price of the consumer good. As the disposal costs of a good are generally much lower than the production costs of the good, consumers have only a very small incentive to change their consumption patterns.

effort on the part of consumers, it should be stimulated by a price-incentive. Moreover, the treatment of organic waste is far less costly than the treatment of rest waste and, therefore, it is difficult to explain to the consumers that the price of collecting organic waste is equal to the price of collecting rest waste. If municipalities charge the same price, they can lose a lot of goodwill, possibly resulting in a situation where consumers are unwilling to put extra effort into waste separation and are more inclined to dispose of waste in an illegal manner.

Waste leakage is one of the possible options consumers have to dispose of rest waste. In this case, consumers throw rest waste away with organic or recyclable waste and thus pollute these waste streams. Households need not to be afraid of being penalized for this undesirable behavior. It is, for example, very expensive to check the quality of the organic waste during collection. The quality of the waste is instead checked at the composting unit. At this point, it is extremely difficult to determine exactly which household has polluted the waste stream. The only option a municipality has in such a case is to appeal to the whole district about the quality of the waste the district has supplied.

In this thesis, I have developed a general equilibrium model to study the problems caused by waste leakage. In Chapter 5, I demonstrate how the effects of waste leakage can be a serious impediment to the introduction of unit-based pricing for the collection of municipal solid waste. The analysis is restricted to the introduction of weight-based pricing for waste collection, but a similar analysis may be made for the introduction of price differentiation on the basis of collection frequency or volume. This only affects the method according to which waste generation is calculated in the model.

The analysis in Chapter 5 shows that consumers start to pollute organic waste due to the introduction of unit-based pricing for collection of rest waste. After the introduction of unit-based pricing, consumers generate about 10% less rest waste. Consumers reduce their generation of rest waste by substituting it for low quality as well as high quality organic waste. High quality organic waste consists of 100% organic waste. Low quality organic waste consists of 70% organic waste and 30% rest waste (non-organic residue). On average, after the introduction of price differentiation consumers generate about 46% more low quality and 22% more high quality organic waste. This means that the percentage of rest waste thrown away with organic waste increases from 5.5% to 6.7%.

Not every household reacts in the same way to the introduction of price differentiation. Depending on the preferences of the consumers, they will be inclined to behave in a more or lesser environment friendly fashion. A ‘traditional’ consumer has little interest in the environment and will be more inclined to pollute waste than a ‘green’ consumer, who cares for the state of the environment. Model results in

Chapter 5 demonstrated that 'green' consumers generate about 14% less rest waste when unit-based pricing is introduced. They will approximately generate 14% more low quality organic waste and 12% more high quality organic waste. The percentage rest waste that is thrown away with organic waste remains constant, about 3.1%. The 'traditional' consumers generate 8% less rest waste. They generate approximately 63% more low quality organic waste and 41% more high quality organic waste. This means that the percentage of rest waste thrown away with organic waste increases from 9.8% to 11%.

Although the overall percentage of rest waste thrown away with organic waste, *i.e.* by both 'green' and 'traditional' consumers, does not increase significantly, *i.e.* from 5.5% to 6.7%, it is important to bear in mind that this is an average percentage. Waste is collected in relative small quantities; approximately 28 tonnes of waste fit in one garbage truck. Some districts in a municipality may be house comparatively more 'green' consumers, while other districts may house proportionally more 'traditional consumers'. In a district with many 'traditional' consumers, the quality of organic waste collected will decline far more than in districts with a lot of 'green' consumers. This can mean that the organic waste collected in these districts cannot be composted. IPH (1995) shows that composting units will in general reject organic waste of such low quality as generated by the 'traditional' consumers.

The introduction of unit-based pricing will lead to the situation where waste collected in some districts is rejected by composting units. Particularly in large cities with a high percentage of 'traditional' consumers, the organic waste will be heavily polluted. When municipalities consider introducing unit-based pricing, it is important that they bear in mind that waste leakage may occur. Large cities can only consider unit-based pricing in those districts where relatively many 'green' consumers live. Such a kind of selective regulation is used more often: for instance, it has already been introduced in large cities, as Utrecht and Amsterdam. In these cities, organic waste and rest waste are collected separately only in a restricted number of districts. In the other districts, the quality of the organic waste was found to be too low to compost and thus could not justify the additional collecting cost.

A possible solution for the problem of waste leakage is the introduction of a unit-based pricing for rest waste as well as organic waste. Since consumers pay the same price for collection of both rest and organic waste, they have no incentive to pollute organic waste. As was already observed in the introduction to this section, the danger of such a system is that the goodwill of the consumer may be compromised. This is illustrated by the quantitative results of Chapter 6. If unit-based pricing for the collection of rest waste as well as organic waste is introduced, consumers do not start to generate more organic waste. The 'traditional' consumers even generate more low quality organic waste and less high quality organic waste. Only if it is possible to increase the price of collection of low quality organic waste as compared to the price of collection of high quality organic waste, does unit-based pricing result in a decrease of the generation of low quality organic waste.

7.5 Choice of the optimal location of waste treatment units

Research question 4:

How is the choice of the optimal location of waste treatment facilities influenced by the quantity and quality of municipal solid waste generated by consumers and, moreover, how will the spatial aspects of the municipal solid waste management problem in turn influence the successfulness of introducing unit-based pricing?

In principle, municipalities determine how waste is treated and at which location. Municipalities are, however, not completely free in this choice. Depending on the type of waste collected, they will decide how to process the waste. The collected organic waste will always go to a composting facility. The collected rest waste is sent to either an incineration plant or a landfill site. The quantity of waste collected influences the optimal waste treatment method and location. Due to economies of scale in waste treatment, large quantities of waste can be processed more cheaply than small quantities. Economies of scale play a particularly important role in the case of incineration, also due to environment regulations that specify the extent of emissions permitted. Thus, it is much cheaper for a municipality to allow waste to be treated in a large waste treatment unit than in a small one, although this will increase transport costs.

The introduction of unit-based pricing alters the composition of the collected municipal solid waste stream. Municipalities collect less rest waste, and more organic waste. The quantitative results of Chapter 6 show that if unit-based pricing for the collection of rest waste is introduced that municipalities will collect about 6% less rest waste. If a unit-based price for the collection of rest waste as well as organic waste is introduced, the total quantity of organic waste and rest waste remains constant.

The change in composition of the municipal solid waste stream has significant consequences for the optimal location choice of waste treatment units. If more organic waste is collected, it is attractive for organic waste to be treated in a large composting unit. If the quality of organic waste declines sharply, it is also attractive to treat waste at lower costs.

The analysis in Chapter 6 demonstrates that the quality of the organic waste in large cities declines significantly due tot the introduction of unit-based pricing. As a result, these municipalities treat their waste in large composting units. This means that transport costs increase significantly, about 60%. As a result, the total costs of waste treatment increase slightly after the introduction of unit-based pricing, from 239.6 million Euros to 240.7 million Euros.

Since municipalities have negotiated contracts with incineration plants, composting units and landfill sites, it is not possible to switch from one installation to another in the short term. This does not, however, mean that the effects of changes in the composition of the municipal solid waste stream due to the introduction of unit-based pricing should be disregarded. By introducing unit-based pricing, the composition of the municipal solid waste stream permanently changes. In the long run, municipalities are not bound by contracts with waste treatment units and they can switch between these waste treatment units.

The results in Chapter 6 show that the environment effects due to the introduction of unit-based pricing are ambiguous. If unit-based pricing is introduced less rest waste will be generated. This means that less rest waste will have to be incinerated or landfilled, resulting in a decrease of both CO_2 and CH_4 emissions. Although more organic waste is generated, part of it is of such low quality that it has to be incinerated, resulting in an increase of CO_2 emissions. Moreover the transport of waste increases, which generates more NO_X emissions. The net effects are an increase of CO_2 emission by 6.8% and an increase of NO_X emissions by 17%. CH_4 emissions decrease by 4%.

It may be clear that there is a strong interaction between quality of waste and treatment of waste. Although unit-based pricing leads to a decrease in the amount of rest waste generated, the costs of waste treatment do not decline. The results in Chapter 6 illustrate that waste leakage influences the optimal treatment option so that municipalities will switch from small composting units to large ones, thus increasing transport costs. Chapter 6 also shows that the introduction of unit-based pricing is only attractive in small municipalities with proportionally more ‘green’ consumers. For larger cities, the consequences of waste leakage are too far-reaching to justify the costs of introducing unit-based pricing.

7.6 Policy recommendations

Research question 5:

Which kinds of policy changes can be recommended to minimize the total social costs of municipal solid waste treatment for our society?

In this thesis four policy options have been analyzed, namely 1) a flat fee, 2) a recycling subsidy, 3) a unit-based price for collection of rest waste and 4) a unit-based price for collection of rest waste as well as organic waste.

As mentioned above, most municipalities in the Netherlands charge a flat fee for the collection of municipal solid waste. If a flat fee is charged, consumers have no incentive to separate or recycle waste, thus consumers generate more waste than is social desirable. The main advantage of the flat fee-pricing scheme is the low collection cost, since municipalities do not have to keep track of the quantity of waste generated by an individual household. Another advantage is that consumers have no incentive to illegally dump their waste or to pollute the recyclable or organic waste stream.

To promote recycling, the municipality can choose to subsidize it. Recycled material is on average more expensive than virgin material; this makes recycled material difficult to sell. By subsidizing the recycling process, the government can stimulate the use of recycled materials. If, however, a recycling subsidy is combined with a flat fee for waste collection, then the introduction of a recycling subsidy will have little effect. The flat fee distorts the municipal solid waste market, for consumers have no price incentive to reduce the generation of rest waste. Unless additional incentives are provided, they will not recycle more, as this is costly for the consumers, not even if it is deemed socially very desirable. For this reason, it is not advisable to for a recycling subsidy to be introduced without accompanying measures, if the municipality in question also charges a flat fee for waste collection.

If a recycling subsidy is jointly implemented with a unit-based pricing scheme, then the recycling subsidy has a positive effect. More waste will be recycled and less virgin materials will be used. This effect, although to a lesser extent, will also be accomplished through the introduction of unit-based pricing without a recycling subsidy. Municipalities may consequently introduce unit-based pricing and leave the use of recycled material with respect to virgin materials up to the market.

By introducing unit-based pricing, municipalities can stimulate recycling and waste separation. Due to the introduction of unit-based pricing, consumers are given a price incentive to decrease the generation of rest waste. Unit-based pricing on basis of weight is especially effective in reducing the generation of rest waste. This greatly decreases the money spent on the incineration and landfilling of waste.

Municipalities can choose whether to introduce unit-based pricing for rest waste, or for rest waste as well as organic waste. If they introduce unit-based pricing for rest waste as well as organic waste, the consumers will not be stimulated to put extra effort into the separation of organic and rest waste. Having to separate waste may also diminish the goodwill of consumers. The consumers have to invest a lot time and energy in separating rest waste and organic waste. When consumers are not financially compensated for such efforts, they may be inclined to stop separating waste partly or altogether. If they are forced to pay the same amount for the collection of organic waste and rest waste, their resistance will be even greater. Organic waste is much cheaper to process than rest waste and the marginal costs of treating organic waste are less than the marginal costs of treating rest waste. Therefore, by demanding an equal price for the collection of both organic and rest waste, the consumers are more or less financially penalized for waste separation.

The introduction of unit-based pricing for the collection of rest waste stimulates consumers to generate less rest waste. They can accomplish this by separating organic waste from rest waste or by preventing waste generation. A major disadvantage of this policy instrument is the possibility of waste leakage. If organic waste is heavily polluted by rest waste, it can no longer be composted. In this case, the polluted waste will have to be incinerated, thus leading to significantly higher waste treatment costs.

To summarize, the question of which policy option can be recommended cannot be answered unequivocally. The introduction of a recycling subsidy is not recommended, since the municipal solid waste market is distorted due to the flat fee for waste collection; a recycling subsidy will, therefore, have no effect on the recycling behavior of consumers. If the government wants to stimulate recycling, the introduction of unit-based pricing for waste collection will be more effective. Unit-based pricing for the collection of rest waste provides the greatest incentive for waste separation, but also provides the prime incentive for waste leakage. In smaller municipalities, the introduction of price differentiation for both organic waste and rest waste is possible. In smaller municipalities with proportionally more 'green' consumers, it is also possible to introduce price differentiation for rest waste only. With the help of a numerical analysis, in Chapter 6 I have shown that the danger of waste leakage is not so great in these small municipalities and therefore the benefits of introducing price differentiation outweigh the costs. Municipalities with proportionally more 'traditional' consumers should be aware of problems created by waste leakage. In these municipalities, unit-based pricing for rest waste as well as organic waste performs far better. In larger municipalities, only the introduction of unit-based pricing for rest waste as well as organic waste is advisable. Due to the problems caused by waste leakage, the introduction unit-based pricing for rest waste alone is not attractive.

If the problems created by waste leakage can be solved, unit-based pricing is certainly an attractive policy instrument. Unit-based pricing is the only policy instrument that can potentially stimulate prevention, recycling, and waste separation simultaneously. Pollution of the recyclable waste steam, like glass and paper, is expensive, but not as problematic as the pollution of the organic waste stream. Heavily polluted recyclable waste can be cleaned and then recycled, whereas heavily polluted organic waste cannot be composted even if it is cleaned.

It is possible to clean polluted organic waste, consider, for example, the cleaning techniques employed by the VAGRAM installation in Groningen, but the quality of the compost produced from polluted organic waste is not good enough to sell. It is important that the government stimulates the development of better separation and cleaning techniques, because the introduction of unit-based pricing is an important means to minimize the generation of municipal solid waste and to increase recycling percentages.

7.7 Modeling of the waste problem

In addition to the above research questions, the following three modeling questions were formulated in Chapter 1:

- How can interactions between the waste sector, government policies, and the rest of the economy be modeled?
- How can the flat fee-pricing system be introduced to a general equilibrium setting?
- How can spatial aspects of the waste management problem, such as a fixed set of possible location of waste treatment centers, economies of scale and transport costs, be introduced to a general equilibrium framework?

In this thesis, I have demonstrated how the different aspects of the waste problem, as formulated in the model questions, can be implemented in a general equilibrium framework. In particular, it was difficult to incorporate the flat fee into a general equilibrium model.

By using the subsidy-*cum*-tax system, the problem of a marginal zero price for waste collection can be avoided. In this system, consumers pay an equilibrium price for collection of waste. The government reimburses consumers with a subsidy that covers the exact price of waste collection. Thus, the price of waste collection, as perceived by the consumers, equals zero. The consumer pays a direct tax for the collection of the waste to the government, the so-called flat fee. In sum, in this system consumers pay a direct tax for waste collection, but this tax is not related to the quantity of waste they generate.

In Chapters 4, 5 and 6, three different general equilibrium models have been developed. All these models were built in the Negishi format. The models have been constructed from a relatively simple (Chapter 4) to a detailed and complex analysis of the waste problem (Chapter 6).

To summarize, the most complex model developed in Chapter 6 had the following characteristics:

1) The economy is divided in four municipalities. Within a municipality, two types of consumers are distinguished: a 'green' consumer and a 'traditional' consumer. The 'green' consumer is more environment friendly oriented than the 'traditional' consumer. The municipalities differ in the share of 'green' and 'traditional' consumers.

2) Municipalities can treat waste in composting units, incineration plants, and landfill sites. Three sizes are distinguished for each waste treatment unit: a small-sized, a medium-sized and a large-sized unit. Economies of scale influence waste treatment costs; a large installation treats waste for a lower price than a small installation. Apart from waste treatment costs, municipalities also face transport costs. As large installations are, on average, located further away from the municipalities, transport costs are higher for these waste treatment units.

3) Policy measures, such as emission reductions, flat fees and unit-based pricing, can be included in the model without any problems.

The models presented in this thesis make it possible to analyze the effects of introducing price differentiation. New to the analysis is the explicit modeling of the quality of waste, the possibility of waste leakage, the link between production of municipal solid waste, the collection of waste by municipalities, and the treatment of waste by waste treatment units and finally the modeling of spatial aspects of waste treatment within a general equilibrium framework.

In contrast to the existing literature, in this thesis a link is made between the generation and treatment of waste. Thus, a detailed analysis of the cost-effectiveness of the introduction of unit-based pricing could be made. In this thesis, I demonstrated that a decrease of the quality of waste, due to the introduction of unit-based pricing, has a significant effect on the costs of waste treatment and thus on the cost-effectiveness of unit-based pricing also.

By using a general equilibrium framework, it was possible to analyze these relations in detail. A general equilibrium model describes all relevant markets in the economy, calculates the interactions between the different markets, and forms a closed system. The results of the models suggest that the success of unit-based pricing to a large extent depends on the preferences of the consumers. In those districts, in which proportionally more ‘traditional’ consumers live, unit-based pricing will not be successful.

By modeling both the consumption and the production sector, I was able to show that, against expectations, unit-based pricing alone is not suitable for stimulating prevention. The price incentive in the waste sector is too small to significantly change consumption patterns. If the government wants to stimulate prevention, they will have to consider other policy measures.

As demonstrated in this thesis, the preferences of the consumers, the location and the economies of scale play an important role in determining the optimal waste management policy. This means that a national waste management plan for municipal solid waste can only be successfully designed if specific local circumstances are explicitly considered. The success of unit-based pricing will depend on the share of ‘green’ and ‘traditional’ consumers living in the municipality, or living in a district of the municipality. Thus, each municipality will have to decide for itself whether unit-based pricing will be a success or not.

7.8 General conclusions

To summarize, this thesis has contributed to our understanding of the impact of the introduction of unit-based pricing.

1) Unit-based pricing will be successful in some municipalities, but not in all due to the possibility of waste leakage. All municipalities will have to accept that some amount of waste leakage occurs, but municipalities with proportionally more ‘green’ consumers will have a lower increase of waste leakage and thus the costs-effectiveness of unit-based pricing is greater in these municipalities. Each municipality should analyze for itself whether the introduction of unit-based pricing is cost-effective. A national waste management plan for municipal solid waste that does not consider the specific characteristics of municipalities and households will therefore be less than optimal.

2) Unit-based pricing is not suitable for stimulating prevention. The results of this thesis demonstrate that a policy change in the waste sector is not sufficient to shift consumption patterns. Unit-based pricing, however, is suitable to solve the market distortion caused by a flat fee pricing system. If unit-based pricing is introduced, other policy tools, such as a recycling subsidy, are more effective. If, for example, a recycling subsidy was to be implemented in combination with a flat fee for waste collection, the recycling subsidy would not stimulate recycling due to the market distortions created by the flat fee-pricing scheme.

3) The impact of the quality and quantity of waste on the costs of waste treatment can be analyzed by explicit modeling of the link between waste generation and waste treatment, thus a more detailed analysis about the cost-effectiveness of unit-based pricing can be made. This thesis shows that the introduction of unit-based pricing is by no means as beneficial for the environment as expected on the basis of the reduction of waste generation. Unit-based pricing decreases generation of rest waste, but also decreases the quality of organic waste. Consumers not only start to separate more organic waste from rest waste, but also pollute more organic waste with rest waste. Due to this waste leakage effect, the costs of composting, the transport costs, and the corresponding emission costs also increase.

7.9 Research recommendations

In this thesis, a model to analyze the effects of waste leakage has been developed. The model calculates the impact unit-based pricing has on the quality of organic waste and the subsequent costs of waste treatment. In this model, I have included three types of emissions, two types of consumers, four municipalities, transport costs, economies of scale and differentiated prices for composting high and low quality organic waste. The model is used in a stylized example with numerical data based on the 'Randstad' in 2000.

The most complex model, as described in Chapter 6, incorporates a large number of aspects of the waste market in the analysis. However, some more aspects could be added to the model to predict the effects of unit-based pricing with greater certainty.

Firstly, the model could be expanded to include several waste streams. In this thesis, I have concentrated on organic waste and rest waste. Other recyclable waste streams, like glass, paper, and aluminum, could also be included in the analysis.

Secondly, it would be interesting to include home composting as well as illegal dumping in the model. The first option provides consumers with a legal option to decrease organic waste generation; home composting on large scale can, however, lead to problems for the composting industry. The second option provides consumers with an illegal option for getting rid of rest waste. This option will, of course, increase the social costs of waste treatment and can be a serious impediment to the introduction of unit-based pricing.

Thirdly, it is important to examine how prevention can play a role in solving the waste management problem in greater detail. Although I considered prevention as an option in this thesis, prevention was modeled rather simplistically. In the model in Chapter 6, only two consumer goods were included, *i.e.* a consumption good with a large waste content and a consumption good with a small waste content. Consumers could influence waste generation by consuming less of the waste intensive good. In reality, consumers will primarily prevent waste by choosing products with less packaging material. It would be interesting to include the packaging degree of a product; the higher the packaging degree, the more waste is generated.

Fourthly, it would also be interesting to model the waste market without the assumption of perfect competition between waste treatment units. As contracts exists between waste treatment units and municipalities, waste treatment units are essentially monopolists. Therefore, the assumption of perfect competition is not realistic.

Finally, the costs of waste leakage should be empirically estimated. As of yet, no empirical data is available about the composition of organic waste stream and the costs of composting organic waste of various qualities. As a consequence, it was impossible to base the numerical values of several key parameters on real data. Since unit-based pricing has already been introduced in several municipalities in the Netherlands, it should be possible to collect empirical data about waste leakage in these municipalities.

Samenvatting, conclusies en aanbevelingen

Inleiding

Per jaar wordt in Nederland ongeveer 3,5 miljard Euro's besteed aan de verwerking van afval. Afvalverwerking kost onze samenleving niet alleen een hoop geld, het veroorzaakt ook milieuproblemen. Door het storten van afval wordt bijvoorbeeld circa 40% van de totale methaanemissies in Nederland gegenereerd. De overheid is er daarom op gebrand de afvalproductie in Nederland te verlagen. Per jaar wordt ongeveer 12 Mton afval geproduceerd door zowel huishoudens als de industriële sectoren. Hoewel dit nog altijd zeer veel is, wordt er al 40% minder afval geproduceerd dan in 1990. Deze reductie wordt voornamelijk veroorzaakt doordat de industriële sectors veel meer zijn gaan recyclen. Gemiddeld recyclen deze sectoren 90% van al hun afval. Dit heeft als resultaat dat steeds minder afval wordt gestort of verbrand. De resultaten voor het huishoudelijk afval zijn minder bemoedigend. Gemiddeld wordt slechts 40% van het huishoudelijk afval gerecycled. Ook stijgt het afvalaanbod nog elk jaar. Dit wordt veroorzaakt door een groeiende economie: het is nog niet gelukt om een ontkoppeling tussen de economische groei en huishoudelijke afval productie tot stand te brengen.

Hoewel de overheid al sinds de jaren negentig bezig is met het promoten van afvalscheiding en recycling, heeft zij nog weinig resultaat geboekt. Gemeentes zijn bijvoorbeeld sinds 1995 verplicht om Groente Fruit en Tuin-afval (GFT) gescheiden op te halen. Ondanks de aanvankelijke toename van GFT-afval in 1995 is sindsdien het aanbod van GFT-afval nauwelijks meer veranderd. Per jaar wordt circa 12% van het huishoudelijk afval als gescheiden GFT-afval ingezameld en gecomposteerd. Lang niet al het GFT-afval wordt gescheiden van het restafval. Gemiddeld bestaat het restafval dat wordt opgehaald uit 34% GFT-afval. Indien huishoudens al het GFT-afval zouden scheiden van het restafval, zou dit een belangrijke besparing op de kosten van afvalverwerking zijn, gemiddeld 170 miljoen. Verbranden van restafval is namelijk twee keer zo duur als het composteren van GFT-afval.

Het falen van het overheidsbeleid gericht op het promoten van recycling en afvalscheiding is toe te wijzen aan een aantal verstoringen in de afvalmarkt. Onderzoek is nodig naar het opheffen van deze marktverstoringen. Alleen dan zal het mogelijk zijn om de afvalproductie en daarmee de maatschappelijke kosten van afvalverwerking te verlagen.

Dit proefschrift richt zich daarom op een economische analyse van de verwerking van huishoudelijk afval in Nederland. De belangrijkste doelstellingen, zoals in Hoofdstuk

1 geformuleerd, zijn: (1) het verkrijgen van inzicht in hoe een meer efficiënt huishoudelijk afvalbeleidsplan ontworpen kan worden om de effecten van verschillende marktverstoringen in de afvalmarkt te voorkomen (2) het analyseren hoe een optimaal huishoudelijk afvalbeleidsplan wordt beïnvloed door invloeden van financiële en andere prikkels voor de consumenten, relaties tussen de afvalsector en de rest van de economie en ruimtelijke aspecten van het afvalprobleem, en (3) laten zien welke beleidsoptie het meest geschikt is om de maatschappelijke kosten van verwerking van huishoudelijk afval te minimaliseren. De onderzochte beleidsopties bestaan uit a) een vaste afvalstofheffing, b) een gedifferentieerd tarief op de opgehaalde hoeveelheid restafval, c) een gedifferentieerd tarief op de opgehaalde hoeveelheid van zowel restafval als organisch afval en d) een recycling subsidie.

Aan de hand van deze doelstellingen zijn in Hoofdstuk 1 vijf onderzoeksvragen geformuleerd. In dit Hoofdstuk worden per onderzoeksvraag de belangrijkste bevindingen van dit proefschrift gepresenteerd.

De economische en milieuvraagstukken rond het huishoudelijke afval probleem

Onderzoeksvraag 1:

Wat zijn de belangrijkste economische en milieuvraagstukken omtrent het afvalprobleem?

Afvalverwerking levert naast financiële kosten ook veel milieuproblemen op. De overheid heeft naar aanleiding van de bezorgdheid over deze milieukosten de ladder van Lansink als beleidsdoelstelling geaccepteerd. Dit is een methode om de verschillende afvalverwerkingsmethoden te prioriteren. Volgens de ladder van Lansink heeft preventie van afval de absolute voorkeur, daarna is recycling en hergebruik de verkiesbare methode, composteren verdient daarna de voorkeur, gevolgd door verbranding en als laatste optie het storten van afval.

Omdat storten als minst verkiesbare verwerkingsmethode wordt gezien heeft de overheid geprobeerd het storten zoveel mogelijk te voorkomen. Hiervoor zijn twee beleidsmaatregelen geïntroduceerd, namelijk een verbod op storten en een heffing op storten. De combinatie van deze twee maatregelen heeft geleid tot een belangrijke vermindering van de hoeveelheid gestort afval. In 2002 werd er bijvoorbeeld 5157 Kton afval gestort, dit is ten opzichte van 1992 een vermindering van ruim 60%. Steeds minder afval wordt gestort en steeds meer afval wordt gerecycled, gecomposteerd of verbrand. Op deze manier tracht de overheid het afval dat ontstaat op een verantwoorde manier en tegen de laagst mogelijke kosten te verwerken. Hoewel het verbod op storten de stortbelasting overbodig lijkt te maken, was het stortverbod alleen niet voldoende. Omdat de verbrandingscapaciteit in Nederland niet groot genoeg was om als het brandbaar afval te verwerken, konden producenten vaak (te) gemakkelijk aan een ontheffing voor dit stortverbod komen. Producenten waren zeer gericht op het krijgen van een ontheffing, aangezien storten een stuk goedkoper was dan verbranden of recyclen. Pas toen er ook een stortbelasting werd ingevoerd waardoor storten duurder werd dan verbranden, kregen de producenten een prijsprikkel om zoveel mogelijk afval zelf te recyclen of te verbranden.

De ladder van Lansink is echter niet zaligmakend. Het is belangrijk om in de gaten te houden dat een strak regime zoals de ladder van Lansink in sommige gevallen het milieu meer schade dan goed doet. Elke verwerkingsmethode van afval, of dit nu hergebruik, recycling, composteren, verbranden of storten is, brengt milieukosten met zich mee. Aangezien er grote verschillen zijn tussen individuele installaties is het belangrijk om in het achterhoofd te houden dat de ene installatie milieuvriendelijker zal verwerken dan de andere. Grotere installaties zullen bijvoorbeeld meer kapitaal hebben om te investeren in emissie beperkende maatregelen dan kleinere installaties. Daarom is het belangrijk dit soort schaalvoordelen mee te nemen om tot een maatschappelijk zo efficiënt mogelijke oplossing te komen.

Niet alleen door het minimaliseren van de maatschappelijke verwerkingskosten kan het afvalprobleem worden opgelost, maar ook door het beperken van de afvalstromen. Door de heffing op storten, hebben de productiesectoren een duidelijke financiële prikkel gekregen om hun afvalproductie te verminderen. Verwerking van afval is duur geworden en daarom zijn de industrieën steeds meer gaan investeren in technieken van recycling.

De huishoudelijke afvalstroom is moeilijker aan te pakken. Huishoudens betalen in de meeste gevallen een vaste afvalstoffenheffing aan de gemeentes. Deze afvalstoffenheffing is niet gekoppeld aan de afvalproductie van die huishoudens. Daarom zal het duurder maken van de afvalverwerking geen enkel effect of slechts een te verwaarlozen effect hebben op het huishoudelijk afvalaanbod. Daar komt nog bij dat de huishoudelijke afvalstroom zeer divers is. Huishoudelijk afval bestaat uit tal van verschillende componenten. Dit maakt het lastig om afval in grote mate te recyclen.

Door de aanwezigheid van marktverstoringen in de huishoudelijke afvalmarkt zijn zowel de afvalproductie als de kosten van afvalverwerking hoger dan wenselijk. De literatuur zoals beschreven in Hoofdstuk 2 besteedt aandacht aan: (1) de vaste afvalstofheffing (2) indirecte subsidiering van het gebruik van ruwe grondstoffen en (3) "wurgcontracten" tussen afvalverwerkingsinstallaties en gemeentes die bepalen hoeveel afval gemeentes leveren en tegen welke prijs.

Hoofdstuk 2 geeft een overzicht van recente literatuur over de mogelijke oplossingen van deze marktverstoringen. Uit de literatuur blijkt dat vooral de vaste afvalstoffenheffing significante gevolgen heeft voor de productie van afval. Daarom wordt voorgesteld de vaste afvalstoffenheffing te vervangen door een ander prijssysteem. Hierbij kan worden gedacht aan een gedifferentieerd tarief voor afvalcollectie. Het voorkomen van illegale dumping van afval vereist dan wel speciale aandacht.

Doordat de literatuur een aantal belangrijke elementen van de afvalmarkt buiten beschouwing heeft gelaten, is de analyse van de effecten van de invoering van tariefdifferentiatie nog incompleet. Ten eerste wordt er geen aandacht besteed aan het verschijnsel “afvalvervuiling”. De consumenten bepalen de kwaliteit van het afval dat zij aanleveren. Zij kunnen er voor kiezen om restafval aan te leveren dat “vervuild” is met bijvoorbeeld organisch afval, of papier. Dit heeft geen directe gevolgen voor de verwerkingskosten van dit afval zoals deze gevoeld worden door de consument, maar het is wel te betreuren dat het glas, papier en organisch afval niet gerecycled of gecomposteerd worden, aangezien recycling en composteren goedkoper zijn en minder milieuvervuiling opleveren. Een andere mogelijkheid is dat de consument het glas, papier of organisch afval gaat vervuilen met restafval. Dit heeft vervelendere consequenties. Vervuild glas en papier afval zullen eerst moeten worden gescheiden voordat het kan worden gerecycled. Vervuild organisch afval kan zelfs afgekeurd worden voor compostering, wat betekent dat dit afval zal moeten worden verbrand. Het kan voor composteerinstallaties namelijk onmogelijk zijn om vervuild organisch afval te verwerken, als de kwaliteit van compost die is vervaardigd uit vervuild organisch afval van dus danig slechte kwaliteit is dat het niet meer af te zetten is[1]. Het invoeren van een gedifferentieerd tarief op afvalcollectie kan dit ongewenste effect van afvalvervuiling bevorderen en daarom is het van belang dat in een analyse voor het oplossen van problemen in de afvalmarkt de mogelijkheid van afvalvervuiling wordt meegenomen.

Een tweede onderwerp dat ontbreekt in de literatuur is een analyse van de ruimtelijke aspecten van het afvalprobleem in een algemeen evenwichtsanalyse voor de afvalmarkt. De kosten van afvalverwerking worden in belangrijke mate bepaald door de vraag waar het afval wordt verwerkt en wat de transportkosten van afvalvervoer zijn. Deze kosten worden beïnvloed door zowel de kwantiteit als de kwaliteit van het afvalaanbod. Indien afvalstromen van samenstelling veranderen onder invloed van een beleidsmaatregel kan dit grote gevolgen hebben voor de optimale locatie van de afvalverwerkingsinstallaties en daarmee de totale verwerkingskosten van afval. De

[1] Ook als het vervuilde organisch afval een scheidingproces ondergaat is de hoeveelheid zware metalen in het organisch afval dusdanig hoog dat er geen goede kwaliteit compost van kan worden gemaakt.

totale kosten van afvalverwerking kunnen op hun beurt weer van grote invloed zijn op de hoeveel afval die aangeboden wordt. Het is dan ook belangrijk deze wisselwerking tussen ruimtelijke aspecten van afvalverwerking en afvalproductie in de analyse te betrekken.

Samenvattend zijn de belangrijkste economische en milieuvraagstukken in betrekking tot het afval probleem onder te verdelen in twee stromingen. Ten eerste de vraag hoe afval op een voor de maatschappij zo goedkoop mogelijke manier kan worden verwerkt. Afvalverwerking veroorzaakt zowel financiële kosten en milieukosten. Omdat milieukosten niet worden geïnternaliseerd in de prijs van afvalverwerking en daardoor de maatschappelijke verwerkingskosten te hoog zijn heeft de overheid als oplossing de Ladder van Lansink geïntroduceerd. Wetenschappelijk onderzoek waarschuwt ervoor dat het strikt houden aan de ladder van Lansink in sommige gevallen ervoor zal zorgen dat afvalverwerking duurder wordt dan noodzakelijk. Uit de literatuur blijkt daarom ook dat de keuze tussen afvalverwerkingmethodes beter per geval bepaald kan worden. Ten tweede de vraag hoe afvalproductie zoveel mogelijk kan worden geminimaliseerd. In Nederland wordt het grootste afvalprobleem veroorzaakt door consumenten. De huidige structuur van de huishoudelijke afvalmarkt veroorzaakt een inefficiënt hoge afvalproductie. Maatregelen zoals een gedifferentieerd tarief voor afvalinzameling, een recycling subsidie of een afvalstoffenheffing op de prijs van een product kunnen mogelijk de afvalproductie verminderen.

In dit proefschrift is een link gemaakt tussen deze twee vraagstukken. De optimale verwerking van afval en het minimaliseren van afvalverwerkingskosten zijn namelijk gekoppeld. De kwaliteit en kwantiteit van afval beïnvloeden de verwerking van het afval. Bijvoorbeeld een goede kwaliteit organisch afval kan gecomposteerd worden, een slechte kwaliteit kan alleen verbrand of gestort worden. Ook de verwerkingskosten zullen de productie van afval beïnvloeden. In een goed werkende markt zullen bijvoorbeeld hogere verwerkingskosten resulteren in een lager afvalaanbod.

De problemen rond een vaste afvalstofheffing

Onderzoeksvraag 2:

Hoe wordt de productie van huishoudelijk afval beïnvloed door de vaste afvalstofheffing en hoe kan op efficiënte wijze deze negatieve effecten worden gereduceerd?

De meeste gemeentes in Nederland vragen een vast bedrag, de zogenaamde vast afvalstofheffing voor het ophalen van afval. Hoewel dat bedrag vaak wel afhangt van het aantal personen in een huishouden, is het bedrag onafhankelijk van de daadwerkelijke hoeveelheid geproduceerd afval. De huishoudens hebben daarom geen financiële prikkel om hun afvalproductie te verminderen.

Uit empirische studies blijkt dat de introductie van een gedifferentieerd tarief op afvalcollectie, het zo gehete DIFTAR-systeem, een sterke invloed kan hebben op de afvalproductie. In gemeentes waarin een DIFTAR-systeem werd ingevoerd vermindert de productie van restafval gemiddeld met 20 à 30% (KPMG, 1999). Het precieze effect van de invoering van het DIFTAR-systeem is onduidelijk, aangezien de introductie van een gedifferentieerd tarief altijd gecombineerd wordt met programma's om mensen te stimuleren tot recyclen en meer bewust te maken van de negatieve gevolgen van reguliere afvalverwerkingsmethoden. Uit theoretische studies blijkt dat tariefdifferentiatie illegale dumping van afval kan veroorzaken, maar hier is tot nu toe weinig empirisch bewijs voor gevonden.

De gemeente kan kiezen tussen verschillende vormen van tariefdifferentiatie. Er kan worden gedacht aan tariefdifferentiatie op basis van volume, gewicht of frequentie van inzameling of een combinatie van de verschillende vormen. Ook is het mogelijk consumenten te verplichten het afval aan te bieden in speciale afvalzakken waaraan een heffing wordt opgelegd, dit systeem wordt tariefdifferentiatie op basis van "dure zak" genoemd. De hier genoemde systemen van tariefdifferentiatie hebben zo hun voor- en nadelen, maar het grote voordeel van elk van deze systemen is dat ze de afvalproductie doen verminderen.

In dit proefschrift heb ik met een algemeen evenwichtsmodel laten zien dat de vaste afvalstoffenheffing inderdaad een inefficiënt hoge afvalproductie veroorzaakt zoals uit de praktijk blijkt. In Hoofdstuk 4 wordt aandacht besteed aan een vergelijking tussen de invoering van een subsidie op recycling gecombineerd met een vaste afvalstoffenheffing, de invoering van een gedifferentieerd tarief voor afvalcollectie, de invoering van een subsidie op recycling gecombineerd met een gedifferentieerd tarief, en een afvalstofheffing op de prijs van het consumptiegoed.

Uit deze analyse blijkt dat een vaste afvalstofheffing een ernstig verstorende werking heeft op de afvalmarkt. Het invoeren van een impuls om recycling te verhogen door middel van een subsidie op recycling heeft geen noemenswaardig effect doordat consumenten geen prikkel krijgen om minder rest afval te produceren. Hoewel recycling goedkoper wordt door de subsidie is het produceren van rest afval gratis. Consumenten willen dan ook geen extra tijd in recycling steken als ze hier niet financieel voor beloont worden. Het invoeren van tariefdifferentiatie heeft een groter effect op het aanbod van afval en recycling. De hoeveelheid restafval dat wordt aangeboden neemt met 2% af. Er wordt circa 6% meer gerecycled. Indien zowel tariefdifferentiatie als een recycling subsidie wordt ingevoerd zijn de resultaten nog indrukwekkender, er wordt 70% minder restafval geproduceerd en 266% meer gerecycled. Bij deze percentages moet de kanttekening geplaatst worden dat in dit model geen technische bovengrens op recycling is gelegd. De afvalstoffenheffing op de prijs van een consumptiegoed vermindert de afvalproductie eveneens, maar in minder sterke mate. De afvalproductie wordt verlaagd met 0,5% indien er een belasting wordt geheven die precies de kosten van het verwerken dekt. De afvalstoffenheffing zal in tegenstelling tot een gedifferentieerd tarief alleen afvalpreventie stimuleren en geen recycling. Daarom is de reductie van afval zoveel kleiner voor deze beleidsmaatregel.

Gezien de kwantitatieve resultaten van dit model kan worden geconcludeerd dat de vaste heffing op afvalinzameling leidt tot afvalproductie die hoger is dan noodzakelijk. Het invoeren van tariefdifferentiatie is de meest effectieve methode om afvalproductie te verminderen, vooral als deze wordt gecombineerd met een recycling subsidie. Het invoeren van een afvalstoffenheffing is veel minder efficiënt. Hoewel de afvalstoffenheffing preventie van afval stimuleert, zal de afname in productie van restafval gering zijn. Ten eerste internaliseert de afvalstoffenheffing de kosten van afvalverwerking in de prijs van het consumptiegoed. De afvalstoffenheffing is te laag, ten opzichte van de prijs van het goed, om sterke schommelingen in de consumptie tot stand te brengen. Ten tweede stimuleert de afvalstoffenheffing geen recycling omdat de consumenten niet meer betalen voor het daadwerkelijke ophalen van afval, zij betalen immers al voor collectie en verwerking van afval als zij het product kopen.

In dit model is echter geen rekening gehouden met mogelijk ontduikgedrag van de consumenten. Consumenten kunnen op makkelijke manier van hun rest afval komen door het bij het GFT-afval te gooien. Bovendien is in dit model ook geen rekening gehouden met de in de vorige sectie genoemde link tussen de productie van afval en de verwerking van afval. Op de vraag hoe deze twee punten de effectiviteit van tariefdifferentiatie beïnvloeden wordt antwoord gegeven in de volgend twee onderzoeksvragen.

De problemen van afvalvervuiling

Onderzoeksvraag 3:

Hoe groot is het probleem van afvalvervuiling en hoe wordt afvalvervuiling beïnvloed door de preferenties van huishoudens?

De gemeente kan er voor kiezen om tariefdifferentiatie op alle afvalstromen of slechts op enkele afvalstromen in te voeren. In de praktijk zien we dat in de meeste gevallen alleen tariefdifferentiatie wordt ingevoerd op het ophalen van restafval en/of GFT-afval. Recyclebaar afval zoals glas en papier wordt gratis opgehaald. Of tariefdifferentiatie moet worden toegepast op GFT-afval is een lastige discussie. Indien de gemeente consumenten wil stimuleren afval beter te scheiden, dan lijkt het niet wenselijk om het ophalen van GFT-afval even duur te maken als het ophalen van restafval. De consumenten moeten tenslotte moeite doen om afval te scheiden en door middel van een prijsprikkel zouden zij hier toe worden gestimuleerd. Bovendien is de verwerking van GFT-afval een stuk goedkoper dan de verwerking van restafval en het is dan ook moeilijk te verkopen aan de burger dat het ophalen van GFT- en restafval even duur is. Hierdoor kan de gemeente veel goodwill verliezen, waardoor de consument zich minder of niet in zal spannen om het afval te scheiden en meer geneigd zal zijn zich op illegale wijze van het afval te ontdoen.

Afvalvervuiling is één van de mogelijke opties die consumenten hebben om zich van hun restafval te ontdoen. In dit geval gooien consumenten restafval bij de organische of recyclebare afvalstroom en vervuilen zo deze afvalstromen. Huishoudens hoeven nauwelijks bang te zijn dat ze worden gestraft voor dit onwenselijke gedrag. Het is bijvoorbeeld zeer kostbaar om de kwaliteit van het organisch afval te controleren terwijl het wordt opgehaald. De kwaliteit wordt in de meeste gevallen pas gecontroleerd als het afval bij de composteerinstallatie aankomt. Op dit punt is het moeilijk te achterhalen welke huishoudens de afvalstroom hebben vervuild. De enige optie die een gemeente heeft is een hele wijk aan te spreken op de kwaliteit van het door die wijk aangeleverde afval.

In dit proefschrift heb ik een algemeen evenwichtsmodel ontwikkeld waarmee de problemen rond de kwaliteit van afval kunnen worden bestudeerd. In Hoofdstuk 5 laat ik zien in hoeverre afvalvervuiling een belemmering kan zijn voor het introduceren van tariefdifferentiatie voor het ophalen van huishoudelijk afval. Ik heb mij in dit voorbeeld beperkt tot een analyse voor de introductie van tariefdifferentiatie op basis van gewicht, maar een vergelijkbare analyse geldt voor de introductie van tariefdifferentiatie op basis van ophaalfrequentie of op basis van volume. Het verschil zit hem namelijk in de methode waarop afvalproductie wordt berekend.

Uit de analyse in Hoofdstuk 5 blijkt dat consumenten afval gaan vervuilen indien tariefdifferentiatie wordt ingevoerd voor de inzameling van restafval. Gemiddeld produceren consumenten 10% minder rest afval na de introductie van tariefdifferentiatie. Deze afname bewerkstelligen ze door zowel meer lage kwaliteit organisch afval als meer hoge kwaliteit organisch afval te produceren. Hoge kwaliteit organisch afval bestaat uit 100% GFT-afval. Lage kwaliteit organisch afval bestaat uit 70% GFT-afval en 30% restafval (niet composteerbaar residu). Er wordt gemiddeld 46% meer lage kwaliteit organisch afval geproduceerd en 22% meer hoge kwaliteit organisch afval in tariefdifferentiatie wordt ingevoerd. Dit houdt in dat het percentage rest afval dat wordt weggegooid met het GFT-afval stijgt van 5,5% tot 6,7%.

Niet elk huishouden zal op een zelfde manier reageren op de invoering van tariefdifferentiatie. Afhankelijk van de preferenties van de consumenten zullen zij meer of minder geneigd zijn milieuonvriendelijk gedrag te vertonen. Een "traditionele" consument, die weinig interesse heeft voor het milieu zal sneller geneigd zijn om afval te vervuilen dan een "groene" consument, die bezorgd is om de stand van het milieu. Uit de modelresultaten in Hoofdstuk 5 blijkt, indien een gedifferentieerd tarief wordt ingevoerd, dat "groene" consumenten circa 14% minder restafval produceren. Ze zullen ongeveer 14% meer lage kwaliteit organisch afval produceren en 12% meer hoge kwaliteit organisch afval. Het percentage restafval dat met het GFT-afval wordt weggooit blijft gelijk, circa 3,1%. De "traditionele" consumenten produceren 8% minder restafval. Zij zullen ongeveer 63% meer lage kwaliteit organisch afval produceren en 41% meer hoge kwaliteit organisch afval aanbieden. Dit houdt in dat het percentage rest afval dat wordt weggegooid met het GFT-afval stijgt van 9,8% tot 11%.

Hoewel de gemiddelde stijging van restafval dat wordt weggegooid met GFT-afval niet zo groot lijkt, van 5,5% tot 6,7%, moet hier wel rekening worden met het feit dat dit een gemiddeld percentage is. Afval wordt in relatief kleine hoeveelheden verzameld, er past ongeveer 28 ton afval in een vuilniswagen. In sommige wijken zullen relatief veel "groene" consumenten wonen, in andere wijken relatief veel "traditionele consumenten". In een wijk met veel "traditionele" consumenten gaat de kwaliteit van het organisch afval danig achteruit. Dit kan betekenen dat afval opgehaald in deze wijken niet meer gecomposteerd kan worden. Zoals bleek uit het onderzoek van IPH (1995), wordt afval van zo een lage kwaliteit als geproduceerd door de "traditionele" consumenten over het algemeen geweigerd door composteerbedrijven.

Het invoeren van tariefdifferentiatie zal er dan ook in sommige wijken toe leiden dat organisch afval niet meer kan worden gecomposteerd. Vooral in grotere steden met een groter percentage "traditionele" consumenten zal het organisch afval sterk vervuild raken. Het is daarom gewenst dat gemeentes rekening houden met het probleem van afvalvervuiling indien zij overwegen tariefdifferentiatie in te voeren. Grote steden zouden kunnen overwegen alleen tariefdifferentiatie in te voeren in wijken met relatief veel "groene" consumenten. Dit soort selectief beleid wordt vaker gevoerd: bijvoorbeeld nu al wordt in grote steden, zoals Utrecht en Amsterdam slechts in een aantal wijken GFT-afval en restafval gescheiden opgehaald. In de andere wijken in deze steden was de kwaliteit van het GFT-afval te laag om te composteren en dus konden de extra inzamelingskosten niet worden gerechtvaardigd.

Een mogelijke oplossing voor het probleem van afvalvervuiling is het invoeren van een gecombineerd differentiatiesysteem voor zowel restafval als GFT-afval. Omdat consumenten hetzelfde betalen voor inzameling van restafval en van GFT-afval zullen zij geen prikkel meer hebben tot het vervuilen van organisch afval. Zoals in de inleiding van deze sectie al is opgemerkt kan het gevaar van zo een systeem zijn dat de goodwill van de consument wordt aangetast. Dit blijkt ook uit de kwantitatieve resultaten van Hoofdstuk 6. Indien een gedifferentieerd tarief op zowel GFT-afval als restafval wordt ingevoerd, gaan consumenten niet meer organisch afval produceren.

De “traditionele” consumenten gaan echter wel meer lage kwaliteit organisch afval produceren en minder hoge kwaliteit organisch afval. Slechts indien het mogelijk zou zijn om consumenten meer te laten betalen voor lage kwaliteit organisch afval dan zal een gedifferentieerd tarief op GFT-afval er voor zorgen dat er minder lage kwaliteit organisch afval wordt geproduceerd.

Keuze van de optimale afvalverwerkingsinstallatie

Onderzoeksvraag 4:

In hoeverre wordt de optimale locatie keuze van afvalverwerkingsinstallaties beïnvloed door de kwaliteit en kwantiteit van afval dat wordt aangeleverd en hoe zullen de ruimtelijke aspecten van afvalverwerking op hun beurt het aanbod van afval beïnvloeden?

In principe bepalen de gemeentes hoe het afval wordt verwerkt en op welke locatie. De gemeentes zijn echter niet geheel vrij in deze keuze. Afhankelijk van het afval dat wordt aangeleverd zullen zij kiezen hoe het wordt verwerkt. Het organisch afval dat zij ophalen zal altijd naar een composteerinstallatie moeten worden gebracht. Het restafval dat zij ophalen gaat of naar een verbrandingsinstallatie of naar een stortplaats.

Ook de kwantiteit van het afval is van invloed op de verwerkingsmethode. Door schaalvoordelen in de verwerking van afval, kunnen grotere hoeveelheden afval goedkoper verwerkt worden dan kleinere hoeveelheden. Vooral in geval van verbranden spelen schaalvoordelen een grote rol, mede vanwege de milieueisen die aan de installaties worden gesteld. Het is dan ook stukken goedkoper voor een gemeente om afval naar een grotere verbrandingsinstallatie te brengen, hoewel dit wel hogere transportkosten met zich meebrengt.

Indien tariefdifferentiatie wordt ingevoerd dan verandert de samenstelling van de afvalstromen opgehaald door de gemeentes. Gemeentes zullen minder restafval ophalen en meer organisch afval. Uit de kwantitatieve resultaten uit Hoofdstuk 6 blijkt dat indien alleen tariefdifferentiatie op restafval wordt ingevoerd, gemeentes circa 6% minder restafval ophalen. Indien een gedifferentieerd tarief op zowel restafval als GFT-afval wordt ingevoerd blijven de hoeveelheden organisch afval en restafval gelijk.

De verandering van de samenstelling van de afvalstromen kan grote gevolgen hebben voor de optimale locatiekeuze van afvalverwerkingsinstallaties. Indien er structureel meer GFT-afval wordt opgehaald is het voor een gemeente aantrekkelijk om er voor te kiezen het afval naar een grotere composteerinstallatie te brengen. Ook indien de kwaliteit van het organisch afval achteruit gaat kan het aantrekkelijk zijn het afval naar een grotere installatie te brengen omdat deze een lagere kwaliteit afval goedkoper kan verwerken.

Uit de analyse in Hoofdstuk 6 blijkt dat door de invoering van tariefdifferentiatie de kwaliteit van het organische afval in grote steden dusdanig achteruit gaat dat het voor deze gemeentes aantrekkelijker wordt om het afval naar een grote composteerinstallatie te brengen. Dit betekent dat de transportkosten aanzienlijk toenemen, met circa 60%. Hierdoor, zullen de totale kosten van afvalverwerking licht toe nemen bij de introductie van tariefdifferentiatie, van 239,6 miljoen Euro naar 240,7 miljoen Euro.

Aangezien gemeentes contracten hebben afgesloten met verbrandingsinstallaties, composteerinstallaties en stortplaatsen zal het niet mogelijk zijn om op korte termijn over te stappen van de ene installatie naar de andere. Dit wil niet zeggen dat het daarom niet noodzakelijk is om de consequenties van de veranderde afvalstroom mee te nemen in een analyse met betrekking tot de invoering van tariefdifferentiatie. Door het invoeren van tariefdifferentiatie, kunnen gemeentes verwachten dat de samenstelling van de door hun opgehaalde afval stroom voorgoed verandert. Op lange termijn zijn de gemeentes niet gebonden aan contracten met afvalverwerkers en kunnen zij wel degelijk overstappen naar andere installaties.

Uit de analyse in Hoofdstuk 6 blijkt dat de milieueffecten voor het invoeren van tariefdifferentiatie niet eenduidig zijn. Indien tariefdifferentiatie wordt ingevoerd wordt er minder restafval geproduceerd, dit betekent dat er minder restafval wordt verbrand of gestort, hetgeen ervoor zorgt dat zowel CO_2-emissies als CH_4-emissies omlaag gaan. Hoewel er meer organisch afval wordt geproduceerd, is een gedeelte hiervan van dusdanig slechte kwaliteit dat het zal moeten worden verbrand. Hierdoor zullen CO_2-emissies toch weer stijgen. Bovendien neemt het transport van afval toe waardoor NO_X-emissies ook toenemen. Netto blijkt dat de aan afvalverwerking en transport verbonden CO_2-emissies met 6,8% en NO_X-emissies met 17% toenemen. CH_4-emissies nemen met 4% af.

Het mag duidelijk zijn dat er een sterke wisselwerking is tussen kwaliteit van afval en de verwerking en de daarmee samenhangende kosten. Hoewel tariefdifferentiatie er voor zorgt dat de productie van restafval wordt verminderd, nemen de kosten van afvalverwerking niet af. Afvalvervuiling beïnvloedt de optimale verwerkingsmethode dusdanig dat gemeentes overschakelen van kleine composteerinstallaties naar grote composteerinstallaties, wat meer transportkosten met zich mee brengt. Wederom blijkt uit de kwantitatieve analyse in Hoofdstuk 6 dat het invoeren van tariefdifferentiatie alleen aantrekkelijk is in kleine gemeentes met relatief veel “groene” consumenten. De gevolgen van afvalvervuiling zijn voor grotere gemeentes te ingrijpend om de kosten van het invoeren van tariefdifferentiatie te kunnen verantwoorden.

Beleidsaanbevelingen

Onderzoeksvraag 5:

Welke beleidsverandering kan worden aanbevolen om de maatschappelijke kosten van afvalverwerking te minimaliseren?

In dit proefschrift zijn vier beleidsopties geanalyseerd, namelijk 1) een vaste afvalstoffenheffing, 2) een recycling subsidie, 3) een gedifferentieerd tarief voor het ophalen van restafval en 4) een gedifferentieerd tarief voor het ophalen van zowel restafval als GFT-afval.

Zoals al eerder genoemd, vragen de meeste gemeentes in Nederland een vaste afvalstoffenheffing voor het ophalen van afval, die alleen afhankelijk is van de gezinsgrootte. Indien een vaste heffing wordt geheven heeft de consument geen prikkel om meer afval te scheiden of the recyclen. De afvalproductie van consumenten is dan ook hoger dan sociaal wenselijk. Hier staat tegenover dat de gemeentes ook relatief lage kosten maken voor het inzamelen van afval omdat zij bij hoeven niet te houden hoeveel afval elk huishouden produceert. De kans is bovendien kleiner dat consumenten hun afval illegaal gaan dumpen of dat de organische en recyclebare afvalstroom wordt vervuild.

Om recycling te promoten kan de gemeente over gaan tot het subsidiëren van recycling. Gerecycled materiaal is over het algemeen duurder dan ruwe grondstoffen. Hierdoor is gerecycled materiaal moeilijk af te zetten. Door het recyclingproces te subsidiëren kan het gebruik van gerecycled materiaal worden gestimuleerd. Als de recycling subsidie echter gecombineerd is met een vaste afvalstoffenheffing, dan heeft het invoeren van een recycling subsidie weinig effect. Indien een vaste afvalstoffenheffing wordt gevraagd voor afvalinzameling heeft de consument geen prijsprikkel om minder restafval aan te bieden. Zij zullen dan, zonder additionele prikkels, ook niet meer gaan recyclen (een proces dat altijd kosten met zich mee brengt) ook niet als dit sociaal gezien zeer wenselijk zou zijn. Daarom is het af te raden om een recycling subsidie in te voeren in die gemeentes die een vaste afvalstoffenheffing vragen voor de inzameling van afval als er geen andere maatregelen worden genomen.

Indien zowel een recycling subsidie als tariefdifferentiatie wordt ingevoerd, heeft de recycling subsidie wel effect. Er wordt meer afval gerecycled en er worden minder ruwe grondstoffen gebruikt. Dit effect wordt echter ook bereikt door het invoeren van alleen tariefdifferentiatie, hoewel in mindere mate. Gemeentes kunnen dus ook volstaan met het invoeren van alleen tariefdifferentiatie en het gebruik van gerecycled materiaal ten opzichte van ruwe grondstoffen over laten aan de markt.

De invoering van tariefdifferentiatie zorgt er voor dat de productie van restafval afneemt. Door het invoeren van tariefdifferentiatie krijgt de consument een prijs prikkel om afval te verminderen. Vooral als er tariefdifferentiatie op basis van het gewicht van afval wordt ingevoerd zal de consument veel minder restafval produceren. Dit betekent dat er behoorlijk bespaard wordt op de kosten van verbranden en storten van afval.

Gemeente kunnen kiezen of zij tariefdifferentiatie op restafval invoeren of tariefdifferentiatie op zowel restafval als GFT-afval. Indien zij moeten betalen voor het ophalen van zowel restafval en GFT-afval zullen ze geen prikkel hebben om GFT-afval en restafval actief te scheiden. Het invoeren van tariefdifferentiatie op zowel restafval als GFT-afval kan ook op veel weerstand bij de burger stuiten. Om restafval en GFT-afval te scheiden moeten consumenten veel moeite doen. Indien zij hier niet (financieel) voor worden beloond kunnen zij geneigd zijn gedeeltelijk of volledig op te houden met afval scheiding. Indien zij hetzelfde bedrag moeten betalen voor het ophalen van GFT-afval en restafval zal de weerstand nog groter zijn. GFT-afval is veel goedkoper om te verwerken dan restafval en de marginale kosten van het produceren van organisch afval zijn daardoor lager dan de marginale kosten van het produceren van restafval. Door een gelijke prijs te vragen voor het inzamelen van beiden soorten afval worden consumenten als het ware financieel gestraft voor het produceren van GFT-afval in plaats van restafval.

Tariefdifferentiatie op alleen het ophalen van restafval stimuleert consumenten om zo min mogelijk restafval te produceren. Dit zullen zij doen door zoveel mogelijk afval te scheiden. Ook de productie van GFT-afval zal in dit scenario toe nemen omdat de consument beloond wordt voor het scheiden van GFT-afval in de vorm van een lagere afvalstoffenheffing. Consumenten zullen minder GFT-afval weggooien met het restafval. Een nadeel van deze vorm van tariefdifferentiatie is dat het consumenten een stimulans geeft om het composteerbaar afval en het recyclebaar afval te vervuilen. Indien organisch afval te vervuild is met restafval, kan het niet meer gecomposteerd worden. Het zal in dat geval moeten worden verbrand, wat tot veel hogere verwerkingskosten leidt.

Samenvattend is de vraag welke beleidsverandering kan worden aanbevolen niet eenduidig te beantwoorden. Het invoeren van een recycling subsidie is niet aan te raden, omdat de huishoudelijke afvalmarkt is verstoord door een vaste afvalstoffenheffing en daarom een recycling subsidie geen effect zal hebben op het recyclinggedrag van de consumenten. Indien de overheid recycling wil stimuleren is het effectiever om voor tariefdifferentiatie te kiezen. Tariefdifferentiatie op de inzameling van restafval geeft de grootste stimulans voor het scheiden van afval, maar ook de grootste kans op afvalvervuiling. In kleinere gemeentes is het invoeren van beide soorten van tariefdifferentiatie mogelijk. In kleinere gemeentes met relatief veel "groene" huishoudens is tariefdifferentiatie op restafval goed toe te passen. Middels een numerieke analyse in Hoofdstuk 6 laat ik zien dat het gevaar van afvalvervuiling minder groot is in deze gemeentes en dat daardoor in deze gemeentes de baten opwegen tegen de kosten. Gemeentes met relatief veel traditionele consumenten moeten bedacht zijn op de problemen rond afvalvervuiling. In deze gemeentes zal tariefdifferentiatie voor het ophalen van zowel restafval als GFT afval beter werken. Ook in grotere gemeentes is het invoeren van tariefdifferentiatie voor zowel restafval als GFT-afval mogelijk. Door afvalvervuiling zal het invoeren van tariefdifferentiatie op alleen restafval hier minder aantrekkelijk zijn.

Indien de problemen rond afvalvervuiling kunnen worden opgelost is het mogelijk om tariefdifferentiatie in alle typen gemeentes in te voeren zodat de productie van restafval afneemt en de recyclingpercentages toenemen. Afvalvervuiling van recyclebaar materiaal, zoals glas en papier, is kostbaar maar niet zo problematisch aangezien deze afvalstromen gezuiverd kunnen worden. De vervuiling van organisch afval is veel problematischer omdat dit betekent dat het afval niet meer kan worden gecomposteerd.

Er zijn wel technieken beschikbaar om vervuild organisch afval te zuiveren, denk bijvoorbeeld aan de technieken die gebruikt worden in de VAGRAM installatie in Groningen, maar de kwaliteit van het compost dat is geproduceerd van vervuild organisch afval is niet goed genoeg om te worden afgezet. Het is belangrijk dat vanuit de overheid een stimulans komt om betere scheidingstechnieken te ontwikkelen aangezien de introductie van tariefdifferentiatie een belangrijk middel is om de productie van huishoudelijk afval te verminderen en recyclingpercentages te vergroten.

Modelleren van het afvalvraagstuk

In Hoofdstuk 1 zijn in aanvulling op de onderzoeksvragen de volgende drie modelleer vragen geformuleerd:

- Hoe kunnen de interacties tussen de afvalsector, overheidsbeleid en de rest van de economie gemodelleerd worden?
- Hoe kan een vaste afvalstoffenheffing geïntroduceerd worden in een algemeen evenwichtsmodel?
- Hoe kunnen de ruimtelijke aspecten van het afval probleem, zoals een exogene set van locaties van afvalverwerkingsinstallaties, schaalvoordelen en transport kosten geïntroduceerd worden in een algemeen evenwichtsmodel?

In dit proefschrift heb ik laten zien hoe de verschillende aspecten van het afvalvraagstuk, zoals geformuleerd in de modelleervragen, in een algemeen evenwichtssetting kunnen worden geïmplementeerd. Vooral de vaste afvalstoffenheffing is niet eenvoudig om op te nemen in een algemeen evenwichtsmodel. Door een subsidie-cum-belasting systeem te gebruiken kan het probleem van een marginale prijs van nul omzeilt worden. In dit systeem betalen consumenten de evenwichtsprijs voor de inzameling van afval. Zij worden hiervoor gecompenseerd door de overheid door middel van een subsidie die de prijs van afvalcollectie exact dekt. Hierdoor is de waargenomen prijs van afvalinzameling gelijk geworden aan nul. De consument betaalt vervolgens een directe belasting voor het inzamelen van afval aan de overheid, de zogenaamde afvalstoffenheffing. Samenvattend, in dit systeem betalen de consumenten dus wel voor afvalinzameling maar het bedrag dat zij betalen is niet rechtstreeks gekoppeld aan de hoeveelheid afval dat zij produceren.

In de Hoofdstukken 4, 5 en 6 zijn drie verschillende algemeen evenwichtsmodellen ontwikkeld. Al deze modellen zijn gebouwd in het Negishi format. De modellen variëren van betrekkelijk simpel (Hoofdstuk 4) tot een gedetailleerde complexe analyse van het afvalprobleem (Hoofdstuk 6).

Samenvattend heeft het meest complexe model ontwikkeld in Hoofdstuk 6 de volgende kenmerken:

- De economie is verdeeld in vier verschillende gemeentes. Per gemeente worden er twee typen consumenten onderscheiden: een "groene" en een "traditionele" consument. De "groene" consument gedraagt zich milieuvriendelijker dan de traditionele consument. De gemeentes verschillen in het aantal "groene" en "traditionele" consumenten dat er wonen.

- Gemeentes kunnen afval laten verwerken in een composteerinstallatie, een verbrandingsinstallatie en een stortplaats. Per afvalverwerkingsinstallatie worden drie groottes onderscheiden: een kleine, een middelgrote en een grote installatie. Verwerkingskosten worden beïnvloed door schaalvoordelen; een grote installatie verwerkt afval goedkoper dan een kleine installatie. Naast verwerkingskosten zullen gemeentes ook transportkosten moeten betalen. Aangezien grotere installaties gemiddeld verder van de gemeentes verwijdert zijn, zijn ook de transportkosten hoger voor deze installaties.

- Beleidsmaatregelen zoals emissiereducties, vaste afvalstoffenheffingen, en tariefdifferentiatie kunnen zonder problemen worden ingevoerd in het model.

De modellen zoals gepresenteerd in dit proefschrift maken het mogelijk om een gedetailleerde analyse te maken over het invoeren van tariefdifferentiatie. Nieuw in de analyse is de modellering van de kwaliteit van afval en de daarmee samenhangende mogelijkheid van afvalvervuiling, de link tussen productie van huishoudelijk afval, collectie van afval en verwerking van afval door afvalverwerkingsinstallaties en tot slot het modelleren van ruimtelijke aspecten van afvalverwerking in een algemeen evenwichtssetting.

In tegenstelling tot de bestaande literatuur, is in dit proefschrift een link gemaakt tussen de productie van afval en de verwerking van afval. Hierdoor is het mogelijk om te analyseren of het kosteneffectief is om tariefdifferentiatie in te voeren. In dit proefschrift is aangetoond dat de afname van de kwaliteit van afval door de invoering van tariefdifferentiatie in belangrijke mate de verwerkingskosten zullen bepalen en daarmee de kosteneffectiviteit van tariefdifferentiatie.

Het bouwen van een algemeen evenwichtsmodel heeft het mogelijk gemaakt om deze relatie concreet te analyseren. Een algemeen evenwichtsmodel beschrijft alle relevante markten in de economie, berekent de interacties tussen de verschillende markten en vormt een gesloten systeem. Uit de modellen blijkt dat het succes van tariefdifferentiatie in belangrijke mate afhangt van de preferenties van de consument. In een wijk waar relatief veel traditionele consumenten wonen, zal tariefdifferentiatie geen doorslaand succes zijn.

Door zowel de consumptie als de productiesector in het model op te nemen, toont het model aan dat in tegenstelling tot de verwachtingen, tariefdifferentiatie niet geschikt is voor het stimuleren van preventie. De prijsprikkel vanuit de afvalsector is te gering om een significante verandering van het consumptiepatroon en dus de productiesector tot stand te brengen. Indien de overheid preventie wil stimuleren zal zij aan andere beleidsmaatregelen moeten denken.

Tot slot nog een opmerking over het toevoegen van ruimtelijke aspecten van het afvalprobleem. Zoals aangetoond in dit proefschrift, spelen zowel preferenties van de consumenten, locatie en schaalvoordelen van afvalverwerkingsinstallaties een belangrijke rol bij het bepalen van het optimale afvalbeleid. Dit betekent dat een nationaal afvalbeleidsplan voor huishoudelijk afval niet succesvol kan zijn indien er niet specifiek rekening is gehouden met de locale kenmerken van gemeentes en consumenten. Afhankelijk van de samenstelling van de gemeente of, op nog gedetailleerder niveau, de samenstelling van een wijk, zal het wel of niet mogelijk zijn tariefdifferentiatie in te voeren. Hierdoor is het noodzakelijk dat een analyse over het optimale afvalbeleid zich afspeelt op gemeenteniveau.

Algemene conclusies

Samenvattend heeft dit onderzoek een aantal nieuwe inzichten met betrekking tot tariefdifferentiatie opgeleverd.

1) Door het opnemen van de mogelijkheid van afvalvervuiling, kan worden gedemonstreerd dat tariefdifferentiatie in sommige gemeentes wel zal werken en in

andere gemeentes niet. Afvalvervuiling zal altijd voorkomen, maar in gemeentes met relatief veel “groene” consumenten levert afvalvervuiling minder kosten op en zal daardoor de kostenefficiëntie van tariefdifferentiatie groter zijn. Het is dan ook belangrijk dat voor elke gemeente een analyse wordt gemaakt of tariefdifferentiatie moet worden toegepast. Een nationaal afvalbeleidsplan voor huishoudelijk afval dat geen rekening houdt met de specifieke karakteristieken van gemeentes en huishoudens zal daarom suboptimaal zijn.

2) Tariefdifferentiatie is niet effectief gebleken om preventie te bevorderen. De prijsprikkel vanuit de afvalsector is niet groot genoeg om consumenten te bewegen hun consumptiepatroon aan te passen. Tariefdifferentiatie is wel geschikt om de marktverstoring veroorzaakt door de vaste afvalstoffenheffing op te heffen. Indien tariefdifferentiatie wordt ingevoerd zullen andere maatregelen, zoals een recyclingsubsidie, wel rechtstreeks effect hebben op het percentage afval dat wordt gerecycled.

3) Door zowel de consumptie en productiesectoren als verschillende afvalverwerkings-opties op te nemen kan worden geanalyseerd hoe kwaliteit en kwantiteit van afval de verwerkingsmarkt van afval beïnvloeden en zo de kosteneffectiviteit van de beleidsmaatregel beïnvloeden. Uit dit proefschrift blijk dat de invoering van tariefdifferentiatie lang niet zo voordelig is voor het milieu. Door tariefdifferentiatie gaat dan wel de productie van rest afval omlaag maar doordat niet alleen meer afval gescheiden wordt maar ook GFT-afval wordt vervuilt gaan verwerkingskosten en transportkosten en de daarbij horende emissies omhoog.

Onderzoeksaanbevelingen

In dit proefschrift is een methode ontwikkeld om de effecten van afvalvervuiling te analyseren. Met het model kan worden berekend wat tariefdifferentiatie voor effect heeft op de kwaliteit van GFT-afval en in hoeverre de kosten van afvalverwerking beïnvloed worden door de invoering van tariefdifferentiatie. In het model worden milieukosten, twee typen consumenten, vier gemeentes, transport kosten, schaalvoordelen en gedifferentieerde tarieven voor het verwerken van hoge en lage kwaliteit organisch afval meegenomen. Het model is toegepast in een gestileerd voorbeeld met numerieke data gebaseerd op de Randstad in 2000.

Het meest uitgebreide model zoals beschreven in Hoofdstuk 6 neemt al een groot aantal aspecten van de afvalmarkt mee in de analyse maar er zouden nog een aantal punten uitgebreid kunnen worden om een gedetailleerdere voorspelling van de effecten van tariefdifferentiatie te geven.

Ten eerste zou het model uitgebreid kunnen worden met meerdere afvalstromen. In dit proefschrift is alleen naar organisch afval en restafval gekeken. Andere recyclebare afvalstromen, zoals glas, papier en blik, zouden ook in de analyse moeten worden betrokken.

Ten tweede zou het interessant zijn om zowel thuiscomposteren als illegale dumping in het model op te nemen. De eerste optie geeft de consument een legale mogelijkheid om haar GFT-afval productie te verminderen, thuiscompostering op grote schaal kan echter wel problemen veroorzaken voor de composteringsindustrie. De tweede optie geeft de consumenten een (weliswaar illegale) optie om van restafval af te komen. Deze optie brengt natuurlijk maatschappelijke kosten met zich mee en kan een belangrijke belemmering vormen bij het invoeren van tariefdifferentiatie.

Ten derde zou het interessant zijn om te onderzoeken in hoeverre preventie een rol kan spelen in het oplossen van het afvalprobleem. In het model, ontwikkeld in dit proefschrift, wordt rekening gehouden met preventie maar dit is op een enigszins simplistische wijze in het model geïmplementeerd. In dit proefschrift wordt preventie gesimuleerd door de introductie van twee goederen, een afval intensief en een afval extensief goed. Consumenten kunnen hun afvalproductie beïnvloeden door te substitueren tussen de twee goederen. In de praktijk zal de consument voornamelijk afval voorkomen door het kiezen van producten met minder verpakkingsmateriaal. Het zou daarom interessant zijn om een verpakkingsgraad op te nemen voor een product. Hoe hoger de verpakkingsgraad, hoe meer afval wordt geproduceerd.

Ten vierde zou het de moeite waard zijn om de aanname van perfectie concurrentie tussen afvalverwerkingsinstallaties los te laten. Doordat er contracten bestaan tussen gemeentes en afvalverwerkingsinstallaties is het niet realistisch om aan te nemen dat verwerkingsinstallaties concurreren met elkaar, in tegenstelling deze installaties gedragen zich veel meer als monopolisten dan als perfecte concurrenten.

Tenslotte zou er onderzoek moeten worden gedaan om de daadwerkelijke kosten van afvalvervuiling te schatten. In dit onderzoek is het onmogelijk gebleken bestaande informatie te vinden over de daadwerkelijke compositie van de organische afvalstroom en de toegenomen kosten door vervuild organisch afval. Hierdoor werd het onmogelijk om een aantal parameterwaarden in het model te funderen op reële data. Aangezien tariefdifferentiatie al in verscheidende gemeentes in Nederland is ingevoerd, zou het interessant zijn om in deze gemeentes gegevens te verzamelen over de mate van afvalvervuiling in de praktijk.

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Appendix I: Specification of the model in GAMS

This appendix presents the specification of the model presented in Chapter 6. The model is written for the GAMS-program (General Algebraic Modeling System), which is developed for solving large mathematical optimization models (Brooke et al., 1998).

Most of the data is read from external files, these are not included in this appendix. Complete versions of the models, including data files, report writing files and sub-models for all scenarios are available on request.

GAMS-specification of the model presented in chapter 6 scenario 5a and 5b

```
* Define files for storing results
  FILE RESULTS         /IO_TABLE_S5A_5B.RES/
  FILE RES_PERCENTAGE  /PERCENTAGE_S5A_5B.XLS/
  FILE ABS_RES         /RESULTS_S5A_5B.XLS/
  FILE PRICE_RES       /PRICE_S5A_5B.XLS/
  FILE RES_EMIS        /EMISSION_S5A_5B.XLS/

* Definition of sets in the model
SETS
K          commodities   /GOOD_WE  "consumption good waste extensive"
                          GOOD_WI  "consumption good waste intensive"
                          CS_G     "collection services rest waste"
                          CS_C     "collection services org. waste"
                          TRANS    "transport"
                          COMP_L   "composting services low quality"
                          COMP_H   "composting services high quality"
                          INCIN    "incineration services"
                          LAND     "landfilling services"
                          CAPITAL  "capital"
                          LABOUR   "labor"
                          TAX      "tax on landfilling"
                          CO2      "co2 emission rights"
                          NOx      "nox emission right"
                          CH4      "methane emission rights"/
G(K)       goods         /GOOD_WE, GOOD_WI, CS_G, CS_C, TRANS
                          COMP_L, COMP_H, INCIN, LAND/
COL(G)     collection    /CS_G, CS_C/
G1(G)      subset goods  /GOOD_WE, GOOD_WI, TRANS/
G2(G)      subset goods  /GOOD_WE, GOOD_WI/
E(K)       emission      /CO2,NOX,CH4/
PF(K)      prod. factors        /CAPITAL, LABOUR/
J          municipality         /M1       "municipality 1"
                                 M2       "municipality 2"
                                 M3       "municipality 3"
                                 M4       "municipality 4"/
TWD(G)     waste treatment      /COMP_L, COMP_H, INCIN, LAND/
```

```
TCOMP(TWD) composting           /COMP_L, COMP_H/
S          size                 /SMALL    "small unit"
                                 MIDDLE   "middle unit 1"
                                 B        "big unit"/
I          consumers            /CONS1    "consumer 1"
                                 CONS2    "consumer 2"
                                 CONS1A   "virtual consumer 1"
                                 CONS2A   "virtual consumer 2"
                                 GOV      "government consumer"/
C(I)       consumers subset     /CONS1, CONS2, CONS1A, CONS2A/
I2(I)      consumer subset      /CONS1, CONS2, GOV/
I3(I)      consumer subset      /CONS1A, CONS2A/
I4(I2)     consumer subset      /CONS1,CONS2/
F          quality org. waste   /LOW, HIGH/
ITWEL      iterations           /1*150/
SCEN       scenario             /FLAT_FEE, VAR_FEE/
FLAT(SCEN) sub set scenario     /FLAT_FEE/;

ALIAS(F,F1);
ALIAS(I,I1);
ALIAS(S,S1);
ALIAS(J,J1);
ALIAS(COL,COL1);
ALIAS(TWD,TWD1);
ALIAS(C,C1);
ALIAS(G2,G2A);

* Import IO benchmark data from excel
TABLE BENCHMARK(*,*)
  $ONDELIM
  $INCLUDE "DATA_MUNICIPALITIES_TOTAL_S10_FINAL3A.CSV"
  $OFFDELIM;
TABLE MUNICIPALITY(*,*,*)
  $ONDELIM
  $INCLUDE "DATA_MUNICIPALITIES_S10_FINAL3A.CSV"
  $OFFDELIM;
TABLE TRANSPORT(*,*,*)
  $ONDELIM
  $INCLUDE "TRANSPORT_COMP.csv"
  $OFFDELIM;

* Table substitution elasticities consumers
TABLE ELASTICITY_CONS(*,*,*)
             SIGMA_C    SIGMA_F    SIGMA_U
CONS1.M1         0.6        0.9      4.5
CONS1.M2         0.6        0.9      4.5
CONS1.M3         0.6        0.9      4.5
CONS1.M4         0.6        0.9      4.5
CONS2.M1         0.3        0.1      4.5
CONS2.M2         0.3        0.1      4.5
CONS2.M3         0.3        0.1      4.5
CONS2.M4         0.3        0.1      4.5
GOV.M1           0.5        0.1      4.5
GOV.M2           0.5        0.1      4.5
GOV.M3           0.5        0.1      4.5
GOV.M4           0.5        0.1      4.5;

* Table substitution elasticities production sectors
TABLE ELASTICITY_PROD(*,*)
       GOOD_WE  GOOD_WI  CS_G CS_C  TRANS COMP_L COMP_H INCIN   LAND
```

```
SIGMA_P    0.8      0.8   0.8    0.8   0.8    0.8   0.8   0.8    0.8
SIGMA_P2   0.5      0.5     0      0   0.5    0.5   0.5   0.5    0.5
SIGMA_P3     0        0     6      0     0      0     0     0      0;

* Table technology parameter production sectors
TABLE TECHNOLOGY(*,*)
      GOOD_WE  GOOD_WI  CS_G   CS_C TRANS COMP_L COMP_H INCIN  LAND
A         1.0      1.0   1.0    1.0   1.0    1.0    1.0   1.0   1.0;

* Table technology parameter waste treatment units
TABLE TECHNOLOGY_WD(*,*,*)
            SMALL   MIDDLE       B
A_WD.COMP_L  0.17      0.3     1.1
A_WD.COMP_H  1.0       1.1     1.3
A_WD.LAND    1.0       1.1     1.3
A_WD.INCIN   1.0       1.3     1.7;

* Table labor costs generating organic waste
TABLE LABOUR_COMP(*,*)
                         LOW               HIGH
MHU                     13.8               11.8;

* Table share green and traditional consumers per municipality
TABLE SHARE_CONSUMERS(*,*)
               CONS1    CONS2
M1              0.72     0.28
M2              0.68     0.32
M3              0.65     0.35
M4              0.50     0.50;

* Table waste percentage per consumption good
TABLE SHARE_WASTE(*,*)
                GOOD_WE  GOOD_WI
SHARE_W            1.2       0.8;

* Definition benchmark parameters in the model
PARAMETERS
BETA(I4,J,G2)        percentage waste in good
Q_BAR(G)             benchmark production
Q_WD_BAR(TWD,S)      benchmark production waste disposal
Q_CSG_BAR(J)         benchmark production collection rest waste
Q_CSC_BAR(J,F)       benchmark production collection organic waste
X_BAR(I,J,G)         benchmark consumption
XCSC_BAR(I,J,F)      benchmark generation compost waste quality f
TWASTEG_BAR(J)       benchmark total generation rest waste
TWASTEC_BAR(J)       benchmark total generation organic waste
LAB_BAR(G)           benchmark labor use
CAP_BAR(G)           benchmark capital use
CO2_ER_BAR(G)        benchmark CO2 emission rights use
NOX_ER_BAR(G)        benchmark NOX emission rights use
CH4_ER_BAR(G)        benchmark CH4 emission rights use
LAB_WD_BAR(TWD,S)    benchmark labor use waste treatment
CAP_WD_BAR(TWD,S)    benchmark capital use waste treatment
CO2_ER_WD_BAR(TWD,S) benchmark CO2 emission waste treatment
NOX_ER_WD_BAR(TWD,S) benchmark NOX emission waste treatment
CH4_ER_WD_BAR(TWD,S) benchmark CH4 emission waste treatment
LAB_CSG_BAR          benchmark labor use collection rest waste
CAP_CSG_BAR          benchmark capital use collection rest waste
CO2_ER_CSG_BAR       benchmark CO2 emission collection rest waste
NOX_ER_CSG_BAR       benchmark NOX emission collection rest waste
```

```
CH4_ER_CSG_BAR       benchmark CH4 emission collection rest waste
LAB_CSC_BAR          benchmark labor use collection organic waste
CAP_CSC_BAR          benchmark capital use collection organic waste
CO2_ER_CSC_BAR       benchmark CO2 emission collection org. waste
NOX_ER_CSC_BAR       benchmark NOX emission collection org. waste
CH4_ER_CSC_BAR       benchmark CH4 emission collection org. waste
TAX_BAR(S)           benchmark tax landfilling
FEE(I,J)             benchmark fee
SUB(I,J)             benchmark subsidy
P_BAR(K)             benchmark price
LAB_C_BAR(I,J,F)     benchmark labor use composting
ENDL_BAR(I,J)        benchmark labor supply
ENDK_BAR(I,J)        benchmark capital supply
ENDCO2_BAR(J)        benchmark CO2 emission rights supply
ENDNOX_BAR(J)        benchmark NOX emission rights supply
ENDCH4_BAR(J)        benchmark CH4 emission rights supply
ENDTAX(J)            benchmark tax
MHU(F)               benchmark labor cost organic waste quality f
WASTE_BAR(I,J)       benchmark waste production
U_BAR(I,J)           benchmark utility
WTS_BAR(J,TWD,S)     benchmark use waste treatment services
WTS_COMP_BAR(S)      benchmark use waste treatment services
LUMPSUM_BAR(I,J)     benchmark lump sum transfer
Z(I,J)               share consumers
Z_W(G2)              share waste;
```

*** Initialization benchmark parameters**

```
Q_BAR(G)=BENCHMARK(G,G);
Q_WD_BAR(TWD,"SMALL")=BENCHMARK(TWD,TWD);
Q_CSG_BAR(J)=MUNICIPALITY("CS_G",J,"CS_G");
X_BAR(I,J,G)=-MUNICIPALITY(G,J,I);
X_BAR("GOV",J,G)=-MUNICIPALITY(G,J,"GOV");
XCSC_BAR(I3,J,"LOW")=-0.3*MUNICIPALITY("CS_C",J,"CONS1")$(ORD(I3)=1)
                     -0.1*MUNICIPALITY("CS_C",J,"CONS2")$(ORD(I3)=2);
XCSC_BAR(I3,J,"HIGH")=-0.7*MUNICIPALITY("CS_C",J,"CONS1")$(ORD(I3)=1)
                    -0.9*MUNICIPALITY("CS_C",J,"CONS2")$(ORD(I3)=2);
Q_CSC_BAR(J,"LOW")=SUM(I3,XCSC_BAR(I3,J,"LOW"));
Q_CSC_BAR(J,"HIGH")=SUM(I3,XCSC_BAR(I3,J,"HIGH"));
TWASTEG_BAR(J)=SUM(I,X_BAR(I,J,"CS_G"));
TWASTEC_BAR(J)=SUM(I,X_BAR(I,J,"CS_C"));
WASTE_BAR(I,J)=X_BAR(I,J,"CS_G")+ X_BAR(I,J,"CS_C");
LAB_BAR(G1)=-BENCHMARK("LABOUR",G1);
LAB_WD_BAR(TWD,S)=-BENCHMARK("LABOUR",TWD);
CAP_WD_BAR(TWD,S)=-BENCHMARK("CAPITAL",TWD);
CO2_ER_WD_BAR(TWD,S)=-BENCHMARK("CO2",TWD);
NOX_ER_WD_BAR(TWD,S)=-BENCHMARK("NOX",TWD);
CH4_ER_WD_BAR(TWD,S)=-BENCHMARK("CH4",TWD);
LAB_CSG_BAR(J)=-MUNICIPALITY("LABOUR",J,"CS_G");
CAP_CSG_BAR(J)=-MUNICIPALITY("CAPITAL",J,"CS_G");
LAB_CSC_BAR(J,"LOW")=-SUM(I3,XCSC_BAR(I3,J,"LOW"))
                     /SUM((I3,F),XCSC_BAR(I3,J,F))
                     *MUNICIPALITY("LABOUR",J,"CS_C");
CAP_CSC_BAR(J,"LOW")=-SUM(I3,XCSC_BAR(I3,J,"LOW"))
                     /SUM((I3,F),XCSC_BAR(I3,J,F))
                     *MUNICIPALITY("CAPITAL",J,"CS_C");
LAB_CSC_BAR(J,"HIGH")=-SUM(I3,XCSC_BAR(I3,J,"HIGH"))
                     /SUM((I3,F),XCSC_BAR(I3,J,F))
                     *MUNICIPALITY("LABOUR",J,"CS_C");
CAP_CSC_BAR(J,"HIGH")=-SUM(I3,XCSC_BAR(I3,J,"HIGH"))
                      /SUM((I3,F),XCSC_BAR(I3,J,F))
```

```
                    *MUNICIPALITY("CAPITAL",J,"CS_C");
CAP_BAR(G1)=-BENCHMARK("CAPITAL",G1);
CO2_ER_BAR(G1)=-BENCHMARK("CO2",G1);
NOX_ER_BAR(G1)=-BENCHMARK("NOX",G1);
CH4_ER_BAR(G1)=-BENCHMARK("CH4",G1);
TAX_BAR(S)=-BENCHMARK("TAX","CS_G");
FEE(I,J)=-MUNICIPALITY("FEE",J,I);
SUB(I,J)=ABS(MUNICIPALITY("SUBSIDY",J,I));
P_BAR(K)=BENCHMARK(K,"PRICE");
MHU(F)=LABOUR_COMP("MHU",F);
LAB_C_BAR(I3,J,F)=XCSC_BAR(I3,J,F)/MHU(F);
ENDL_BAR(I4,J)=MUNICIPALITY("LABOUR",J,I4)
            +SUM(F,LAB_C_BAR("CONS1A",J,F))$(ORD(I4)=1)
            +SUM(F,LAB_C_BAR("CONS2A",J,F))$(ORD(I4)=2);
ENDK_BAR(I,J)=MUNICIPALITY("CAPITAL",J,I);
ENDCO2_BAR(J)=MUNICIPALITY("CO2",J,"GOV");
ENDNOX_BAR(J)=MUNICIPALITY("NOX",J,"GOV");
ENDCH4_BAR(J)=MUNICIPALITY("CH4",J,"GOV");
ENDTAX(J)=MUNICIPALITY("TAX",J,"GOV");
U_BAR(I3,J)=SUM(F,XCSC_BAR(I3,J,F));
WTS_BAR(J,TWD,"SMALL")=-MUNICIPALITY(TWD,J,"CS_G");
WTS_BAR(J,"COMP_L","SMALL")=-MUNICIPALITY("COMP_L",J,"CS_C");
WTS_BAR(J,"COMP_H","SMALL")=-MUNICIPALITY("COMP_H",J,"CS_C");
WTS_COMP_BAR(S)=-SUM(J,MUNICIPALITY("INCIN",J,"COMP_L"));
*LUMPSUM_BAR(C,J)=CONSUMER("LUMPSUM",C,J);
LUMPSUM_BAR(I,J)=ABS(MUNICIPALITY("LUMPSUM",J,I));
Z(I,J)=SHARE_CONSUMERS(J,I);
Z_W(G2)=SHARE_WASTE("SHARE_W",G2);
BETA(I4,J,G2)=Z_W(G2)*WASTE_BAR(I4,J)/SUM(G2A,X_BAR(I4,J,G2A));

* Definition parameters substitution elasticity
PARAMETERS
RHO(G)           subs. parameter labor versus capital and emission
RHO2(G)          subs. parameter capital versus emission
RHO3             subs. parameter incineration versus landfilling
RHO_C(I,J)       subs. parameter rest waste versus organic waste
RHO_F(I,J)       subs. parameter low versus high quality org. waste
RHO_U(I,J)       subs. parameter consumption costs
SIGMA_BAR(G)     subs. elas. labor versus capital emission rights
SIGMA2_BAR(G)    subs. elas. capital versus emission rights
SIGMA3_BAR       subs. elas. incineration versus landfilling
SIGMA_C_BAR(I,J) subs. elas. rest versus org. waste
SIGMA_F_BAR(I,J) subs. elas. low versus high quality org. waste
SIGMA_U_BAR(I,J) subs. elas. low versus high quality org. waste;

* Initialization parameters substitution elasticity utility
SIGMA_BAR(G)=ELASTICITY_PROD("SIGMA_P",G);
SIGMA2_BAR(G)=ELASTICITY_PROD("SIGMA_P2",G);
SIGMA3_BAR=ELASTICITY_PROD("SIGMA_P3","CS_G");
SIGMA_C_BAR(I4,J)=ELASTICITY_CONS(I4,J,"SIGMA_C");
SIGMA_F_BAR(I3,J)=ELASTICITY_CONS("CONS1",J,"SIGMA_F")$(ORD(I3)=1)
                  +ELASTICITY_CONS("CONS2",J,"SIGMA_F")$(ORD(I3)=2);
SIGMA_U_BAR(I2,J)=ELASTICITY_CONS(I2,J,"SIGMA_U");
RHO(G)=(1-SIGMA_BAR(G))/SIGMA_BAR(G);
RHO2(G1)=(1-SIGMA2_BAR(G1))/SIGMA2_BAR(G1);
RHO2(TWD)=(1-SIGMA2_BAR(TWD))/SIGMA2_BAR(TWD);
RHO3=(1-SIGMA3_BAR)/SIGMA3_BAR;
RHO_C(I4,J)=(1-SIGMA_C_BAR(I4,J))/SIGMA_C_BAR(I4,J);
RHO_F(I3,J)=(1-SIGMA_F_BAR(I3,J))/SIGMA_F_BAR(I3,J);
RHO_F(I4,J)=RHO_F("CONS1A",J)$(ORD(I4)=1)+RHO_F("CONS2A",J)
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```
            $(ORD(I4)=2);
RHO_U(I2,J)=(1-SIGMA_U_BAR(I2,J))/SIGMA_U_BAR(I2,J);
U_BAR(I2,J)=SUM(G2,0.5*X_bar(I2,J,G2)**(-RHO_U(I2,J)))
            **(-1/RHO_U(I2,J));

* Definition parameters necessary for model
PARAMETERS
THETA(*,K)                value share of factor input
THETA_U(I,J,G)            value share consumption goods
THETA_C(I,J,G)            value share of waste
THETA_F(I,J,F)            value share quality organic waste
THETA_CS(J,S)             value share waste treatment options
THETA_CSG(K,J)            value share collection rest waste
THETA_CSC(K,J,F)          value share collection organic waste
THETA_WD(*,TWD,S)         value share waste treatment options
MHU(F)                    labor cost quality organic waste
Y(I,J)                    income
Y_SUB(I,J)                income
Y0_SUB(I,J)               benchmark income
GAP(I,J)                  gap between income and expenditure
TAU(G)                    tax rate
XI(G,*)                   tax wedge rest waste
XI_C(G,J,F)               tax wedge organic waste
NWT(I,J)                  Negishi weight
NWTSUM                    sum Negishi weight
NWTNORM                   normalization Negishi weight
TRANS_CSG(J)              transfer rest waste
TRANS_CSC(J)              transfer organic waste
TRANS_WD                  transfer landfilling
Y0(I,J)                   initial income
RHOn                      parameter Negishi adjustment
SMALL_P                   parameter iteration
SCALE                     scaling parameter
PT_CSG(J)                 price including subsidy rest waste
PT_CSC(J,F)               price including subsidy org. waste
P(K)                      price
P_CSG(J)                  price collection rest waste
P_CSC(J,F)                price collection organic waste
P_WD(TWD,S)               price waste disposal
P_WD_BAR(TWD,S)           benchmark price waste disposal
P0(K)                     initial price
TRANS_C(J)                transfer
SUMVAR                    parameter for iteration
TAU(G)                    subsidy rate
TRANS(I,J)                transfers
A(G)                      technology parameter
A_WD(TWD,S)               technology parameter waste disposal
A_COMP(Tcomp)             technology parameter composting
A_CSC(J)                  technology parameter waste collection
ITER                      iteration count
SENSI                     sensitivity parameter
SOLVES                    parameter needed for two scenarios
PT                        price including subsidy;

* Initialization technology parameter and prizes
A(G)=TECHNOLOGY("A",G);
A_WD(TWD,S)=TECHNOLOGY_WD("A_WD",TWD,S);
A_COMP("COMP_L")=0.8;
A_COMP("COMP_H")=1.0;
A_CSC(J)=1;
```

```
P(K)=P_BAR(K);
P_CSG(J)=P_BAR("CS_G");
P_CSC(J,F)=P_BAR("CS_C");
P_WD(TWD,S)=P_BAR(TWD);
P_WD_BAR(TWD,s)=P_BAR(TWD);
PT(TWD,S)=0;
PT("LAND",S)=P_BAR("INCIN");

* Initialization value share production and consumption
THETA("LABOUR",G1)=P_BAR("LABOUR")*LAB_BAR(G1)
                  /(P_BAR("LABOUR")*LAB_BAR(G1)
                  +P_BAR("CAPITAL")*CAP_BAR(G1)
                  +P_BAR("CO2")*CO2_ER_BAR(G1)
                  +P_BAR("NOX")*NOX_ER_BAR(G1)
                  +P_BAR("CH4")*CH4_ER_BAR(G1));
THETA("CAPITAL",G1)=(P_BAR("CO2")*CO2_ER_BAR(G1)
                  +P_BAR("NOX")*NOX_ER_BAR(G1)
                  +P_BAR("CH4")*CH4_ER_BAR(G1)
                  +P_BAR("CAPITAL")*CAP_BAR(G1))
                  /(P_BAR("LABOUR")*LAB_BAR(G1)
                  +P_BAR("CAPITAL")*CAP_BAR(G1)
                  +P_BAR("CO2")*CO2_ER_BAR(G1)
                  +P_BAR("NOX")*NOX_ER_BAR(G1)
                  +P_BAR("CH4")*CH4_ER_BAR(G1));
THETA("CO2",G1)$(CO2_ER_BAR(G1) NE 0)=P_BAR("CO2")*CO2_ER_BAR(G1)
                  /(P_BAR("CAPITAL")*CAP_BAR(G1)
                  +P_BAR("CO2")*CO2_ER_BAR(G1)
                  +P_BAR("NOX")*NOX_ER_BAR(G1)
                  +P_BAR("CH4")*CH4_ER_BAR(G1));
THETA("NOX",G1)$(NOX_ER_BAR(G1) NE 0)=P_BAR("NOX")*NOX_ER_BAR(G1)
                  /(P_BAR("CAPITAL")*CAP_BAR(G1)
                  +P_BAR("CO2")*CO2_ER_BAR(G1)
                  +P_BAR("NOX")*NOX_ER_BAR(G1)
                  +P_BAR("CH4")*CH4_ER_BAR(G1));
THETA("CH4",G1)$(CH4_ER_BAR(G1) NE 0)=P_BAR("CH4")*CH4_ER_BAR(G1)
                  /(P_BAR("CAPITAL")*CAP_BAR(G1)
                  +P_BAR("CO2")*CO2_ER_BAR(G1)
                  +P_BAR("NOX")*NOX_ER_BAR(G1)
                  +P_BAR("CH4")*CH4_ER_BAR(G1));
THETA_CSG("LABOUR",J)=P_BAR("LABOUR")*LAB_CSG_BAR(J)
                  /(P_BAR("LABOUR")*LAB_CSG_BAR(J)+P_BAR("CAPITAL")
                  *CAP_CSG_BAR(J));
THETA_CSG("CAPITAL",J)=P_BAR("CAPITAL")*CAP_CSG_BAR(J)
                  /(P_BAR("LABOUR")*LAB_CSG_BAR(J)+P_BAR("CAPITAL")
                  *CAP_CSG_BAR(J));
THETA_CSC("LABOUR",J,F)=P_BAR("LABOUR")*LAB_CSC_BAR(J,F)
                  /(P_BAR("LABOUR")*LAB_CSC_BAR(J,F)+P_BAR("CAPITAL")
                  *CAP_CSC_BAR(J,F));
THETA_CSC("CAPITAL",J,F)=P_BAR("CAPITAL")*CAP_CSC_BAR(J,F)
                  /(P_BAR("LABOUR")*LAB_CSC_BAR(J,F)+P_BAR("CAPITAL")
                  *CAP_CSC_BAR(J,F));
THETA_CSG("INCIN",J)=P_BAR("INCIN")*SUM(S,WTS_BAR(J,"INCIN",S))
                  /(P_BAR("INCIN")*SUM(S,WTS_BAR(J,"INCIN",S))
                  +SUM(S,PT("LAND",S)*WTS_BAR(J,"LAND",S)));
THETA_CSG("LAND",J)=SUM(S,PT("LAND",S)*WTS_BAR(J,"LAND",S))
                  /(P_BAR("INCIN")*SUM(S,WTS_BAR(J,"INCIN",S))
                  +SUM(S,PT("LAND",S)*WTS_BAR(J,"LAND",S)));
THETA_WD("LABOUR",TWD,S)=P_BAR("LABOUR")*LAB_WD_BAR(TWD,S)
                  /(P_BAR("LABOUR")*LAB_WD_BAR(TWD,S)
                  +P_BAR("CAPITAL")*CAP_WD_BAR(TWD,S)
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                  +P_BAR("CO2")*CO2_ER_WD_BAR(TWD,S)
                  +P_BAR("NOX")*NOX_ER_WD_BAR(TWD,S)+P_BAR("CH4")
                  *CH4_ER_WD_BAR(TWD,S));
THETA_WD("ER_CAP",TWD,S)=(P_BAR("CAPITAL")*CAP_WD_BAR(TWD,S)
                  +P_BAR("CO2")*CO2_ER_WD_BAR(TWD,S)+P_BAR("NOX")
                  *NOX_ER_WD_BAR(TWD,S)+P_BAR("CH4")
                  *CH4_ER_WD_BAR(TWD,S))
                  /(P_BAR("LABOUR")*LAB_WD_BAR(TWD,S)
                  +P_BAR("CAPITAL")*CAP_WD_BAR(TWD,S)
                  +P_BAR("CO2")*CO2_ER_WD_BAR(TWD,S)
                  +P_BAR("NOX")*NOX_ER_WD_BAR(TWD,S)+P_BAR("CH4")
                  *CH4_ER_WD_BAR(TWD,S));
THETA_WD("CAPITAL",TWD,S)=P_BAR("CAPITAL")*CAP_WD_BAR(TWD,S)
                  /(P_BAR("CAPITAL")*CAP_WD_BAR(TWD,S)
                  +P_BAR("CO2")*CO2_ER_WD_BAR(TWD,S)
                  +P_BAR("NOX")*NOX_ER_WD_BAR(TWD,S)+P_BAR("CH4")
                  *CH4_ER_WD_BAR(TWD,S));
THETA_WD("CO2",TWD,S)=P_BAR("CO2")*CO2_ER_WD_BAR(TWD,S)
                  /(P_BAR("CAPITAL")*CAP_WD_BAR(TWD,S)
                  +P_BAR("CO2")*CO2_ER_WD_BAR(TWD,S)
                  +P_BAR("NOX")*NOX_ER_WD_BAR(TWD,S)+P_BAR("CH4")
                  *CH4_ER_WD_BAR(TWD,S));
THETA_WD("NOX",TWD,S)=P_BAR("NOX")*NOX_ER_WD_BAR(TWD,S)
                  /(P_BAR("CAPITAL")*CAP_WD_BAR(TWD,S)
                  +P_BAR("CO2")*CO2_ER_WD_BAR(TWD,S)
                  +P_BAR("NOX")*NOX_ER_WD_BAR(TWD,S)+P_BAR("CH4")
                  *CH4_ER_WD_BAR(TWD,S));
THETA_WD("CH4",TWD,S)=P_BAR("CH4")*CH4_ER_WD_BAR(TWD,S)
                  /(P_BAR("CAPITAL")*CAP_WD_BAR(TWD,S)
                  +P_BAR("CO2")*CO2_ER_WD_BAR(TWD,S)
                  +P_BAR("NOX")*NOX_ER_WD_BAR(TWD,S)+P_BAR("CH4")
                  *CH4_ER_WD_BAR(TWD,S));
THETA_C(I4,J,"CS_G")=P_BAR("CS_G")*X_BAR(I4,J,"CS_G")
                  /(P_BAR("CS_G")*X_BAR(I4,J,"CS_G")+P_BAR("CS_C")
                  *X_BAR(I4,J,"CS_C"));
THETA_C(I4,J,"CS_C")=P_BAR("CS_C")*X_BAR(I4,J,"CS_C")
                  /(P_BAR("CS_G")*X_BAR(I4,J,"CS_G")+P_BAR("CS_C")
                  *X_BAR(I4,J,"CS_C"));
THETA_F(I3,J,F)=P_BAR("LABOUR")*LAB_C_BAR(I3,J,F)/(P_BAR("LABOUR")
                  *SUM(F1,LAB_C_BAR(I3,J,F1)));
THETA_F(I4,J,F)=(P_BAR("CS_C")*XCSC_BAR("CONS1A",J,F)
                  /SUM(F1,P_BAR("CS_C")*XCSC_BAR("CONS1A",J,F1)))
                  $(ORD(I4)=1)
                  +(P_BAR("CS_C")*XCSC_BAR("CONS2A",J,F)
                  /SUM(F1,P_BAR("CS_C")*XCSC_BAR("CONS2A",J,F1)))
                  $(ORD(I4)=2);
THETA_U(I2,J,"GOOD_WE")=P_BAR("GOOD_WE")*X_BAR(I2,J,"GOOD_WE")
                  /(P_BAR("GOOD_WE")*X_BAR(I2,J,"GOOD_WE")
                  +P_BAR("GOOD_WI")*X_BAR(I2,J,"GOOD_WI"));
THETA_U(I2,J,"GOOD_WI")=P_BAR("GOOD_WI")*X_BAR(I2,J,"GOOD_WI")
                  /(P_BAR("GOOD_WE")*X_BAR(I2,J,"GOOD_WE")
                  +P_BAR("GOOD_WI")*X_BAR(I2,J,"GOOD_WI"));


* Definition parameters transport costs
PARAMETERS transport
T(J,TWD,S)                transport matrix
TC                        transport costs
TS_BAR(J,TWD,S)           benchmark demand transport services;
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```
* Initialization parameters transport costs
T(J,TWD,S)=TRANSPORT(TWD,S,J);
TC=P("TRANS");
TS_BAR(J,TWD,S)=T(J,TWD,S)*WTS_BAR(J,TWD,S)/1000;

* Initialization income and Negishi weights
Y(C,J)=P_BAR("CAPITAL")*ENDK_BAR(C,J)+P_BAR("LABOUR")*(ENDL_BAR(C,J))
     -FEE(C,J)-LUMPSUM_BAR(C,J)
     -(P_BAR("LABOUR")*SUM(F,LAB_C_BAR("CONS1A",J,F)))
      $(ORD(C)=1)
     -(P_BAR("LABOUR")*SUM(F,LAB_C_BAR("CONS2A",J,F)))
      $(ORD(C)=2);
Y("GOV",J)=P("TAX")*ENDTAX(J)+P_BAR("CO2")*ENDCO2_BAR(J)
              +P_BAR("NOX")*ENDNOX_BAR(J)
              +P_BAR("CH4")*ENDCH4_BAR(J)+SUM((C),FEE(C,J))
              -SUB("GOV",J)+SUM((C),LUMPSUM_BAR(C,J));
Y("CONS1A",J)=P_BAR("LABOUR")*SUM(F,LAB_C_BAR("CONS1A",J,F));
Y("CONS2A",J)=P_BAR("LABOUR")*SUM(F,LAB_C_BAR("CONS2A",J,F));
FEE("GOV",J)=0;
NWT(I,J)=Y(I,J)/SUM((I1,J1),Y(I1,J1));
XI("CS_G","M1")=-((P_BAR("CS_G")/P_BAR("GOOD_WE"))
                 *NWT("GOV","m1")*100)
                 /(X_BAR("GOV","m1","GOOD_WE")
                 -(3*P_BAR("LAND")/P_BAR("CS_G")
                 *3-16*P_BAR("CS_C")/P_BAR("CS_G")-4)*(P_BAR("CS_G")
                 /P_BAR("GOOD_WE"))*NWT("GOV","m1"));
XI("CS_G",J)=XI("CS_G","M1");
XI_C("CS_C",J,F)=P_BAR("CS_C")/P_BAR("CS_G")*XI("CS_G","M1");
XI("LAND",S)=-3*P_BAR("LAND")/P_BAR("CS_G")*XI("CS_G","M1");
TRANS_CSG(J)=XI("CS_G",J)*SUM(I,X_BAR(I,J,"CS_G"));
TRANS_CSC(J)=XI("CS_C",J)*SUM(I,X_BAR(I,J,"CS_C"));
TRANS_WD("LAND",J)=-MUNICIPALITY("TAX",J,"CS_G");
NWTNORM=100+SUM(J,XI("CS_G",J))+SUM(S,XI("LAND",S))
        +SUM((J,F),XI_C("CS_C",J,F));
NWTSUM=SUM((I,J),NWT(I,J));
NWT(I,J)=NWT(I,J)*NWTNORM/NWTSUM;
TAU("CS_C")=-1;
TAU("CS_G")=-1;
TAU("LAND")=3;
ITER=0;

* Definition of dummy parameter to ensure that inaction of waste
* treatment unit is possible
PARAMETER
M(K,TWD,S)    dummy

* Initialization of dummy parameter
M("CAPITAL",TWD,S)=0.0001*CAP_WD_BAR(TWD,S)
                   /(CAP_WD_BAR(TWD,S)+LAB_WD_BAR(TWD,S));
M("LABOUR",TWD,S)=0.0001*LAB_WD_BAR(TWD,S)
                   /(CAP_WD_BAR(TWD,S)+LAB_WD_BAR(TWD,S));

* Initialization of benchmark waste treatment middle and large units
CAP_WD_BAR(TWD,"MIDDLE")=0;
LAB_WD_BAR(TWD,"MIDDLE")=0;
WTS_COMP_BAR("MIDDLE")=0;
TAX_BAR("MIDDLE")=0;
CAP_WD_BAR(TWD,"B")=0;
LAB_WD_BAR(TWD,"B")=0;
WTS_COMP_BAR("B")=0;
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TAX_BAR("B")=0;

* Definition variables
POSITIVE VARIABLES
Q(G)                production
Q_CSG               production collection services rest waste
Q_CSC(J,F)          production collection services organic waste
Q_WD(TWD,S)         production waste disposal services
X(I2,J,G)           consumption
CAP(G)              capital use
CAP_CSG             capital use collection rest waste
CAP_CSC             capital use collection organic waste
CAP_WD(TWD,S)       capital use waste treatment
LAB(G)              labor use
LAB_CSG             labor use collection rest waste
LAB_CSC             labor use collection organic waste
LAB_WD(TWD,S)       labor use waste treatment
TAX(S)              tax landfilling
U(I,J)              utility
TWG                 total supply rest waste
TWC                 total supply organic waste
ENDOWL(I4,J)        labor supply
WASTE(I,J)          generation waste
XCSC(I,J,F)         quality organic waste
LAB_C(I,J,F)        labor use composting
WTS(J,TWD,S)        use waste treatment services
WTS_COMP(S)         use waste treatment services
TS(J,TWD,S)         transport services
X_R_WASTE(I,J)      rest waste
X_O_WASTE(I,J,F)    organic waste
O_WASTE             total organic waste
Tland               total demand landfilling services;

VARIABLES
WELFARE             total welfare;

* Definition equations
EQUATIONS
QWELFARE            equation total welfare
QUTIL(I,J)          utility function
QUTIL2(I,J)         utility function
QPROD(G)            production function goods
QPRODWD(TWD,S)      production function waste disposal services
QPRODWD2(S)         production function waste disposal services
QPRODCSG_1          production function collection rest waste
QPRODCSG_2          production function collection rest waste
QPRODCSG_2B         production function collection rest waste
QPRODCSG_3          production function collection rest waste
QPRODCSC_1          production function collection organic waste
QPRODCSC_2          production function collection organic waste
QPRODCSC_2B         production function collection organic waste
QPRODCSC_3          production function collection organic waste
QPRODCSC_3B         production function collection organic waste
QPRODCSC_3C(J,S)    production function collection organic waste
QBALFACL            balance equation labor
QBALFACK            balance equation capital
QBALFACCO2          balance equation CO2
QBALFACNOX          balance equation NOX
QBALFACCH4          balance equation CH4
QBALWTS_INCIN       balance equation incineration
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QBALWTS_INCIN2      balance equation incineration
QBALWTS2            balance equation composting services
QBALLAND1           balance equation landfilling
QBALLAND2           balance equation landfilling
QBALTRANS           balance function transportation services
QBALGOOD            balance equation demand good
QPRODW1             production function waste
QPRODW2             production function waste
QBALCSG1            balance equation collection rest waste flat fee
QBALCSG2            balance equation collection rest waste flat fee
QBALCSG3            balance equation collection rest waste unit price
QBALCSC1            balance equation collection compost flat fee
QBALCSC2            balance equation collection compost flat fee
QPRODENDL           calculation labor availability
QPRODXCSC            production organic waste
QBALCOMPOST          balance compost virtual consumer
QBALTS_1(J,TWD,S)    balance equation demand transport services
QPRODWASTE1          equation calculating rest waste in tons
QPRODWASTE2          equation calculating organic waste in tonnes
QPRODWASTE2A(I3,J,F) equation calculating organic waste in tonnes
QPRODWASTE3          equation calculating organic waste in tonnes;

* Total welfare function
QWELFARE..       WELFARE=E=SUM(J,NWT("GOV",J)*LOG(U("GOV",J)))
                          +SUM((C,J),NWT(C,J)*LOG(U(C,J)))
                          -SUM(J,XI("CS_G",J)*TWG(J))
                          -SUM((J,F),XI_C("CS_C",J,F)*TWC(J,F))
                          -SUM(S,XI("LAND",S)*TLAND(S));
* Calculation utility consumer
QUTIL(I2,J)..    U(I2,J)=L=U_BAR(I2,J)*(THETA_U(I2,J,"GOOD_WE")
                          *(X(I2,J,"GOOD_WE")/X_BAR(I2,J,"GOOD_WE"))
                          **(-RHO_U(I2,J))
                          +(THETA_U(I2,J,"GOOD_WI")
                          *(X(I2,J,"GOOD_WI")/X_BAR(I2,J,"GOOD_WI"))
                          **(-RHO_U(I2,J)))
                          $(THETA_U(I2,J,"GOOD_WI") NE 0))
                          **(-1/RHO_U(I2,J));
QUTIL2(I3,J)..   U(I3,J)=L=U_BAR(I3,J)*(THETA_F(I3,J,"LOW")
                          *(LAB_C(I3,J,"LOW")/LAB_C_BAR(I3,J,"LOW"))
                          **(-RHO_F(I3,J))
                          + THETA_F(I3,J,"HIGH")
                          *(LAB_C(I3,J,"HIGH")
                          /LAB_C_BAR(I3,J,"HIGH"))
                          **(-RHO_F(I3,J)))**(-1/RHO_F(I3,J));
* Production function consumption good and transport services
QPROD(G1)..      Q(G1)=L=A(G1)*Q_BAR(G1)*
                        (THETA("LABOUR",G1)*(LAB(G1)/LAB_BAR(G1))
                        **(-RHO(G1))
                        +THETA("CAPITAL",G1)*(CAP(G1)/CAP_BAR(G1))
                        **(-RHO(G1)))**(-1/RHO(G1));
* Nested CES Leontief production function collection rest waste
QPRODCSG_1(J)..  Q_CSG(J)=L=A("CS_G")*Q_CSG_BAR(J)
                           *(THETA_CSG("CAPITAL",J)
                           *(CAP_CSG(J)/CAP_CSG_BAR(J))
                           **(-RHO("CS_G"))
                           +THETA_CSG("LABOUR",J)
                           *(LAB_CSG(J)/LAB_CSG_BAR(J))
                           **(-RHO("CS_G")))
                           **(-1/RHO("CS_G"));
QPRODCSG_2(J)..  Q_CSG(J)=L=Q_CSG_BAR(J)
```

```
                             *(THETA_CSG("LAND",J)
                             *((SUM(S,WTS(J,"LAND",S)))
                             /(SUM(S,WTS_BAR(J,"LAND",S))))**(-RHO3)
                             +THETA_CSG("INCIN",J)
                             *((SUM(S,WTS(J,"INCIN",S)))
                             /(SUM(S,WTS_BAR(J,"INCIN",S))))**(-RHO3))
                             **(-1/RHO3);
QPRODCSG_2B(J,TWD)$(ORD(TWD) NE 1 OR 2)..   SUM(S,WTS(J,TWD,S))=G=0.1
                             *SUM(S,WTS_BAR(J,TWD,S));
QPRODCSG_3(J)..   Q_CSG(J)=L=Q_CSG_BAR(J)/SUM((S),TS_BAR(J,"INCIN",S)
                             +TS_BAR(J,"LAND",S))*SUM(S,TS(J,"LAND",S)
                             +TS(J,"INCIN",S));
```

*** Production function waste treatment**

```
QPRODWD(TWD,S)..  Q_WD(TWD,S)+A_WD(TWD,S)*SUM(PF,M(PF,TWD,S))
                             /P_WD_BAR(TWD,S)=L=A_WD(TWD,S)
                             *(Q_WD_BAR(TWD,S)+SUM(PF,M(PF,TWD,S))
                             /P_WD_BAR(TWD,S))
                             *(THETA_WD("LABOUR",TWD,S)
                             *((LAB_WD(TWD,S)+M("LABOUR",TWD,S))
                             /(LAB_WD_BAR(TWD,S)+M("LABOUR",TWD,S)))
                             **(-RHO(TWD))
                             +THETA_WD("ER_CAP",TWD,S)
                             *((THETA_WD("CAPITAL",TWD,S)
                             *((CAP_WD(TWD,S)+M("CAPITAL",TWD,S))
                             /(CAP_WD_BAR(TWD,S)+M("CAPITAL",TWD,S)))
                             **(-RHO2(TWD))
                             +(THETA_WD("CO2",TWD,S)
                             *((CO2_ER_WD(TWD,S)+M("CO2",TWD,S))
                             /(CO2_ER_WD_BAR(TWD,S)+M("CO2",TWD,S)))
                             **(-RHO2(TWD)))
                             $(THETA_WD("CO2",TWD,S) NE 0)
                             +(THETA_WD("NOX",TWD,S)
                             *((NOX_ER_WD(TWD,S)+M("NOX",TWD,S))
                             /(NOX_ER_WD_BAR(TWD,S)+M("NOX",TWD,S)))
                             **(-RHO2(TWD)))
                             $(THETA_WD("NOX",TWD,S) NE 0)
                            +(THETA_WD("CH4",TWD,S)
                             *((CH4_ER_WD(TWD,S)+M("CH4",TWD,S))
                             /(CH4_ER_WD_BAR(TWD,S)+M("CH4",TWD,S)))
                             **(-RHO2(TWD)))
                             $(THETA_WD("CH4",TWD,S) NE 0))
                             **(-1/RHO2(TWD)))**(-RHO(TWD)))
                             **(-1/RHO(TWD));
QPRODWD2(S)..     Q_WD("COMP_L",S)=L=WTS_COMP(S)
                             *(Q_WD_BAR("COMP_L","SMALL")
                              /WTS_COMP_BAR("SMALL"));
```

*** Nested CES Leontief production function collection organic waste**

```
QPRODCSC_1(J,F).. Q_CSC(J,F)=L=A("CS_C")*Q_CSC_BAR(J,F)
                             *(THETA_CSC("CAPITAL",J,F)
                             *(CAP_CSC(J,F)/CAP_CSC_BAR(J,F))
                             **(-RHO("CS_C"))
                             +THETA_CSC("LABOUR",J,F)
                             *(LAB_CSC(J,F)/LAB_CSC_BAR(J,F))
                             **(-RHO("CS_C")))
                             **(-1/RHO("CS_C"));
QPRODCSC_2(J)..   Q_CSC(J,"LOW")=L=A_CSC(J)*Q_CSC_BAR(J,"LOW")
                             *SUM(S,WTS(J,"COMP_L",S))
                             /SUM(S,WTS_BAR(J,"COMP_L",S));
QPRODCSC_2B(J)..  Q_CSC(J,"HIGH")=L=A_CSC(J)*Q_CSC_BAR(J,"HIGH")
                             *SUM(S,WTS(J,"COMP_H",S))
```

```
                              /SUM(S,WTS_BAR(J,"COMP_H",S));
QPRODCSC_3(J)..    Q_CSC(J,"LOW")=L=A_CSC(J)*Q_CSC_BAR(J,"LOW")
                              /SUM((S),TS_BAR(J,"COMP_L",S))
                              *SUM((S),TS(J,"COMP_L",S));
QPRODCSC_3B(J)..    Q_CSC(J,"HIGH")=L=A_CSC(J)*Q_CSC_BAR(J,"HIGH")
                              /SUM((S),TS_BAR(J,"COMP_H",S))
                              *SUM((S),TS(J,"COMP_H",S));
QPRODCSC_3C(J,S).. WTS(J,"COMP_L",S)/Q_CSC(J,"LOW")=E=
                              WTS(J,"COMP_H",S)/Q_CSC(J,"HIGH");
* Balance equation labor, capital, and emission rights
QBALFACL..        SUM(G1,LAB(G1))+SUM(J,LAB_CSG(J))
                     +SUM((J,F),LAB_CSC(J,F))
                     +SUM((TWD,S),LAB_WD(TWD,S))=L=
                      SUM((I4,J),ENDOWL(I4,J));
QBALFACK..         SUM(G1,CAP(G1))+SUM(J,CAP_CSG(J))
                     +SUM((J,F),CAP_CSC(J,F))
                     +SUM((TWD,S),CAP_WD(TWD,S))=L=
                      SUM((I,J),ENDK_BAR(I,J));
* Balance equation demand incineration services
QBALWTS_INCIN..    SUM((J),WTS(J,"INCIN","SMALL"))
                     +SUM((S),WTS_COMP(S))=L=Q_WD("INCIN","SMALL");
QBALWTS_INCIN2(S)$(ORD(S) NE 1).. SUM((J),WTS(J,"INCIN",S))
                                    =L=Q_WD("INCIN",S);
* Balance equation demand composting services
QBALWTS2(TCOMP,S)..   SUM((J),WTS(J,TCOMP,S))=L=Q_WD(TCOMP,S);
* Balance equation demand waste treatment
QBALLAND1(S)..        SUM(J,WTS(J,"LAND",S))=E=TLAND(S);
QBALLAND2(S)..        TLAND(S)=E=Q_WD("LAND",S);
* Demand transport services
QBALTS_1(J,TWD,S)..   TS(J,TWD,S)=E=T(J,TWD,S)*WTS(J,TWD,S)/1000;
* Balance equation demand transport services
QBALTRANS..           SUM((J,TWD,S),TS(J,TWD,S))=L=Q("TRANS");
* Balance equation demand consumption good
QBALGOOD(G2)..        SUM((I2,J),X(I2,J,G2))=L=Q(G2);
* Generation waste as function consumption
QPRODW1(I4,J)..       WASTE(I4,J)=E=SUM(G2,BETA(I4,J,G2)*X(I4,J,G2));
* Demand collection services
QPRODW2(I4,J)..       WASTE(I4,J)=E=WASTE_BAR(I4,J)
                              *(THETA_C(I4,J,"CS_G")
                              *(X(I4,J,"CS_G")/X_BAR(I4,J,"CS_G"))
                              **(-RHO_C(I4,J))
                              +THETA_C(I4,J,"CS_C")*
                              ((THETA_F(I4,J,"LOW")
                              *(((XCSC(I4,J,"LOW")
                              +XCSC("CONS1A",J,"LOW"))
                              /XCSC_BAR("CONS1A",J,"LOW"))$(ORD(I4)=1)
                              + (((XCSC(I4,J,"LOW")
                              +XCSC("CONS2A",J,"LOW"))
                              /XCSC_BAR("CONS2A",J,"LOW")))
                              $(ORD(I4)=2))**(-RHO_F(I4,J))
                              +THETA_F(I4,J,"HIGH")
                              *((((XCSC(I4,J,"HIGH")
                              +XCSC("CONS1A",J,"HIGH"))
                              /XCSC_BAR("CONS1A",J,"HIGH")))
                              $(ORD(I4)=1)
                              + (((XCSC(I4,J,"HIGH")
                              +XCSC("CONS2A",J,"HIGH"))
                              /XCSC_BAR("CONS2A",J,"HIGH")))
                              $(ORD(I4)=2))
                              **(-RHO_F(I4,J)))**(-1/RHO_F(I4,J)))
```

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                           **(-RHO_C(I4,J)))**(-1/RHO_C(I4,J));

* Calculation demand collection services in tonnes
QPRODWASTE1(I4,J)..       X_R_WASTE(I4,J)=E=X(I4,J,"CS_G");
QPRODWASTE2(I4,J,F)..     X_O_WASTE(I4,J,F)=E=(XCSC(I4,J,F)
                               +XCSC("CONS1A",J,F)$(ORD(I4)=1)
                               +XCSC("CONS2A",J,F)$(ORD(I4)=2))
                               /SUM(F1,XCSC(I4,J,f1)
                               +XCSC("CONS1A",J,F1)$(ORD(I4)=1)
                               +XCSC("CONS2A",J,F1)$(ORD(I4)=2))
                               *(WASTE(I4,J)-X(I4,J,"CS_G"));
QPRODWASTE2A(I3,J,F)..    X_O_WASTE(I3,J,F)=E=XCSC(I3,J,F)
                               /(SUM(F1,XCSC(I3,J,F1)
                               +XCSC("CONS1",J,F1)$(ORD(I3)=1)
                               +XCSC("CONS2",J,F1)$(ORD(I3)=2)))
                               *((WASTE("CONS1",J)
                               -X("CONS1",J,"CS_G"))$(ORD(I3)=1)
                               +(WASTE("CONS2",J)
                               -X("CONS2",J,"CS_G"))$(ORD(I3)=2));
QPRODWASTE3(I4,J)..       X_R_WASTE(I4,J)+SUM(F,X_O_WASTE(I4,J,F))
                                           =E=WASTE(I4,J);
* Balance equation demand collection rest waste
QBALCSG1(J)$(SOLVES=0)..      SUM(I4,X_R_WASTE(I4,J))=E=TWG(J);
QBALCSG2(J)$(SOLVES=0)..      TWG(J)=E=Q_CSG(J);
QBALCSG3(J)$(SOLVES=1)..      SUM(I4,X_R_WASTE(I4,J))=L=Q_CSG(J);
* Balance equation demand collection organic waste
QBALCSC1(J,F)..               SUM((I4),X_O_WASTE(I4,J,F))=E=TWC(J,F);
QBALCSC2(J,F)..               TWC(J,F)=E=Q_CSC(J,F);
*QBALCSC3$(SOLVES=1)..        SUM((C,F),XCSC(I,J,F))=L=Q("CS_C");
* Calculation labor supply endogenously determined in model
QPRODENDL(I4,J)..             ENDOWL(I4,J)=L=ENDL_BAR(I4,J)
                              -SUM(F,LAB_C(I4,J,F))
                              -(SUM(F,LAB_C("CONS1A",J,F)))$(ORD(I4)=1)
                              -(SUM(F,LAB_C("CONS2A",J,F)))
                              $(ORD(I4)=2);
* Labor input generation organic waste
QPRODXCSC(C,J,F)..            XCSC(C,J,F)=E=MHU(F)*LAB_C(C,J,F);
* Equation ensuring that organic waste is generated in benchmark
QBALCOMPOST(I3,J)..           SUM(F,LAB_C(I3,J,F))=L=
                               SUM(F,Lab_C_BAR(I3,J,F));

* Definition of model
MODEL QUALITY /ALL/;
QUALITY.SCALEOPT=1;

* Defining lower and upper bounds variables
Q.LO(G1)=0.1*Q_BAR(G1);
Q.UP(G1)=10*Q_BAR(G1);
Q_CSC.LO(J,F)=0.1*Q_CSC_BAR(J,F);
Q_CSC.UP(J,F)=10*Q_CSC_BAR(J,F);
Q_CSG.LO(J)=0.1*Q_CSG_BAR(J);
Q_CSG.UP(J)=10*Q_CSG_BAR(J);
Q_WD.UP(TWD,S)=10*Q_WD_BAR(TWD,"SMALL");
X.LO(I4,J,"GOOD_WE")=0.1*X_BAR(I4,J,"GOOD_WE");
X.LO(I4,J,"GOOD_WI")=0.1*X_BAR(I4,J,"GOOD_WI");
X.LO(I4,J,"CS_G")=0.1*X_BAR(I4,J,"CS_G");
X.LO("GOV",J,"GOOD_WE")=0.1*X_BAR("GOV",J,"GOOD_WE");
X.FX("GOV",J,"GOOD_WI")=0;
X.UP(I2,J,G)=10*X_BAR(I2,J,G);
CAP.LO(G1)=0.1*CAP_BAR(G1);
```

```
LAB.LO(G1)=0.1*LAB_BAR(G1);
CAP.UP(G1)=10*CAP_BAR(G1);
LAB.UP(G1)=10*LAB_BAR(G1);
CAP_WD.UP(TWD,S)=10*CAP_WD_BAR(TWD,"SMALL");
LAB_WD.UP(TWD,S)=10*LAB_WD_BAR(TWD,"SMALL");
WTS_COMP.UP(S)=10*WTS_COMP_BAR("SMALL");
CAP_CSG.LO(J)=0.1*CAP_CSG_BAR(J);
LAB_CSG.LO(J)=0.1*LAB_CSG_BAR(J);
CAP_CSG.UP(J)=10*CAP_CSG_BAR(J);
LAB_CSG.UP(J)=10*LAB_CSG_BAR(J);
CAP_CSC.LO(J,F)=0.1*CAP_CSC_BAR(J,F);
LAB_CSC.LO(J,F)=0.1*LAB_CSC_BAR(J,F);
CAP_CSC.UP(J,F)=10*CAP_CSC_BAR(J,F);
LAB_CSC.UP(J,F)=10*LAB_CSC_BAR(J,F);
U.LO(I,J)=0.1*U_BAR(I,J);
TWG.LO(J)=0.1*TWASTEG_BAR(J);
TWC.LO(J,F)=0.1*TWASTEC_BAR(J);
TWG.UP(J)=10*TWASTEG_BAR(J);
TWC.UP(J,F)=10*TWASTEC_BAR(J);
ENDOWL.LO(I4,J)=0.9*ENDL_BAR(I4,J);
ENDOWL.UP(I4,J)=ENDL_BAR(I4,J);
WASTE.LO(I4,J)=0.1*WASTE_BAR(I4,J);
WASTE.UP(I4,J)=10*WASTE_BAR(I4,J);
XCSC.LO(I3,J,F)=0.9*XCSC_BAR(I3,J,F);
XCSC.UP(I3,J,F)=1.5*XCSC_BAR(I3,J,F);
XCSC.UP(I4,J,F)=10*XCSC_BAR("CONS1A",J,F)$(ORD(I4)=1)
                +10*XCSC_BAR("CONS2A",J,F)$(ORD(I4)=2);
X.LO(I4,J,"CS_G")=0.1*X_BAR(I4,J,"CS_G");
X.UP(I4,J,"CS_G")=10*X_BAR(I4,J,"CS_G");
X_O_WASTE.LO(C,J,F)=0.1*XCSC_BAR(C,J,F);
X_O_WASTE.UP(I3,J,F)=10*XCSC_BAR(I3,J,F);
X_O_WASTE.UP(I4,J,F)=10*XCSC_BAR("CONS1A",J,F)$(ORD(I4)=1)
                     +10*XCSC_BAR("CONS2A",J,F)$(ORD(I4)=2);
XCSC.FX("GOV",J,F)=0;
LAB_C.UP(I4,J,F)=0.02*ENDL_BAR(I4,J);
LAB_C.LO(I3,J,F)=0.5*LAB_C_BAR(I3,J,F);
LAB_C.UP(I3,J,F)=1.5*LAB_C_BAR(I3,J,F);
WTS.UP(J,TWD,S)=10*WTS_BAR(J,TWD,"SMALL");
TS.UP(J,TWD,S)=10*TS_BAR(J,TWD,"SMALL");
U.UP(I,J)=10*U_BAR(I,J);
TLAND.UP(S)=10*Q_WD_BAR("LAND","SMALL");

* Defining initial levels of variables
Q.L(G)=Q_BAR(G);
Q.L("TRANS")=Q_BAR("TRANS");
Q_WD.L(TWD,S)=Q_WD_BAR(TWD,S);
Q_WD.L(TWD,"MIDDLE")=0;
Q_WD.L(TWD,"B")=0;
Q_CSG.L(J)=Q_CSG_BAR(J);
Q_CSC.L(J,F)=Q_CSC_BAR(J,F);
X.L(I2,J,G)=X_BAR(I2,J,G);
X.L("GOV",J,G)=X_BAR("GOV",J,G);
CAP.L(G1)=CAP_BAR(G1);
LAB.L(G1)=LAB_BAR(G1);
CO2_ER.L(G1)=CO2_ER_BAR(G1);
NOX_ER.L(G1)=NOX_ER_BAR(G1);
CH4_ER.L(G1)=CH4_ER_BAR(G1);
CAP_WD.L(TWD,"SMALL")=CAP_WD_BAR(TWD,"SMALL");
LAB_WD.L(TWD,"SMALL")=LAB_WD_BAR(TWD,"SMALL");
WTS_COMP.L("SMALL")=WTS_COMP_BAR("SMALL");
```

```
CO2_ER_WD.L(TWD,"SMALL")=CO2_ER_WD_BAR(TWD,"SMALL");
NOX_ER_WD.L(TWD,"SMALL")=NOX_ER_WD_BAR(TWD,"SMALL");
CH4_ER_WD.L(TWD,"SMALL")=CH4_ER_WD_BAR(TWD,"SMALL");
WTS.L(J,"LAND",S)=WTS_BAR(J,"LAND",S);
WTS.L(J,"INCIN",S)=WTS_BAR(J,"INCIN",S);
WTS.L(J,TCOMP,S)=WTS_BAR(J,TCOMP,S);
TS.L(J,TWD,S)=TS_BAR(J,TWD,S);
CAP_CSG.L(J)=CAP_CSG_BAR(J);
LAB_CSG.L(J)=LAB_CSG_BAR(J);
CAP_CSC.L(J,F)=CAP_CSC_BAR(J,F);
LAB_CSC.L(J,F)=LAB_CSC_BAR(J,F);
U.L(I2,J)=U_BAR(I2,J);
U.L("GOV",J)=U_BAR("GOV",J);
U.L(I3,J)=U_BAR(I3,J);
TWG.L(J)=TWASTEG_BAR(J);
TWC.L(J,"LOW")=SUM(I3,XCSC_BAR(I3,J,"LOW"))
               /SUM((I3,F),XCSC_BAR(I3,J,F))*TWASTEC_BAR(J);
TWC.L(J,"HIGH")=SUM(I3,XCSC_BAR(I3,J,"HIGH"))
               /SUM((I3,F),XCSC_BAR(I3,J,F))*TWASTEC_BAR(J);
ENDOWL.L("CONS1",J)=ENDL_BAR("CONS1",J)
               -SUM(F,LAB_C_BAR("CONS1A",J,F));
ENDOWL.L("CONS2",J)=ENDL_BAR("CONS2",J)
               -SUM(F,LAB_C_BAR("CONS2A",J,F));
WASTE.L(I,J)=X_BAR(I,J,"CS_G")+X_BAR(I,J,"CS_C");
XCSC.L(I,J,F)=XCSC_BAR(I,J,F);
X_R_WASTE.L(I4,J)=X_BAR(I4,J,"CS_G");
X_O_WASTE.L(I4,J,F)=XCSC_BAR("CONS1A",J,F)$(ORD(I4)=1)
                    +XCSC_BAR("CONS2A",J,F)$(ORD(I4)=2);
X_O_WASTE.L(I3,J,F)=XCSC_BAR(I3,J,F);
LAB_C.L(I,J,F)=LAB_C_BAR(I,J,F);
TLAND.L(S)=Q_WD_BAR("LAND",S);

* Defining parameters for iteration
RHOn = 0.030;
SMALL_P = 0.000018;
SUMVAR=1000;
GAP(I,J)=0;

* Include files for report writing
$INCLUDE "PARAMETER_IO_TABLE_S10.GMS";
$INCLUDE "BENCHMARK_IO_TABLE_S10.GMS";
$INCLUDE "EMISSION_TABLE_S10.GMS";

* Start loop for finding equilibrium solution for flat fee and unit-
based price
LOOP(SCEN,

* Parameter value in flat fee scenario
  IF(FLAT(scen),
       solves = 0;

* Change values parameters in unit-based price scenario
  ELSE
    solves = 1;
    TWG.FX(J)=0;
    TWC.L(J,F)=TWASTEC_BAR(J);
    FEE("CONS1",J)=-MUNICIPALITY("CS_C",J,"CONS1")
                   *P_BAR("CS_C")/1.06;
    FEE("CONS2",J)=-MUNICIPALITY("CS_C",J,"CONS2")
                   *P_BAR("CS_C")/1.06;
```

```
    SUB("GOV",J)=1.06*SUM(I4,FEE(I4,J));
    XI("CS_G",J)=0;
    TAU("CS_G")=0;
    SUMVAR=1000;
    ITER=0;
    TRANS(I,J)=0;
    TRANS_CSG(J)=0;
  );
* Start loop for determining optimal Negishi weights
LOOP (ITWEL $(SUMVAR GT SMALL_P),
  ITER = ITER+1;
* Solve model
  SOLVE QUALITY USING DNLP MAXIMIZING WELFARE;
* Calculate prices, income and budget constraint
  P0(K)=P(K);
  P("CAPITAL")=ABS(QBALFACK.M);
  P("LABOUR")=ABS(QBALFACL.M);
  P(G2)=ABS(QBALGOOD.M(G2));
  PT_CSG(J)=ABS(QBALCSG1.M(J))$(SOLVES=0)+0$(SOLVES=1);
  P_CSG(J)=ABS(QBALCSG2.M(J))$(SOLVES=0)
           +ABS(QBALCSG3.M(J))$(SOLVES=1);
  PT_CSC(J,F)=ABS(QBALCSC1.M(J,F));
  P_CSC(J,F)=ABS(QBALCSC2.M(J,F));
  P("TRANS")=ABS(QBALTRANS.M);
  P_WD("INCIN",S)$(ORD(S)=1)=ABS(QBALWTS_INCIN.M);
  P_WD("INCIN",S)$(ORD(S) NE 1)=ABS(QBALWTS_INCIN2.M(S));
  P_WD(TCOMP,S)=ABS(QBALWTS2.M(TCOMP,S));
  P("CO2")=ABS(QBALFACCO2.M);
  P("NOX")=ABS(QBALFACNOX.M);
  P("CH4")=ABS(QBALFACCH4.M);
  PT("LAND",S)=ABS(QBALLAND1.M(S));
  P_WD("LAND",S)=ABS(QBALLAND2.M(S));
  SCALE=P("CAPITAL");
  XI("CS_G",J)=TAU("CS_G")*P_CSG(J);
  XI_C("CS_C",J,F)=TAU("CS_C")*P_CSC(J,F);
  XI("LAND",S)=TAU("LAND")*P_WD("LAND",S);
  P(K)=P(K)/SCALE;
  P("CS_G")=0;
  P("CS_C")=0;
  P_CSC(J,F)=P_CSC(J,F)/SCALE;
  P_CSG(J)=P_CSG(J)/SCALE;
  P_WD(TWD,S)=P_WD(TWD,S)/SCALE;
  P(TWD)=P_WD(TWD,"SMALL");
  TRANS_CSG(J)=SUM((I4),XI("CS_G",J)/SCALE*X_R_WASTE.L(I4,J));
  TRANS_CSC(J)=SUM((I4,F),XI_C("CS_C",J,F)/SCALE
               *X_O_WASTE.L(I4,J,F));
  TRANS_WD("LAND",J)=SUM((S),XI("LAND",S)/SCALE*WTS.L(J,"LAND",S));
  PT_CSG(J)=PT_CSG(J)/SCALE;
  PT_CSC(J,F)=PT_CSC(J,F)/SCALE;
  PT("LAND",S)=PT("LAND",S)/SCALE;
  p("tax")=p0("tax");
  TRANS("GOV",J)=TRANS_CSG(J)+TRANS_CSC(J)+TRANS_WD("LAND",J);
  Y0(I,J)=Y(I,J);
  Y("GOV",J)=P("CO2")*ENDCO2_BAR(J)+P("NOX")*ENDNOX_BAR(J)
               +P("CH4")*ENDCH4_BAR(J)
               +SUM((I4),FEE(I4,J))+SUM((I4),LUMPSUM_BAR(I4,J))
               +TRANS("GOV",J);
  TRANS_C(J)=Y0("GOV",J)-Y("GOV",J);
  Y("CONS1",J)=P("CAPITAL")*ENDK_BAR("CONS1",J)+P("LABOUR")
               *ENDOWL.L("CONS1",J)
```

```
               -FEE("CONS1",J)-LUMPSUM_BAR("CONS1",J)
               -Z("CONS1",J)*TRANS_C(J);
  Y("CONS2",J)=P("CAPITAL")*ENDK_BAR("CONS2",J)+P("LABOUR")
               *ENDOWL.L("CONS2",J)
               -FEE("CONS2",J)-LUMPSUM_BAR("CONS2",J)
               -Z("CONS2",J)*TRANS_C(J);
  Y("GOV",J)=Y0("GOV",J);
  GAP("CONS1",J)=Y("CONS1",J)-SUM(G2,P(G2)*X.L("CONS1",J,G2))
               -(PT_CSG(J)+P_CSG(J)$(SOLVES=1))
              *X_R_WASTE.L("CONS1",J)
               -SUM(F,PT_CSC(J,F)*X_O_WASTE.L("CONS1",J,F));
  GAP("CONS2",J)=Y("CONS2",J)-SUM(G2,P(G2)*X.L("CONS2",J,G2))
               -(PT_CSG(J)+P_CSG(J)$(SOLVES=1))
               *X_R_WASTE.L("CONS2",J)
               -SUM(F,PT_CSC(J,F)*X_O_WASTE.L("CONS2",J,F));
  GAP("GOV",J)=Y("GOV",J)-P("GOOD_WE")*X.L("GOV",J,"GOOD_WE");
* Calculate new Negishi weights
  SUMVAR = SUM((I,J), ABS(GAP(I,J)))/SUM((I,J), Y(I,J));
  SUMVAR = SUMVAR + (ABS(P("CAPITAL")
                  -P0("CAPITAL"))*SUM((I,J),ENDK_BAR(I,J))
                  +ABS(P("LABOUR")-P0("LABOUR"))
                  *SUM((I4,J),ENDOWL.L(I4,J))
                  +ABS(P("CO2")-P0("CO2"))*SUM(J,ENDCO2_BAR(J))
                  +ABS(P("NOX")-P0("NOX"))*SUM(J,ENDNOX_BAR(J))
                  +ABS(P("CH4")-P0("CH4"))*SUM(J,ENDCH4_BAR(J)))
                  /(P0("CAPITAL")*SUM((I,J),ENDK_BAR(I,J))
                  +P0("LABOUR")*SUM((I4,J),ENDOWL.L(I4,J))
                  +P0("CO2")*SUM(J,ENDCO2_BAR(J))
                  +P0("NOX")*SUM(J,ENDNOX_BAR(J))
                  +P0("CH4")*SUM(J,ENDCH4_BAR(J)));
  NWT(I,J)$(Y(I,J) NE 0) = NWT(I,J)/NWTNORM + RHON*GAP(I,J)/Y(I,J);
  NWT(I,J)$(Y(I,J) EQ 0)=0;
  NWT(I,J) = MAX(NWT(I,J),0);
  NWTNORM = 100+SUM((J),XI("CS_G",J))+SUM((J,F),XI_C("CS_C",J,F))
            +SUM(S,XI("LAND",S));
  NWTSUM = SUM((I,J), NWT(I,J));
  NWT(I,J) = NWT(I,J)*NWTNORM/NWTSUM;
* Stop iterations if income or Negishi weights equal zero
LOOP(I,
loop(j,
IF (NWT(I,J) EQ 0 OR Y(I,j) LE 0,
    SUMVAR=0;
    DISPLAY "NEGISHI WEIGHT OF ONE CONSUMER EQUALS ZERO,
    SOLVER STOPPED";
*    ERRORMESS=1;
 ););););
DISPLAY ITER;
* End of iterative loop (ITWEL)
);
* Include files for report writing
$INCLUDE "PERCENTAGE_S10.GMS";
$INCLUDE "ABS_RESULTS_S10.GMS";
$INCLUDE "PRICE_s10.GMS";
$INCLUDE "EMISSIONS.GMS";
$INCLUDE "RESULTS_IO_TABLE_S10.GMS";
* End of scenario loop (SCEN)
);
* Display warning messages if model did not find equilibrium solution
IF (ITER=CARD(ITWEL), DISPLAY "WARNING: MAXIMUM AMOUNT
                      OF ITERATIONS REACHED");
```

Curriculum Vitae

Heleen Bartelings was born May 23, 1975 in Veldhoven, the Netherlands. In 1993, she completed the secondary school (VWO) at Eindhoven Protestants Lyceum in Eindhoven. She then started the Msc study Agricultural and Environmental Economics at Wageningen University, with a specialization in Environmental Economics. She graduated in September 1998 and started her PhD-research in October 1998 on the topic 'Optimization models and life cycle analysis for agricultural, industrial and domestic waste management' at the Environmental Economics and Natural Resources Group at Wageningen University. This project was part of Materials Use and Spatial Scales in Industrial Metabolism (MUSSIM) research program financed by the Netherlands Organization for Scientific Research (NWO). Results of this project have been presented at national and international conferences, including the Second World Congress of Environmental and Resource Economics in Monterey, the 7th International Conference on Solid Waste Technology and Management in Philadelphia, the International Workshop on Empirical Modeling of the Economy and the Environment in Mannheim, and the Science and Culture of Industrial Ecology ISIE 2001 Meeting in Leiden. In 2002, she received the diploma of the Netherlands Network of Economics (NAKE) research school and in 2003 she received the diploma of the Socio-Economic and Natural Sciences of the Environment (SENSE) research school. Since May 2003, she has been working as a consultant at the consultancy and research bureau: Aarts, de Jong, Wilms and Goudriaan Public Economics BV (APE) in The Hague, The Netherlands.

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