Regulating the Dutch Waste Market

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> "Ik wensch lessen te ontvangen; ik gevoel, in het besef mijner zwakheid, de behoefte om mijne mening door het oordeel van achtbare regters bevestigd te zien. Maar tevens wensch ik van niemand afhankelijk te zijn; het *nemini me mancipavi* (ik heb mij aan niemand verslaafd), in den zin waarin het, zonder zelfverheffing, een iegelijk past, mij ten regel te blijven stellen; bij het raadplegen van menigen schrijver, op de Schrift bij uitnemendheid het oog gevestigd te houden; en op het gebied der wetenschap, met erkentelijk gebruik der aan anderen verleende wijsheid, de uitspraak van den Zaligmaker gedachtig te zijn: één is uw Meester; namelijk God."

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Rotterdam, augustus 2004

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Publications

This thesis is based on the following publications:

Chapter 2 Dijkgraaf, E. and R.H.J.M. Gradus (forthcoming), Cost savings in unit-based pricing of household waste, Resource and Energy Economics.

Chapter 3 Dijkgraaf, E. and R.H.J.M. Gradus (2003), Cost savings of contracting out refuse collection, Empirica 30(2), 149-161.

Chapter 4 Dijkgraaf, E., R.H.J.M. Gradus and B. Melenberg (2003), Contracting out refuse collection, Empirical Economics, 28, 553-570.

Chapter 5 Dijkgraaf, E. and H.R.J. Vollebergh (forthcoming), Burn or bury? A social cost comparison of final waste disposal methods, Ecological Economics.

Chapter 1

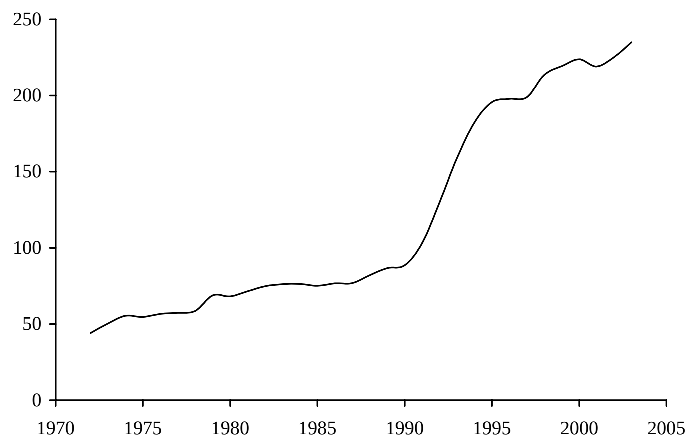
Introduction

1.1 Introduction

The regulation of the Dutch waste market changed considerable in the last thirty years. Where at first waste was simply an issue left to the producer, the environmental consequences of the rise in the amount of waste combined with a lack of waste collection initiated local governments to stimulate a proper infrastructure. Livability and health were improved in this way. For the same reasons governments started to regulate the waste disposal market when it became clear that landfilling and incineration had large environmental and health effects. For example, landfills and incineration plants nowadays have to comply with tight emission limits while the use of landfilling is discouraged to the advantage of incineration as it is believed that from an environmental point of view this option performs better.

The changes in waste management policy resulted in an acceleration of collection and treatment costs. In 1972 an average Dutch household paid 44 euro per year (in prices of 2003) for the collection and treatment of waste. In 1990 the real costs were already two times as high, while in 2003 an average household paid more than 5 times as much (see figure 1.1). Between 1972 and 2003 the amount of waste rose with 44%. Thus, costs per ton collected waste increased by 270%. This sharp rise is not only a consequence of the introduction of sharp emission limits, but also of the increased use of more expensive treatment options. Furthermore, regulation influenced competition in such a way that monopolistic behaviour could result in higher prices. In 2000 the waste treatment sector

Figure 1.1: Costs waste collection and treatment households (real euro per year)



had a turnover of 4.6 billion euro (AOO, 2003). The waste treatment sector has not only a considerable size when measured in private costs. The environmental effects are, notwithstanding the much tighter regulation, still large. Compared with a total level of national environmental expenditures of 11 billion euro, waste treatment is responsible for approximately 40% of this amount.

The sharp rise in waste collection and treatment costs and the high level of environmental costs asks for an evaluation of the relation between total costs and the chosen regulation instruments. This thesis, thus, wonders whether the rise in costs is a necessary consequence of the chosen policy goals or whether other instruments are available that perform better. This theme is motivated by theoretical and empirical evidence. Alternative instruments are not only suggested by the economic literature, but also by the differences in choices made in practice. As the types of regulation governments chose differ a lot, both within countries and between countries, apparently decision makers have different opinions about the optimal regulation package. Where some countries still landfill the main part of waste produced, other countries stimulate heavily waste incineration. Furthermore, differences exist with respect to the way waste collection is organized (through the market or not), the payment scheme of waste collection (fixed or dependent on the quantity of waste) and the regional dimension of the waste disposal market (local, regional or international). From an economic perspective policy options should be chosen that minimizes total costs. In this thesis a number of case studies are presented that may help to guide such a choice.

In this chapter we first present a short outline of the Dutch waste market. Second, we describe the main questions analysed in this thesis. Finally, the contents of the different chapters is presented.

Table 1.1: Overview history waste market regulation

Year	Event
1976	First Dutch waste law: provincial selfsufficiency for waste disposal
1979	Waste preference accepted in Dutch parliament
1985	First Dutch directive regulating emissions waste incineration plants
1989	Second Dutch directive regulating emissions waste incineration plants
1990	Foundation Dutch AOO: regional selfsufficiency for waste disposal
1991	EU framework directive (European selfsufficiency)
1993	Dutch directive regulating emissions of landfills
1993	First Dutch municipality uses unit-based pricing based on frequency
1993	Six Dutch municipalities have already system based on bags
1994	Dutch obligation to collect organic waste separately
1994	First Dutch municipality uses unit-based pricing based on weight
1996	Introduction of Dutch landfill ban
1996	Dutch landfill tax introduced (13 euro)
1998	Higher Dutch landfill tax (29 euro)
1999	EU-directive regulating emissions landfills
2000	National selfsufficiency in the Netherlands
2000	Higher Dutch landfill tax (64 euro)
2000	EU-directive incineration
2002	Higher Dutch landfill tax (75 euro)
2003	Moratorium on incineration capacity in the Netherlands abolished
2005?	Export of Dutch waste to foreign incineration plants allowed

1.2 The Dutch waste market

The Dutch waste market is nowadays a complex market. Whereas forty years ago the waste households and firms produced was simply landfilled or burned in a open furnace, now a number of high tech treatment options is used. Currently, nearly 80% of Dutch waste is recycled, 10% is incinerated and only 9% is landfilled. The Dutch government aims even at a 0%-target for landfilling of burnable waste for the coming years.[1] This follows from an explicitly defined preference order of waste treatment options. The Dutch government decided at the end of the seventies of the twentieth century that (i) waste should be prevented when possible, (ii) when prevention is not possible recycling should be maximized, (iii) when recycling is not possible incineration with energy

[1] Burnable waste is the waste that has characteristics that make incineration possible. Of course no 0%-target for landfilling exists for unburnable waste.

recovery is the preferred option and (iv) only when these options are exhausted landfilling should be used. As the costs of the treatment options are in many cases not in accordance with this preference order regulation instruments were used to promote the implementation of this policy goal.

These instruments had two dimensions. First, they were aimed at specific parts of the preference order. Second, they had a regional component. As the combination of these two dimensions shaped the Dutch waste market, they are described in more detail in the next two paragraphs. Finally, the last paragraph describes the possible consequences of the specific characteristics for the cost level of the waste market and describes whether changes in regulation would be possible to diminish these costs.

1.2.1 Preference order regulation

The implementation of the waste preference order initiated two main changes in waste management policy: (i) regulation was initiated to reduce the amount of waste and (ii) regulation was changed to promote incineration above landfilling.

Firstly, a number of municipalities started to experiment with better incentives to reduce the amount of household waste. A first step was the promotion of separate collection to increase recycling. While in 1989 vegetable, fruit and garden waste was only separately collected by 1% of Dutch households, in 1995 already 95% had a special collection can for this type of waste. Since 1994 municipalities are obliged to supply an infrastructure for the separate collection of compostable waste. The separate collection of glass, paper, textiles and hazardous waste increased also significantly and municipalities have to provide collection opportunities that can be easily accessed by citizens. A second step, some municipalities took during the last 15 years was the introduction of unit-based pricing systems. In these system citizens pay per unit of collected waste, whereas formerly they paid only a fixed rate per year. The introduction of a unit related price is thought to give citizens incentives to reduce the amount of waste resulting in lower collection and treatment costs. However, although a growing number of municipalities use unit-based pricing systems, in 2000 more than 75% still financed waste collection by a fixed fee.

Secondly, the preference order asked for a change in waste management policy related to landfilling. As landfilling was the cheapest available treatment option, no financial incentives existed to incinerate waste. At the end of the seventies 11 waste incineration plants existed in the Netherlands. They were build by local governments that not only owned the waste incineration plant, but also the waste needed to fill the plant as they used the plant primarily for the waste generated by their citizens. Other municipalities however, decided not to invest large amounts of money in waste incineration plants and still landfilled their waste.[2] Furthermore, the waste of firms primarily went to cheap landfill sites. During the eighties and nineties the financial incentive to use landfilling even increased. High concentration of dioxines, furanes and other harmful emissions in the smoke of waste incineration plants resulted in tight environmental regulation. In 1989 specific limits were given for emissions like dioxines and furanes.[3] Waste incineration plants had to investment significant sums in technologies to clean the smoke. As a result the tariffs of waste incineration plants rose in the beginning of the nineties with nearly 40%.[4] Therefore a landfill tax was introduced in 1996 to give all waste suppliers an incentive to seek for better options than landfilling.[5] Starting with a level of 13 euro per ton, the tax reached a level of 75 euro per ton of waste in 2002. This made incineration for all waste suppliers a cheaper treatment option than landfilling. Currently, the amount of waste incinerated is indeed significantly higher than 10 years ago. Since 1992 the waste incineration capacity was increased with 90% (see figure 1.2).

1.2.2 Regional regulation

For the waste treatment market these changes in waste management policy took place in a market environment that can be initially characterized as monopolistic for both waste treatment and collection.[6]

[2] Note that municipalities had the obligation to incinerate the waste since 1996 but had no obligation to invest in incineration plants. The lack of national incineration capacity made landfilling still possible.

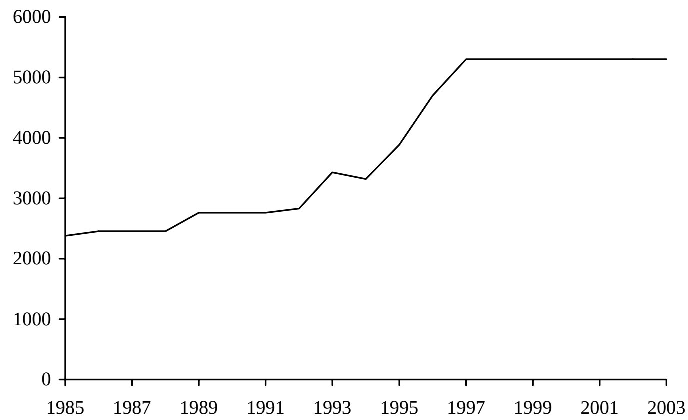
[3] The EU-regulation that came in force in 2000 (COM(00)76) formulated emissions limits that were comparable to the 1989 Dutch limits.

[4] Although tighter regulation was also set up for landfill sites in 1993, the resulting cost increase was far less than for incineration plants. The EU-regulation of 1999 (COM(99)31) confirmed the tight Dutch regulation for landfills.

[5] In 1996 also a landfill ban for burnable waste came in force. However, it showed that this ban was not effective as (necessarily) a lot of exemptions had to be given due to undercapacity of incineration.

[6] The market for recycling, which falls outside the scope of this thesis, is since long an international market with a lot of competition.

Figure 1.2: Capacity waste incineration plants in kton per year



The waste treatment market

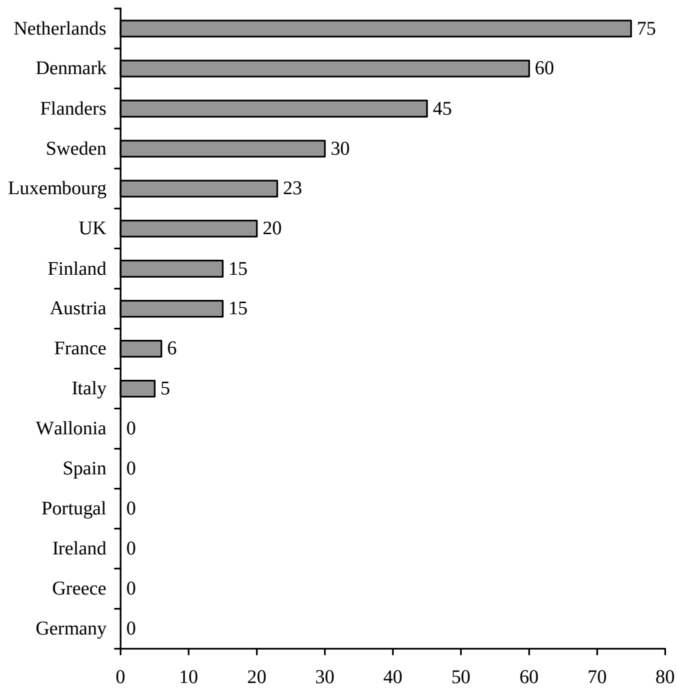
Till 2000 the Dutch waste policy for the waste treatment market was based on regional selfsufficiency. The capacity of waste incineration plants and landfill sites was planned by the AOO, the Dutch Waste Management Council. In this council the national, provincial and local governments decided together which investments should take place in which region. Transport of waste between the four waste regions was prohibited and entry of new firms was impossible due to a moratorium to build more incineration capacity than planned.

In 1996 a national committee concluded that these characteristics resulted in monopolistic behaviour of waste treatment plants. Per region only a few incineration plants and landfill sites existed. Therefore, waste collection firms had nearly no choice in contracting treatment capacity. The lack of competition could result in treatment tariffs that were higher than necessary. The committee suggested to stimulate competition by creating a national market. In 2000 a allowance was given to transport waste across regional borders. International transport for final waste treatment (incineration and landfilling) was still prohibited as national selfsufficiency remained an important policy goal.[7] However, in spite of this prohibition the pressure on waste incineration plants and landfill sites increased further as a result of the growing incentives to export waste.

[7] EU-regulation (COM(91)) defines selfsufficiency at the EU-level, but gives memberstates the possibility to remain selfsufficient at the national level.

For waste suppliers a contract with foreign waste treatment firms was very attractive as these countries did not have a (comparable) landfill tax (see figure 1.3). As a result waste treatment was cheaper in surrounding countries.[8]

Figure 1.3: Landfill tax in EU-countries 2002 (euro per ton)



To challenge the prohibition lawsuits were started by waste collection firms. Meanwhile, the export of waste increased already as collection firms succeeded to contract foreign separation firms. As long as more than 50% of the waste was reused after separation the Dutch prohibition for export was overruled by an European directive that stated that for reuse no limitations were allowed. Opponents claimed however that waste collection firms abused the European directive as in fact frequently less than 50% of the waste was reused. The problem was that the standard of 50% could only be tested on a yearly basis. The yearly figures of a separation firm were used to decide whether this firm reused on average more than 50% of the incoming waste. This made it possible that for an individual Dutch waste stream far less than 50% was reused in practice as long

[8] Note that for recycling other market circumstances exist. For example, the Netherlands imported 1.3 Mton paper (mainly from Germany) in 2002.

as this was compensated by other waste streams with a higher percentage reuse than 50%. Using this route Dutch waste could in fact be landfilled very cheaply in other countries while the high Dutch landfill tax was evaded.

Summarizing, the goal of Dutch government to be selfsufficient for waste treatment was challenged by the differences in regulation between the Netherlands and surrounding countries. In 2002 the Dutch government stated that in the near future export and import of waste for incineration will be allowed when differences in regulation are changed towards a level playing field. Anticipating this change the moratorium on new capacity for incineration plants was abolished in 2003.

The waste collection market

An important part of the waste collection market can also be characterized as monopolistic. Dutch municipalities have an obligation to provide a waste collection infrastructure for municipal waste. They can choose to provide this infrastructure themselves or to contract out waste collection. As most municipalities choose for the first option, a lack of competition characterizes the market. Furthermore, municipalities are also free to choose the way citizens pay for waste collection. As autonomy was given to municipalities to decide on the chosen pay scheme, it was possible that some municipalities chose for unit-based pricing while other municipalities used a fixed fee to cover the waste collection costs.

1.2.3 Changes in regulation

The changes in waste management policy resulted in an acceleration of collection and treatment costs (see paragraph 1.1). The sharp rise in waste collection and treatment costs asks for an evaluation of the relation between total costs and the chosen regulation instruments. Is the rise in costs a necessary consequence of the chosen policy goals or are other instruments available that perform better? Based on theoretical and empirical evidence the necessity of the risen costs can be questioned. A number of studies are available that indicate that alternative instruments are available. The following examples might illustrate this:

1. Jenkins (1993) proves both on theoretical and empirical grounds that a user fee performs better than a fixed fee as it gives households an incentive to diminish

the amount of waste collected by both more prevention and a higher level of recycling. She concludes that unit-based pricing would save a significant part of municipal waste collection costs and that it is better (page 129):

> "to stop financing residential solid waste services by flat fees or through property taxes and to instead charge user fees."

2. Domberger and Jensen (1997) show in an overview article that contracting out suggests cost savings in order of twenty percent without sacrificing the quality of service provided for a number of government services. This suggests that using contracting out for waste collection might save a significant part of municipal waste collection costs.

3. Ley et al. (2002) conclude that restricting competition to regional waste markets results in higher costs than necessary, Allowing for interstate trade in the US saves on waste treatment costs according to their model.

Although these examples indicate that the current regulation regime for the Dutch market results in higher costs than necessary, the literature is not decisive as a number of caveats introduce uncertainty whether this is indeed the case. The main questions left open in the literature and analysed in this thesis are:

1. Are the results for instruments based on the analysis of waste markets in other countries representative for the Netherlands? As countries differ with respect to policy goals, preferences of citizens, specific market circumstances and country characteristics this question can only be answered on the basis of Dutch data.

   (a) The conclusion of Ley et al. (2002) might be dependent on the low level of population density in the United States. It is possible that restrictions on inter regional trade have different effects in the Netherlands as within regions more waste firms are present in this country.

   (b) The fact that the Netherlands is a small country surrounded by countries with regulation regimes that are totally different might also effect the consequences of inter regional competition. Competition might have adverse effects when a unlevel playing field exists with respect to regulation.

   (c) The small distances between Dutch municipalities might both influence the attractiveness of contracting out and the unwanted side-effects of unit-based pricing as the first makes it possible to enhance economies of scale and the second stimulates illegal dumping.

2. Is unit-based pricing still the preferred instrument when illegal dumping and administrative costs are included in the analysis? Jenkins (1993) shows that illegal dumping and high administrative costs might lead to a preference for flat fees although for the United States the positive effects outweigh the negative effects. It is, however, questionable whether this is also the case for the Netherlands.

3. Is it possible to be more precise on the effects of different types of unit-based pricing? The results presented in the literature do not make it possible to compare the available types of unit-based pricing, while in the Netherlands five different types are used. As these instruments may differ with respect to their effects on waste quantities and also have different levels of administrative costs, a direct comparison between instruments might enhance the choice for the best available pricing scheme.

4. Is it possible to explain the fact that municipalities make less use of contracting out than suggested by the lower cost level of this instrument? Maybe specific municipal characteristics might explain that a large number of municipalities do not use this instrument.

5. Can the preference for incineration above landfilling be explained on the basis of a proper comparison of costs and benefits or both options?

This thesis tries to fill these caveats to make a more thorough evaluation of the current regulation possible and to suggest instruments that might save costs for waste collection and treatment.

1.3 Central questions

The central question in this thesis is:

> Can changes in regulation help to decrease total social costs of waste collection and treatment?

Total social costs of waste collection and treatment is defined as the sum of private and external costs. Private costs are equal to the necessary production costs. Social costs can be higher than private costs when external costs exist. External costs are costs related to production, but which are not paid for in the market. For example, the price paid for waste incineration is based on the private costs of capital, labor and other inputs, but not on the amount of emitted carbon dioxide or dioxins. As these emissions impose in fact a cost burden on society when they increase for example the greenhouse effect or the occurrence of cancer, a proper economic analysis is not only based on private costs, but also on external costs.

Mathematically total social costs are equal to:

$$TSC = W\left(C_C^P + C_C^E + C_T^P + C_T^E\right), \tag{1.3.1}$$

with TSC total social costs (the sum of private and external costs), W the total quantity of waste collected, C_C^P the private cost of waste collection (excluding treatment costs), C_C^E the external cost of waste collection, C_T^P the private costs of waste treatment and C_T^E the external costs of waste treatment.

The central question of this thesis is whether changes in regulation can influence the different parts of equation 1.3.1 such that total social cost is minimized. Each of the following chapters analyses a part of equation 1.3.1. Four specific questions are dealt with:

1. Is it possible to decrease the quantity of waste collected (W) by making use of unit-based pricing systems and what are the consequences for the collection costs related to administrative costs (C_C^P) and illegal dumping (C_C^E) (chapter 2)?

2. Given the amount of collected waste (W), is it possible to decrease total private collection costs (C_C^P) by contracting out the waste collection of municipalities (chapter 3 and 4)?

3. Given the amount of collected waste, is it possible to decrease total treatment costs ($C_T^P + C_T^E$) by changing the preference for incineration above landfilling (chapter 5)?

4. Given the amount of collected waste (W) and the preferred treatment options, is it possible to decrease total private treatment costs (C_T^P) by increasing international competition (chapter 6)?

The **first** question is about the effects of unit-based pricing. This instrument tries to introduce the right incentive for citizens to decide on the optimal level of waste supplied to waste collectors. Whether this indicates that total social costs are also lower when unit-based pricing is introduced is not only dependent on the effect on total waste, but also on unwanted side-effects (like illegal dumping and bringing the waste to neighbouring municipalities without unit-based pricing) and the height of administrative costs.

The **second** question deals with the effects of contracting out waste collection. Contracting out is seen as an instrument to stimulate the production efficiency of the collection market. Interestingly, important differences in the use of these instruments exist between municipalities.[9] Only a minority of municipalities use unit-based pricing and contracting out as instruments to reduce waste collection and treatment costs. Though evidence presented in the international literature indicates that these instruments are indeed effective in lowering total waste collected (W), evidence on administrative costs (C_C^P) and illegal dumping (C_C^E) is more mixed. Furthermore, the fact that the majority of the municipalities do not use these instruments might indicate that contracting out and unit-based pricing systems are not cost minimizing instruments in the Netherlands. As general experience with the evaluation of regulation instruments does show that international evidence might be misleading, we analyse whether this evidence is representative for the Netherlands.

The **third** question concerns the choice between waste incineration and landfilling. This question is analysed for a number of reasons:

- The first reason why this choice is evaluated is that the preference for incineration above landfilling leads to a significant rise in treatment costs (C_T^P) while it is unsure whether this rise in costs is compensated by a high enough decrease in external costs (C_T^E).
- The second reason is that, although the preference for incineration was at the time motivated by environmental concerns regarding landfilling, no explicit comparison was made between the two options on the basis of total social costs.
- The third reason is that, although most developed countries have adopted a hierarchical approach preferring incineration above landfilling, these countries do not

[9] The waste collection of firm waste is not analysed as collection of waste for firms is generally done by contracting out, while firms pay per unit waste collected.

show equal behaviour in implementation policies. Where some countries heavily invest in WTE-facilities, other countries still rely on landfilling. One reason for this gap between official policy statement and implementation might be the far higher private costs of incineration.

- The fourth reason is that a number of environmental platforms heavily oppose WTE-facilities, mainly on the basis of their emission patterns. Although they emphasize the use of prevention and recycling, they see landfilling not as a worse alternative for incineration. This may be due to the improvement of landfill technology. Where leakage and emissions of methane were a major problem related to 'old' technology, nowadays sound layers, leakage collection systems and recycling of methane is possible.
- The fifth reason to focus on the choice between incineration and landfilling is that, as a result of differences in the policies to implement the preference for incineration, there is an unlevel playing field between the Netherlands and other surrounding countries. The landfill tax needed to implement the preference for incineration is far higher than the tax levied in surrounding countries. This implies that the effect may be that (using the separation plants) Dutch waste is landfilled in other countries. Therefore, it is questionable whether the Dutch policy is maintainable in an international market.

For all this reasons a proper comparison on the basis of social costs between landfilling and incineration is essential.

The **fourth** question regards the possibilities to use international competition as an instrument to decrease monopolistic behaviour and thus to lower private treatment costs (C_T^P). The reason why this question is analysed is that a growing debate is going on about the pros and cons of international competition. In the literature nearly no evidence is available about the consequences when international competition is introduced in the waste market. Analysing the influence of international competition on waste treatment costs shows what the possibilities are to decrease total costs. Furthermore, it also makes clear what the relations are between national regulation and international competition. Insight in these relations is necessary to answer the maintainability question of the Dutch policy.

For most questions a **dynamic approach** is essential as changes in regulation might take a number of years before the final effects are measurable. If for instance the tax on landfilling is changed, the effect on the short term might be different to the long term effect as only on long term the waste treatment capacity can be changed. A second example is the difference in time related emission patterns of landfilling versus incineration. Where the emissions of incineration are largely concentrated in the time period the waste is treated, emissions of landfilling are long lasting.

The four questions cover the most important determinants of total social waste collection and treatment costs that can be influenced by changing regulation. This means that **not all aspects** of the waste market are dealt with. In general, we focus on the determinants of total social costs for which it can be expected that changes in national regulation will change these costs significantly. The prior is that this may be the case when regulation exists that hinders the minimization of social costs. In fact, the chosen most important determinants of social costs that can be changed by regulation is (except for the preference order between landfilling and incineration) based on Dijkgraaf et al. (1999). They showed that the markets for prevention, recycling and collection of firm waste are not significantly hindered by national regulation. For example, the role of regulating prevention and recycling is not dealt with as the market for prevention and recycling (according to EU-regulation) is already international oriented. Finally, the role of regulating the hazardous waste market is not part of this thesis.

1.4 Contents

In this paragraph we introduce the following chapters. **Chapter 2** estimates the effects of four different unit-based pricing systems of household waste collection (weight-based, bag-based, frequency-based and volume-based) on the quantity of waste collected using a panel data-set for Dutch municipalities. This is done not only for total collected waste, but also for its components: unsorted waste, glass, paper and vegetables, fruit and garden waste. Thus, both total effects and substitution effects are estimated. The literature is extended in three directions. Firstly, explicitly distinguishing between the different systems of unit-based pricing (weight-based, bag-based, frequency-based and volume-based pricing) contributes to the literature because no study presents a direct comparison of the possible unit-based pricing systems. Secondly, the question is analysed whether the influence of environmental activism is responsible for part of the estimated price effect. Thirdly, we test whether citizens in municipalities with a unit-based pricing system supply their waste (illegally) in surrounding municipalities without unit-based pricing systems. Finally, we investigate the height of administrative costs.

Chapter 3 estimates the cost difference between municipalities that contract out their waste collection and municipalities that provide this service themselves using a cross-section database of Dutch waste collection firms. The literature is extended in three directions. Firstly, by estimating the influence of contracting out on collection costs the general result found in the literature that contracting out results in lower collection costs is tested for the Netherlands. Secondly, we extend the literature by explicitly checking for the validity of pooling with respect to institutional choice and municipality size. Where the previous literature usually assumes that the same cost function applies to outside and inside firms and to municipalities of different sizes, the estimated effects of contracting out might depend on this assumption. Thirdly, compared with previous studies more emphasis is put on the fiscal system. Due to the Dutch fiscal system there is a disincentive for contracting out. Even though we can estimate significant cost savings when waste collection is contracted out, households will not experience these cost savings on a one to one basis. In the Netherlands private collection firms have to pay VAT while public firms are exempt. Countries such as the United Kingdom and Denmark have a compensating system, in that local authorities are tax-neutral towards contracting outside or inside. Thus, the current fiscal system in the Netherlands renounces the role for private collection firms.

Chapter 4 starts where chapter 3 ends. As contracting out the collection of municipal waste leads to lower costs, one would expect that municipalities favor private collection. Indeed, 40% of the Dutch municipalities chose for the option to collect waste by a private firm. The question arises why the other 60% did not choose this option as well. Chapter 4 examines for the Netherlands the determinants of the provision mode of refuse collection based on data for almost all Dutch municipalities. The prior is that contracting out is not only dependent on the achievable efficiency gain, but also on other factors. Firstly, the relation between the lack of contracting out and the specific goals politicians want to achieve is analysed. Furthermore the wealth of the local government as a ground for contracting out, but also the relation between the possible efficiency gain and the size of the municipality may play a role. The results found confirm the international literature. Moreover, the existing literature is extended by investigating more general (semi-parametric) specifications.

Chapter 5 extents the literature by presenting an encompassing empirical analysis of the entire final waste disposal system including the indirect effects of their recovery functions from a social cost perspective. We have data describing a reasonable set of (technical) available options for each disposal method, as well as on their associated private costs and cost performance in terms of environmental externalities and energy and material recovery. We present the results of a comprehensive data set on the average social cost of two "best practice technologies" for incineration and landfilling. The data are taken entirely from the Netherlands and reflect (partly revealed) cost estimates of technologies that comply with the strictest waste disposal regulation of the world. Moreover, environmental conditions for final waste disposal are relatively difficult because the Netherlands is not only densely populated, like Singapore and Japan and some areas in the US, but also faces pretty bad soil conditions for landfills. If somewhere one would expect Dutch WTE-plants to signal lower social cost compared with landfilling.

Chapter 6 analyses the relations between the Dutch waste market and the waste markets of other EU-member states. As Dutch regulation differs significantly from surrounding countries international competition might lead to adverse effects. It is argued that it is questionable whether the movements of the Dutch waste market towards international competition is compatible with the specific goals (self sufficiency for waste disposal and a preference for incineration above landfilling) the Dutch governments want to achieve. Furthermore, it is unclear what the relation is between these goals and the development of waste treatment costs. The central question of chapter 6 is which regulation package minimizes total costs and at the same time makes it possible to achieve the goals set for the waste market. As nearly no literature exists to guide policy makers on this issue, a simulation model is presented with which the influence of changes in regulation and degree of (inter)national competition can be estimated on total costs. This model includes all EU-memberstates, the supply of waste of both households and firms, the demand of waste by all relevant waste treatment firms (landfilling, separation and incineration in waste incineration plants, coal-fired electricity plants and cement kilns), the transport costs between regions and the different regulation instruments. The model is calibrated using actual data of all EU-waste markets and deals also with long term effects by making entry of new waste treatment firms possible. With the model we analyse whether it is possible to implement national regulation instruments or whether harmonization between countries is necessary to achieve cost minimization. As one of the goals the Dutch government has set does not corresponds with the outcome of a social cost benefit analysis (see chapter 5), it is interesting to simulate what happens when the preference order between the different treatment options is set according to this analysis. Setting taxes for the treatment options in the model equal to their external costs, this makes clear what the costs are of this specific goal the government has chosen, both in terms of financial costs as in terms of possibilities to implement national regulation.

Chapter 7 presents the conclusions and summary of this thesis.

Chapter 2

Cost savings in unit-based pricing of household waste

2.1 Introduction

More and more Dutch communities have implemented unit-based user fees to finance waste collection. These user fees require households to pay for each kilogram, bag or can presented at the curb for collection. By 2000, approximately 20% of all Dutch municipalities had implemented such a system. In this chapter, we estimate household reactions to the implementation of unit-based pricing for the collection of residential waste. Our estimates show significant and sizable price effects, which depend on the type of unit-based pricing.

Two streams of literature that estimate household reactions to the implementation of unit-based pricing systems can be distinguished. The first uses cross-sectional analyses of municipalities and the second applies household survey data. Most of the studies show considerable impacts from a pricing system.Table 2.1 summarizes the existing econometric literature with respect to the effects of unit-based pricing. In general, nearly all studies find a negative and significant own-price effect from unit-based pricing. The results are more mixed for the cross-price effect on collected recyclable waste.

Most studies evaluate bag- or volume-based systems. Only Linderhof et al. (2001) study the effects of the most refined, weight-based system. Table 2.1 indicates that own-price elasticities overlap for the different unit-based pricing systems. For example, Strathman et al. (1995) found an elasticity of -0.45 for the volume-based system, which is higher than the elasticities of the bag-based systems, while Hong et al. (1993) found a non-significant elasticity.

Direct comparison of systems is limited to Van Houtven and Morris (1999). This paper compares the effects of bag- and volume-based systems and finds a significantly higher elasticity for the bag-based system for curbside-collected unsorted waste. The effect on the quantity of waste recycled is found to be insignificant in both cases.

Table 2.1: Overview of the econometric literature on unit-based pricing

				Elasticities	
Study	Country	System	Price[a]	Own[b]	Cross[c]
Household surveys					
Hong et al. (1993)	USA	Volume	3.63	not sig.	>0
Van Houtven et al. (1999)	USA	Volume	0.22	-0.10	not sig.
Jenkins et al. (2003)	USA	Volume[d]	2.13		not sig.
Reschovsky et al. (1994)	USA	Bag (recycling)	0.85		not sig.
Reschovsky et al. (1994)	USA	Bag (compost)	0.85		>0
Fullerton et al. (1996)	USA	Bag	0.89	-0.08	0.07
Van Houtven et al. (1999)	USA	Bag	0.86	-0.26	not sig.
Hong (1999)	Korea	Bag	1.49	-0.15	0.46
Linderhof et al. (2001)	Netherlands	Weight (compost)[e]	3.86	-1.39	
Linderhof et al. (2001)	NL	Weight (unsorted)[e]	4.14	-0.34	
Aggregate municipality data					
Wertz (1976)	USA	Volume	5.85	-0.15	
Jenkins (1993)	USA	Volume	1.46	-0.12	
Strathman et al. (1995)	USA	Volume	5.69	-0.45	
Van Houtven et al. (1999)[f]	USA	Volume	0.22	<0	
Kinnaman et al. (1997)	USA	Bag	0.16	-0.19	0.23
Podolsky et al. (1998)	USA	Bag	3.62	-0.39	
Van Houtven et al. (1999)[f]	USA	Bag	0.86	-0.15	
Kinnaman et al. (2000)	USA	Bag	0.09	<0	not sig.
Callan et al. (1997)	USA	Mixed[g]	n.a.		0.07

[a] Average tariff in real US dollars (2000) per 30 gallons (114 liters) of unsorted waste.
[b] Elasticity of the amount of collected unsorted waste with respect to the price of unsorted waste collected at the curbside.
[c] Elasticity of the amount of collected recyclable (and/or compostable) waste with respect to the price of unsorted waste collected at the curbside.
[d] Of the 1,049 households, 116 face a positive unit price, of which 104 subscribe to collection of a pre-specified number of cans and 12 pay per bag/tag/sticker.
[e] In Oostzaan, the city Linderhof et al. (2001) study, both compostable and unsorted waste are priced on a weight basis.
[f] Data are aggregated per sanitation route.
[g] In Massachusetts, different unit-based pricing systems exist (bag, tag, volume). This study does not discriminate between the different programs.

We extend the literature in three directions. Firstly, we explicitly distinguish between the different systems of unit-based pricing (weight-based, bag-based, frequency-based and volume-based pricing). This contributes to the literature because no study presents a direct comparison of the possible unit-based pricing systems. Our results clearly indicate that the bag- and weight-based systems perform far better than the other systems. Secondly, we investigate whether environmental activism is responsible for part of the estimated price effect. Our research shows that municipalities that introduce a unit-based pricing system already produce less waste on average before its introduction. When no correction is made for this effect, price effects estimated on the basis of cross-section data might overestimate the true effects. Thirdly, we test whether surrounding municipalities without unit-based pricing systems in fact collect part of the waste produced in municipalities with unit-based pricing systems. No such effect seems to be present in Dutch municipalities.

2.2 Effects of unit-based pricing

2.2.1 Method and data

In previous studies using cross-sections of municipalities, waste per capita is a function of price, the municipality's mean level of income, the share of homeowners, the age distribution, the average number of people in a household and other demographic variables (see, for example, Fullerton and Kinnaman, 1996).

We use the quantity of waste collected (in kilograms per inhabitant) also as the dependent variable. However, we are able to discriminate between different waste streams. In the Netherlands, municipalities are obliged to collect three types of waste separately: compostable waste such as vegetable, food and garden waste; recyclable waste such as glass, paper and textiles; and unsorted waste. Furthermore, municipalities are obliged to collect compostable and unsorted waste at the curbside. For recyclable waste, municipalities can choose whether they collect at the curbside or provide drop-off centers.[1] For municipalities without curbside collection of recyclable waste, the number and location of drop-off centers must be such that the collection infrastructure is easily

[1] In some municipalities, there is a free curbside collection program for recyclable paper organized by local associations, such as sports clubs and schools. Our data include the waste collected by these associations.

accessible for all citizens. For example, municipalities place collection units at shopping centers and at entrance roads of neighborhoods.

Table 2.2: Descriptive statistics

	Mean	Max.	Min.	St. dev.	Observat.	Municipalities
$Waste_{total}$	431	707	222	62	1323	507
$Waste_{unsorted}$	218	450	52	54	1451	530
$Waste_{compostable}$	117	239	12	39	1449	529
$Waste_{recyclable}$	99	217	19	20	1334	508
UBP_{weight}	0.02	1	0	0.14	1451	530
$UBP_{bagunscom}$	0.01	1	0	0.11	1451	530
UBP_{baguns}	0.02	1	0	0.15	1451	530
UBP_{fre}	0.07	1	0	0.26	1451	530
UBP_{vol}	0.05	1	0	0.22	1451	530
UBP_{oth}	0.01	1	0	0.12	1451	530
Retire	13.31	27.77	6.38	2.90	1451	530
Fam size	2.56	3.70	1.72	0.20	1451	530
Foreigner	0.04	0.31	0	0.04	1451	530
City	0.05	1	0	0.22	1451	530
Village	0.57	1	0	0.50	1451	530
Density	0.50	27.46	0.02	1.35	1451	530
Ownhouse	10.05	30.59	1.34	3.12	1451	530
Ownflat	1.68	16.53	0	2.20	1451	530
Income	39.04	44.60	28.5	2.34	1451	530

Data on the dependent variables - the quantities collected of total, unsorted, recyclable and compostable waste in kilograms per inhabitant - come from studies by the Dutch Waste Management Council (AOO). Total waste collected is calculated as the sum of unsorted, recyclable and compostable waste. The AOO-studies present data on the quantities of paper, glass, textiles, compostable and unsorted waste collected for 1998, 1999 and 2000. The AOO uses an annual inquiry from the CBS (the Dutch Central Bureau for Statistics), which is sent to the waste collection units of all Dutch municipalities. These units have reliable figures for the quantity of waste collected as the bill they have to pay is based on the quantity of waste supplied to waste treatment firms. These firms weigh the waste each time a collection vehicle brings waste to the treatment plant. The CBS checks the quality of the data by comparison with other years and by comparison with additional information from waste treatment companies.[2] Additionally, as the data for sorted waste are partly collected by schools and charitable organizations, information from regional and national representative organizations for glass, paper and textiles recycling is used to check these data. The response rate of

[2] In the inquiry, municipalities are asked which companies treat the waste. Information from these companies is gathered to make comparison possible.

the inquiry is 91%. Thus, our data-set comprises nearly all Dutch municipalities. The actual number of municipalities included differs for each dependent variable due to data availability. The first four rows in Table 2.2 present summary and availability statistics for the dependent variables (see the Appendix for the variable definitions).[3]

Dutch municipalities are free to choose the financing mechanism for waste collection. Most municipalities finance waste collection by a flat rate (see Table 2.3). This results in a marginal price of zero. In order to promote waste prevention and recycling, a number of municipalities have introduced a unit-based pricing system. In general, the Dutch unit-based pricing systems generate marginal prices for unsorted and compostable waste, while the collection of recyclable waste (glass, paper and textiles) is still free. This gives citizens the incentive to sort their waste and to change their buying behavior. Different Dutch municipalities have introduced different types of unit-based pricing systems. These systems can be ordered with respect to the refinement of the pricing system. It could be expected on theoretical grounds that as marginal pricing becomes more and more refined, households respond with greater reductions in priced waste streams and a growing supply of unpriced waste streams.

Table 2.3: Occurrence of unit-based pricing systems

	1998	1999	2000
Municipalities with unit-based pricing systems			
- Weight-based system	9	10	13
- Bag-based system for unsorted and compostable waste	6	6	6
- Bag-based system for only unsorted waste	13	12	14
- Frequency-based system	19	43	54
- Volume-based system	24	30	29
- Unspecified type of system	6	8	10
Total	77	109	126
Municipalities without unit-based pricing systems	461	429	412
Total	538	538	538

In general, four different systems are present: volume-based, frequency-based, bag-based and weight-based.[4] Table 2.3 gives an overview of the pricing systems used by Dutch municipalities in the period 1998-2000 based on the annual AOO inquiry.

[3]As not for all municipalities data are available for all years, the number of observations is not exactly equal to the number of years multiplied by the number of cross-sections.

[4]Some municipalities have a combination of the different unit-based pricing systems or apply the pricing system to only part of their municipality. These are included in Table 2.3 as 'unspecified type of system'.

The volume-based program allows households to choose between different volumes of collection can. Most municipalities supply a standard can with a volume of 140 liters (37 gallons), with the possibility of upgrading to a 240-liter (63 gallon) can or of subscribing to more 140-liter cans. In general, citizens can choose different volumes for unsorted and compostable waste. The marginal price in the volume-based system is rather crude, as the decision on the optimal level of waste supply can only be made at the beginning of the contract period and at certain review times (usually annual). In 2000, 29 municipalities in the Netherlands used a volume-based pricing system.

A more refined marginal price results from a frequency-based system, in which the household pays for the number of times the can is presented at the curbside. The payment is not dependent on the actual amount of waste the can contains. Whether the can is filled or half empty, the bill household receive is just equal to the number of times the can is presented. The occurrence of frequency-based pricing systems shows a notable rise between 1998 and 2000. In 2000, this type of system was the most frequently used pricing system.

In the bag-based system, households have to buy a special bag with specific marks. In most cases, these bags can be bought at supermarkets, petrol stations and the town hall. Other bags without the relevant marks are not collected. The bag-based system is a more refined pricing system than the frequency-based system, as the volume of the bag is significantly less than that of the can. In the Netherlands, the volume of bags is 50 or 60 liters (13 or 16 gallons). An important difference compared with other unit-based pricing systems is that the most frequently used bag-based system leaves compostable waste unpriced. In 2000, 14 municipalities used a bag-based system for unsorted waste in combination with a free collection can for compostable waste. Only a minority of municipalities that have a bag-based system use bags for both unsorted and compostable waste (6 municipalities in 2000). As the incentives of the two systems differ, we include both types separately in the estimations.

Maximum flexibility results from a weight-based system. The collection vehicle weighs the can and combines this information with the identity of the owner, stored in a chip integrated in the collection can. In this case, a greater weight of waste results in a higher collection fee. While the number of municipalities using a weight-based system has increased, in 2000 still only 13 municipalities had introduced such a system.

As data are available for 1998-2000, we estimate a panel model using both the cross-section and the time-related variation.[5] For each waste stream (total waste, unsorted waste, recyclable waste and compostable waste), we estimate:

$$Waste_{w,i,t} = \alpha_s UPB_s + \beta SE + c_i + d_t + \epsilon_{i,t} \tag{2.2.1}$$

where $Waste_{w,i,t}$ is the quantity of waste stream w in municipality i in year t, UBP_s are dummies with the value 1 if municipality i has a unit-based pricing system of type s in year t, SE is a vector of socio-economic characteristics, c_i are time-invariant regional fixed effects,[6] d_t are time fixed effects and $\epsilon_{i,t}$ is the normally distributed error term (where necessary corrected for cross-sectional heteroskedasticity).[7]

To correct for differences between municipalities, we include the following socio-economic characteristics: the area of a municipality per inhabitant (and its square), the average family size, the number of non-western foreigners per inhabitant, the percentage of total inhabitants earning a median income, the number of houses sold per inhabitant, the number of flats sold per inhabitant, a dummy for small municipalities, a dummy for large municipalities and the percentage of inhabitants older than 65.[8] Data for the socio-economic characteristics come from the CBS (the Dutch Central Bureau for Statistics). Descriptive statistics for the variables are given in Table 2.2.

[5] We tested the assumption that pooling the different years is valid. An F-test on the sum of squared residuals rejected this assumption at the 99% level (F-statistic is 2.04). However, we only present results for the pooled model because a comparison with results for the separate years showed that the estimated coefficients are very robust. Only for the frequency variable was the coefficient significantly different from the panel estimates at the 95% level for 1998 (-0.11) and 2000 (-0.29). The reason for this is the sharp rise in the number of municipalities using the frequency system.

[6] Ideally, we would include a fixed effect for each municipality. However, as the unit-based pricing system dummies are highly invariant with respect to time, this is not possible. As a second best, we include a dummy for each province. Results for these fixed effects are available upon request.

[7] We tested all specifications for heteroskedasticity using the Breusch-Pagan test. It showed that for estimations with the independent variables in levels, heteroskedasticity could not be rejected. Therefore we estimated with the independent and, where possible, right-hand-side variables in logs (see Appendix). In cases where heteroskedasticity could still not be rejected, we corrected the standard errors with the White procedure (see Table 2.4).

[8] We tested the robustness of the estimated coefficients for the unit-based pricing systems by estimating a wide variety of different equations. Excluding some of the control variables or including extra control variables (such as the percentage of inhabitants in full-time work, the percentage of western foreigners, the number of families with 1, 2 or more children, the amount of property tax paid and the size of the agriculture sector) showed that the estimated coefficients for the unit-based pricing systems are very robust. For example, the coefficients for total waste are between -0.48 and -0.53 for the weight-based system and between -0.23 and -0.26 for the frequency-based systems. Further results are available upon request.

2.2.2 Results

Table 2.4 presents the estimation results. The F-statistics show that the equations are significant, while the relatively high (adjusted) R^2's indicate that the explained variation is not small.

Table 2.4: Estimation results

	Total	Unsorted	Compostable	Recyclable
UBP_{weight}	-0.48	-0.68	-0.95	0.19
	(0.02)	(0.03)	(0.05)	(0.03)
$UBP_{bagunscom}$	-0.44	-0.68	-0.93	0.26
	(0.02)	(0.04)	(0.06)	(0.03)
UBP_{baguns}	-0.15	-0.74	0.31	0.15
	(0.02)	(0.03)	(0.04)	(0.02)
UBP_{fre}	-0.24	-0.32	-0.46	0.09
	(0.01)	(0.02)	(0.03)	(0.02)
UBP_{vol}	-0.07	-0.13	-0.01#	0.03#
	(0.02)	(0.02)	(0.03)	(0.02)
UBP_{oth}	-0.15	-0.47	-0.02#	-0.01#
	(0.03)	(0.04)	(0.06)	(0.05)
ln(Retire)	0.11	0.04#	0.27	0.09**
	(0.02)	(0.03)	(0.05)	(0.04)
ln(Fam size)	-0.24**	-0.61	0.55	0.31*
	(0.08)	(0.11)	(0.17)	(0.16)
ln(Foreigner)	-0.03	-0.00#	-0.12	-0.02*
	(0.01)	(0.01)	(0.02)	(0.01)
City	-0.05	0.01#	-0.23	-0.15
	(0.01)	(0.02)	(0.04)	(0.03)
Village	0.01#	-0.03**	0.03*	0.05
	(0.01)	(0.01)	(0.02)	(0.02)
ln(Density)	0.03	0.09	0.03*	0.00#
	(0.01)	(0.01)	(0.01)	(0.03)
ln(Density) squared	0.004*	0.028	-0.016	0.002#
	(0.002)	(0.003)	(0.005)	(0.009)
Ownhouse	0.002**	0.003*	0.015	0.002#
	(0.001)	(0.002)	(0.003)	(0.002)
Ownflat	-0.007	0.001#	-0.024	-0.013
	(0.002)	(0.003)	(0.004)	(0.004)
ln(Income)	0.24	0.24	0.07#	0.27#
	(0.06)	(0.09)	(0.14)	(0.17)
R^2 (adjusted)	0.63	0.68	0.63	0.26
F-statistic	77.31	106.85	87.80	17.50
White correction	Yes	No	No	Yes
No. of observations	1323	1451	1449	1334

Notes: Equations are estimated including a constant. Standard errors are given in parentheses. All coefficients are significant at the 99% confidence level, except for coefficients with */** which denotes significance at the 90%/95% level and for coefficients with # which denotes non-significance at the usual levels.

Pricing waste on the basis of weight has a highly negative and significant effect on total waste of 38%.[9] This effect differs for the underlying waste streams. Compostable waste diminishes by more than 60%. It seems that many Dutch households use home composting methods to reduce this type of waste. Also, the effect on unsorted waste - the most environmentally unfriendly waste stream - is large: introducing a weighing system reduces the amount by nearly 50%. From the estimations, it is clear that one of the important mechanisms generating this result is that the amount of recyclable waste increases when a unit-based pricing system is introduced: introducing the weight-based system leads to higher efforts in recycling glass, paper and textiles (up 21%). Of course, this is due to the fact that Dutch citizens do not have to pay a marginal price for the collection of this type of waste. Given the cross-price effect, the net decrease in unsorted waste is 29%.

Introducing a bag-based pricing system also reduces the amount of total waste. In municipalities that use the bag-based system both for unsorted and for compostable waste, total waste diminishes by 36%. For municipalities that collect compostable waste by using a free collection can, the reduction is only 14%. While the effects on unsorted waste are comparable for the two systems (-49% and -52%), the effects on the supply of compostable waste differ a lot. In municipalities with unpriced compostable waste collection, compostable waste increases (by 36%), while in the other municipalities (using a bag system for compostable waste as well as for unsorted waste), this waste decreases (by 61%). Interestingly, the effect on recyclable waste is also larger for municipalities that use the bag-based system for compostable waste. This suggests that in municipalities using a bag-based system only for unsorted waste, part of the recyclable waste is 'dumped' in the free compostable waste can. The intuition behind this result is that it takes less effort to use this can than to use the recyclables facility. The compostable waste can is in the direct vicinity of the house, while the collection infrastructure for recyclable waste is farther away, resulting in more time needed to deliver the recyclables. Interestingly, the effects of the bag-based system that prices both unsorted and compostable waste are comparable to those of the weight-based system.

The system based on frequency reduces the total amount of waste by 21%, due to a reduction in both unsorted waste (27%) and compostable waste (37%). As the effects

[9] As the dependent variable is in logs, the effects of the pricing dummies are calculated using e^x-1, where x is the estimated coefficient.

on unsorted waste are less pronounced than in the weight-based and bag-based systems, the stimulating effect on the collection of recyclable waste is smaller as well (up 10%).

The effects of introducing a system based only on the volume of the collection are smaller. Total waste decreases by only 6%, mainly due to the effect on unsorted waste as the effects on compostable and recyclable waste are insignificant. This result is not surprising since the volume-based system is less refined than the other systems.

Turning to the socio-economic characteristics, we find economies of scale for total waste. This corresponds to the results found in the literature. An increase in household size of one standard deviation reduces collected waste per inhabitant by 5%. Diseconomies of scale are found for compostable waste. A possible explanation is that households with three or more people are more likely to have a garden.

In addition, the amount of waste per capita is larger for municipalities with a larger population of elderly people or a smaller population of foreign people. This is especially the case for compostable waste. As the garden area of the household primarily determines the amount of compostable waste, it is clear that living in a city has a highly significant and negative effect on compostable waste and living in a village has a positive effect. Furthermore, as we should expect, the sign on compostable waste is negative for municipalities with many flats. Moreover, a larger area per inhabitant increases the waste stream. The coefficients on income for total and unsorted waste are in accordance with the literature and positive, while income has no influence on compostable and recyclable waste.

2.2.3 The price elasticities of the pricing systems

So far, we have estimated the effects of unit-based pricing systems using dummies for the different systems, as no information is available on tariffs for 1998-2000. However, we do have data on the tariffs in 2003.[10] Assuming that these tariffs are a proxy for the real tariffs in 1998-2000, we can estimate the price elasticities of the different unit-based pricing systems. This makes comparison with results found in the literature easier.

Table 2.5 presents the estimated elasticities. Consistent with the results presented in

[10] In the estimations we use the tariffs charged each time a can is emptied for the frequency system. For the volume system, we use the marginal weekly increase in the collection fee if a household subscribes to a larger can. To make comparisons between systems possible, the reported tariffs in Tables 2.1 and 2.5 are in real (2000) US dollars (using the GDP deflator) per 30 gallons (114 liters) of unsorted waste. Tariffs per mass unit are transformed to tariffs per volume unit using a regularly

Table 2.5: Estimated price elasticities

System	Price	Total	Unsorted	Compostable	Recyclable
Standard model					
Weight	4.39	-0.47	-0.67	-0.92	0.16
Bag, unsorted + compostable	2.02	-0.43	-0.66	-0.97	0.25
Bag, unsorted	2.15	-0.14	-0.71	0.29	0.14
Frequency	3.91	-0.22	-0.28	-0.40	0.08
Volume	1.94	-0.06	-0.12	-0.01#	0.01#
Model with environmental activism					
Weight	4.39	-0.40	-0.53	-0.81	0.12
Bag, unsorted + compostable	2.02	-0.36	-0.51	-0.85	0.20
Bag, unsorted	2.15	-0.07	-0.58	0.40	0.09
Frequency	3.91	-0.16	-0.16	-0.31	0.04*
Volume	1.94	-0.00#	0.01#	0.09	-0.03#

Note: Equations are estimated including the same socio-economic characteristics as presented in Table 2.4 (results are highly comparable and available on request).

Table 2.4, the price elasticities are highest for the weight-based system and the bag-based system that prices both unsorted and compostable waste. This is interesting, as the average tariff for the weight-based system is more than twice that for the bag-based system. The better results for the bag-based system are very clear when the elasticities of the volume and frequency systems are compared. While the average tariff for the volume-based system is more or less equal to that of the bag-based system and the tariff for the frequency system is 1.89 dollars higher, their elasticities are significantly lower.

The higher price elasticity for unsorted waste in the bag-based system than in the volume-based system is in line with the results of Van Houtven and Morris (1999). The much smaller average tariff for the bag based system in their study than in ours might explain the lower own-price elasticity found for the bag-based system and the insignificant effects on recycling compared with our findings.

Compared with the elasticities found in the literature, our estimated own-price elasticities for the bag-based and weight-based systems are high. For example, the study with the highest elasticity for the bag-based system (Podolsky and Spiegel, 1998) finds an elasticity of only -0.39.

Comparing the average tariffs charged in Dutch municipalities with the average prices charged by communities whose elasticities are estimated in the literature reveals that

reported maximum weight of 0.76 kilograms per gallon (3.79 liters).

the average Dutch tariff for the volume-based system is similar to the average tariffs reported in other studies (compare Tables 2.1 and 2.5). The average Dutch tariffs for the frequency-based, bag-based and weight-based systems are inside the range of tariffs evaluated in the literature. Thus, the higher own-price effects we estimated are not the result of higher prices in the Netherlands.

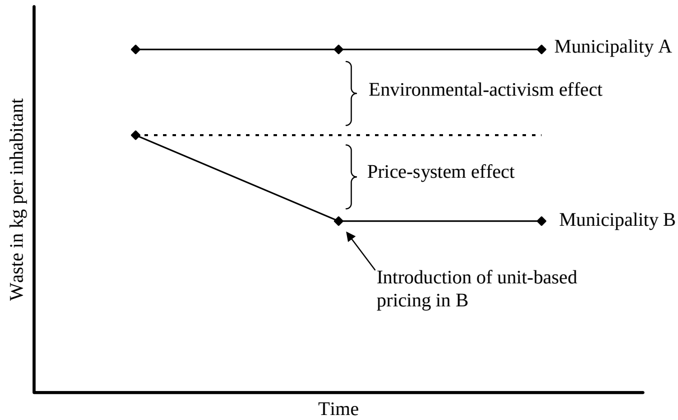
Interestingly, the cross-price elasticities we found for recyclable waste are not outside the range found in the literature. This suggests that the larger effects of bag-based and weight-based pricing in the Netherlands are not the result of more substitution between unsorted and recyclable waste. In the next two sections, we analyze whether the high Dutch elasticities are influenced by citizens' environmental activism and by leakage effects to neighboring municipalities.

2.3 The importance of environmental activism

Section 2.2 shows that unit-based pricing systems have a significant effect on the quantity of collected waste. Part of this effect may, however, result from a higher level of environmental activism. Figure 2.1 illustrates this point. Assume that citizens in municipality B (where unit-based pricing is introduced in the second period) are more concerned about the waste problem than citizens in the flat-fee municipality, A. Our method to estimate the effects of unit-based pricing systems compares the waste quantities of both municipalities, resulting in an estimate that is the sum of the environmental-activism effect and the price-system effect. The true effect of the price system for municipalities with a level of environmental activism comparable to that in municipality B is, however, equal to the difference in the second period minus the difference in the first period. The figures presented in Table 2.4 thus can overestimate the effects of unit-based pricing on the waste quantity of municipalities where such a system is introduced.

A way to deal with the environmental-activism effect is to take into account the political affiliation of the population. For example, Linderhof et al. (2001) suggest that because of the political affiliation of Oostzaan the estimated effects of the weight-based system of Oostzaan may not generalize for other municipalities. They evaluate the introduction of weight-based pricing in this small Dutch city using data before and after introduction of the pricing system. The largest political party in Oostzaan is Green Left (38% of

Figure 2.1: Influence of environmental activism on quantity of waste



the total vote), which is the most environmentally-friendly-oriented political party in the Netherlands. Green Left received only 7% of the votes nationwide in the parliamentary elections of 1998. This suggests that environmental activism is relatively high in Oostzaan, resulting in less-than-average amounts of waste before the introduction of the weight-based pricing system. Thus, the effect of introducing such a system in municipalities with less environmentally conscious citizens might be larger.

To check the influence of environmental activism, we included the fractions of the vote attained by each political party (based on the local election results of March 1998) in the estimations presented in Table 2.4. The Dutch political parties have different preferences with respect to environmental issues. For example, based on an evaluation of election programs, the Dutch Friends of the Earth gave Green Left an 8 for environmentally friendly policy proposals, while the right liberal party (VVD) was only given a 4.[11] It could be expected that municipalities in which green parties received a high percentage of the votes produce less waste than right-wing municipalities. However, statistical analysis shows that none of the Dutch political parties has a significant influence on the total amount of waste and therefore we conclude that political affiliation is a weak explanatory variable for environmental activism.[12]

[11] See Milieudefensie of April 1998, www.milieudefensie.nl/blad/1998/april98/twverkie.htm.

[12] Some significant effects were found for vegetable, food and garden (VFG), glass, paper and textiles (GPT) and solid waste, but the coefficients are very small. When the liberal party VVD's share of the vote increases by 10% percentage points, VFG waste increases by only 0.6%. While this increase is very small, the effects of other parties are lower still. Results are available upon request. In other research,

Therefore, we check the influence of environmental activism in another way. The communities that most want to recycle and to minimize waste going to disposal might be the ones that choose unit-pricing systems. If so, the pricing system and environmental activism are simultaneously determined with waste quantity. Therefore, the estimated effects of a unit-pricing system might already include the effect of environmental activism. To check this, we test whether municipalities that have introduced a unit-based pricing system in later years (1999 or 2000) already have lower waste quantities in the years before introduction. We do this by including a dummy variable that has the value 1 for each municipality with a unit-based pricing system in one or more years of our sample and the value 0 otherwise.[13] Including this activism dummy now corrects for the initial lower level of waste due to environmental activism in municipalities that introduce a unit-based pricing system.

Table 2.6: Estimation results including environmental activism

	Dependent variable is ln(Waste)			
	Total	Unsorted	Compostable	Recyclable
Activism	-0.07	-0.13	-0.10	0.04*
	(0.01)	(-0.02)	(-0.03)	(0.02)
UBP_{weight}	-0.42	-0.56	-0.83	0.15
	(0.03)	(0.04)	(0.06)	(0.03)
$UBP_{bagunscom}$	-0.38	-0.55	-0.83	0.22
	(0.03)	(0.05)	(0.07)	(0.03)
UBP_{baguns}	-0.09	-0.62	0.40	0.12
	(0.02)	(0.03)	(0.05)	(0.03)
UBP_{fre}	-0.18	-0.20	-0.37	0.06
	(0.02)	(0.02)	(0.04)	(0.02)
UBP_{vol}	-0.01#	-0.01#	0.08**	-0.01#
	(0.02)	(0.03)	(0.04)	(0.03)
UBP_{oth}	-0.09	-0.35	0.12*	-0.05#
	(0.03)	(0.04)	(0.07)	(0.05)
R^2 (adjusted)	0.64	0.69	0.64	0.26
F-statistic	77.75	108.02	85.99	17.01
White correction	Yes	No	No	Yes
Fixed effects	Yes	Yes	Yes	Yes
No. of observations	1323	1451	1449	1334

Note: Equations are estimated including the same socio-economic characteristics as presented in Table 2.4 (results are highly comparable and available on request).

we found also very weak evidence that political variables influence the institutional organization of refuse collection (see chapter 4).

[13]We also included such a dummy for each different type of unit-based pricing system. As expected, the activism effect is larger for municipalities with weight- and bag-based systems than for those with the other systems. However, as the change over time is not large for the individual systems, we only present results for the systems together.

As Table 2.6 shows, the activism dummy is significant for all waste streams. The results indicate that municipalities with a high level of environmental activism have 7% less waste. This means that a significant part of the estimated reduction in waste is due to environmental activism and not to the unit-based pricing system. Municipalities with a high level of environmental activism have 13% less unsorted waste, while the amount of compostable waste is 10% lower. As recyclable waste in such 'green' municipalities is 4% higher, households in municipalities with a unit-based pricing system are more active in sorting their waste regardless of the presence of such a system. Correction for environmental activism results in somewhat lower effects for the frequency-, weight- and bag-based systems, while the effect of the volume-based system on total waste is now insignificant. The environmental-activism dummy is also positive and significant for the estimations with tariffs. The estimated price elasticities are, on average, 0.13 smaller for unsorted waste, 0.10 smaller for compostable waste and 0.05 lower for recyclable waste (see Table 2.5).

The activism effect may explain part of the differences found in the literature. For example, the results based on household data in Fullerton and Kinnaman (1996) and Linderhof et al. (2001) will not be biased as they result from a comparison of the same households over different time periods. In this case, the environmental-activism effect is automatically excluded from the estimations. In contrast, studies that rely on cross-section analysis may overestimate the effects of unit-based pricing. This might explain why studies based on aggregate municipality data generally find larger elasticities than studies based on household surveys (see Table 2.1).

2.4 The effect of surrounding municipalities

Section 2.2 shows that unit-based pricing has a significant effect on the total amount of collected waste. The estimations suggest that one of the reasons for this result is that more waste is sorted. However, no attention was paid in that section to adverse behavioral effects. One of these effects is that unit-based pricing systems may introduce incentives for citizens to take their waste to municipalities without unit-based pricing systems. It seems logical to suppose that surrounding municipalities experience waste tourism as social contacts (family, friends) can be used to avoid the pricing system. For example, Linderhof et al. (2001) report a study by the city of Oostzaan, which estimates that about 4-5% of waste is taken to surrounding municipalities (which is approximately 13-17% of the reduction in waste prompted by the introduction of a weight-based pricing system).

To test whether municipalities without unit-based pricing systems collect part of the waste produced in surrounding municipalities with unit-based pricing systems, we estimate the models presented in Table 2.4 including impact factors. These factors measure how many inhabitants in surrounding municipalities have an incentive to take their waste to another municipality. Inhabitants of a municipality with a unit-based pricing system with one or more municipalities in their neighborhood without such a system do have an incentive for this behavior. Impact factors are calculated using the following equation:

$$IF_{s,i} \;=\; \sum_{j} \left((1 - \delta D_{i,j}) \, \frac{Inh_j}{Inh_i} \, S_i \right) \tag{2.4.2}$$

where $IF_{s,i}$ is the impact factor of municipality i having a unit-based pricing system s, i is a vector of all municipalities, j is a vector of the municipalities with a unit-based pricing system s in the neighborhood of municipality i, δ is a factor between 0 and 1, $D_{i,j}$ is the distance between municipality i and municipality j, Inh_i is the number of inhabitants of municipality i, Inh_j is the number of inhabitants of municipality j and S_i is a dummy with value 0 if municipality i itself has a unit-based pricing system and value 1 if it does not.

The impact factor for municipality i is a function of the distance to and the size of municipalities j (municipalities with unit-based pricing systems). The impact factor is larger when: (i) The distance from a municipality with a unit-based pricing system to a municipality without such a system is smaller. A linear relationship between impact and distance is assumed, while only municipalities with a distance less than 50 kilometers are included, i.e. $\delta = 0.02$ (the impact of municipalities which are more than 50 kilometers away is set to zero). Thus, we assume that taking waste to relatives and acquaintances is less likely if the distance is larger. (ii) There are more surrounding municipalities with unit-based pricing systems. If more municipalities with unit-based pricing systems surround a municipality without a unit-based pricing system, the effect will be larger. An extreme example in the Netherlands is Helmond, which does not have a unit-based pricing system and which borders 7 municipalities that have unit-based pricing systems and has 40 municipalities with unit-based pricing systems within a distance of 50 kilometers. On the other hand, 13 municipalities do not have any municipalities with unit-based pricing systems within this distance (consequently, their impact factors are 0). On average, a municipality without a unit-based pricing system has 6 municipalities with unit-based pricing systems in its vicinity. (iii) A surrounding municipality with a unit-based pricing system is larger. A surrounding municipality with a unit-based pricing system having the same number as a neighboring municipality without a unit-based pricing system will have less effect on the quantity of waste collected in this latter municipality than will a municipality with 10 times as many inhabitants. The impact factor is 0 when municipality i itself has a unit-based pricing system. The impact factors are calculated for the different unit-based pricing systems s. For example, $IF_{weight,i}$ is a measure of the impact on collected waste in a municipality without a unit-based pricing system of surrounding municipalities with a weight-based system. Table 2.7 presents the means and standard deviations of the impact factors.

Table 2.7: Estimation results: models with impact factors

	Descriptive stats		Effect of impact variables on ln(Waste)			
	Mean	St. dev.	Total	Unsorted	Compostable	Recyclable
IF_{weight}	0.19	0.44	$0.012^{\#}$	0.028^{**}	$0.034^{\#}$	$0.016^{\#}$
			(0.007)	(0.012)	(0.021)	(0.018)
IF_{bag}	0.86	2.93	$-0.001^{\#}$	$0.003^{\#}$	$-0.004^{\#}$	-0.010^{*}
			(0.001)	(0.002)	(0.003)	(0.005)
IF_{fre}	0.59	2.00	$-0.002^{\#}$	-0.012	$0.000^{\#}$	0.009^{**}
			(0.001)	(0.003)	(0.005)	(0.004)
IF_{vol}	1.02	2.35	$0.000^{\#}$	-0.000	$-0.002^{\#}$	$-0.003^{\#}$
			(0.001)	(0.002)	(0.004)	(0.005)
IF_{oth}	0.29	0.84	0.010	$0.007^{\#}$	0.041	$-0.009^{\#}$
			(0.003)	(0.006)	(0.010)	(0.007)

Note: Equations are estimated including the same socio-economic characteristics as presented in Table 2.4 (results are highly comparable and available on request).

As is shown in Table 2.7, the estimations give little indication of a significant effect from waste tourism. Only 4 out of 20 coefficients are positive and significant, while the size of these coefficients is very small. Furthermore, 3 of the 4 coefficients for the weight-based system are insignificant at 90%, while this system is expected to have the largest effect on surrounding municipalities (evaluated at the mean, the significant effect of the weight-based system is an increase of only 0.6% in the quantity of collected unsorted waste).

To test for misspecification, we also estimated with a non-linear impact factor (decreasing with distance) omitting the scale effect. In this case, only two coefficients are significant. Other estimations also produce few significant coefficients.[14] Therefore, we conclude that the taking of waste to municipalities without unit-based pricing systems is relatively unimportant in the Netherlands.

2.5 Administrative costs and illegal dumping

Section 2.2 shows that the effectiveness of bag-based pricing is comparable to that of weight-based pricing. This is an interesting result because the administrative costs for bag-based pricing are much lower. VROM (1997) evaluates weight-, bag- and frequency-based pricing systems in 12 Dutch municipalities.[15] According to this study, average administrative costs are higher for the weight-based pricing system (6.86 euro per inhabitant) than for the other systems (3.18 euro for the bag-based system, 4.28 euro for the frequency-based system).

Given the large reductions in unsorted waste, municipalities can save a lot of money by introducing (especially) a bag-based pricing system. For example, the saving in disposal costs is 5 euro per inhabitant larger than the rise in administrative costs for the bag-based system.[16]

The introduction of unit-based pricing systems may, however, have adverse effects. Citizens may take their waste to neighboring municipalities or may dump their waste illegally. Analysis of the behavior of Dutch citizens in Section 2.4 shows that there is no evidence that surrounding municipalities without unit-based pricing systems in fact collect part of the waste produced in municipalities with unit-based pricing systems. The evidence on illegal dumping is more mixed. Some studies give support for the hypothesis that illegal dumping is an important issue. Fullerton and Kinnaman (1996) estimate that illegal dumping constitutes 28% of the total reduction in waste collected

[14] We estimated models including impact factors calculated with higher (δ = 0.013 and maximum distance of 75 kilometers) or lower (δ = 0.04 and maximum distance of 25 kilometers) influence from neighboring municipalities, impact factors that are only 0 if the same unit-based pricing system applies and the environmental-activism dummy. As there was no clear pattern in the results, except that the estimations give insignificant coefficients for nearly all impact variables, we only present the results of estimations with the scale-related linear impact factors with δ = 0.02. Other results are available on request.

[15] The administrative costs for 1997 are given in 2000 prices.

[16] This calculation is based on the cost of incineration (the cheapest available and allowed option in the Netherlands). According to Dijkgraaf et al. (2001), total cost per tonne for an efficient incineration plant built in accordance with European law is 77 euro per tonne. Furthermore, VROM (1997) shows that only 2 municipalities (with frequency-based systems) report savings in disposal costs smaller than the rise in administrative costs.

at the curb. Hong (1999) shows that dumping was substantial after the adoption of the unit-based pricing system in Korea. On the other hand, Reschovsky and Stone (1994) find no relation between illegal dumping and unit-based pricing, while Van Houtven and Morris (1999) report that "officials ... found little to no evidence of more littering or increased use of accessible dumpsters."

For the Netherlands, Linderhof et al. (2001) state that illegal dumping is virtually non-existent in Oostzaan. According to them, the monitoring system in Oostzaan, with fines for illegal dumping, appears to be very effective in terms of deterrence. Moreover, another explanation for the absence of illegal dumping is that a small municipality such as Oostzaan has a large degree of social control. In general, the high population density of the Netherlands would suggest a low level of illegal dumping compared with other countries. This is confirmed by the lack of clear anecdotal evidence despite the large number of municipalities with unit-based pricing. However, as the main disadvantage of unit-based pricing systems is the potential effect on illegal dumping, it seems worthwhile investigating an effective monitoring and fining system and the conditions under which such a system would work.

2.6 Conclusions

This chapter provides an empirical analysis of the effects of unit-based pricing of household waste for the Netherlands. We find that the weight- and bag-based pricing systems perform far better than the frequency- and volume-based pricing systems. The bag-based system seems to be the best option, as its effects are comparable to those of the weight-based system and yet its administrative costs are far lower.

Compared with the elasticities found in the literature, the estimated Dutch own-price elasticities for the bag-based and weight-based systems are high. The higher elasticities are not the result of higher marginal tariffs in the Netherlands or of higher cross-price elasticities. A possible explanation might be that more waste is taken to other municipalities (without unit-based pricing systems). However, statistical analysis does not provide evidence that neighboring municipalities do collect part of the waste of municipalities that have unit-based pricing systems. Another possibility is that more waste is illegally dumped. Unfortunately, we have no data with which to estimate the effects on illegal dumping. Monitoring and fining may be important to deter this behavior Given the high population density of the Netherlands and the lack of anecdotal evidence, it seems implausible that a large part of the reduction in unsorted waste is due to illegal dumping.

Therefore, it seems likely that the introduction of unit-based pricing results in a significant change in citizens' behavior. Analysis of the waste quantities before and after introduction of a unit-based pricing system shows that environmental activism does play a role. Waste quantities are lower in municipalities that introduce unit-based pricing in later years. Thus the estimated effects of unit-based pricing may overestimate the effects of unit-based pricing when it is introduced in 'green' municipalities. On average, the estimated price elasticities are 0.13 smaller for unsorted waste, 0.10 smaller for compostable waste and 0.05 lower for recyclable waste when we correct for the environmental-activism effect. However, for municipalities with a low level of environmental activism, the estimated effects based on the dummy-variable approach may be applicable, as introduction of a unit-based pricing system internalizes the lack of environmental activism.

Furthermore, this chapter illustrates that refining unit-based pricing results in greater reductions in collected waste. A simple explanation of why the estimated elasticities for the bag-based system are higher in the Netherlands than elsewhere might be the significantly smaller volume of the bags used (50 to 60 liters or 13 to 16 gallons) compared with those in the USA (113 to 121 liters or 30 to 32 gallons). That this might be an important issue is indicated by the estimated elasticities of the frequency-based system. While the volume of the Dutch cans in the frequency-based system is comparable to that of the bags in the USA, the estimated Dutch elasticities for the frequency system are also comparable to the elasticities found for the bag program in the USA. Furthermore, the relatively small volume of the Dutch bags might explain why weight-based systems have comparable elasticities.

The smaller bag volume may explain why elasticities for the bag-based system are higher in the Netherlands, but not how Dutch citizens manage to achieve such large decreases in waste as estimated in this chapter. Detailed case studies might be necessary in order to generate enough information to get a grasp of the changes in citizens' behavior when they are confronted with marginal pricing.

Appendix Definition of variables

Waste_{total}	Annual total waste collected, in kilograms per inhabitant (sum of unsorted, compostable and recyclable waste)*
$\text{Waste}_{unsorted}$	Annual unsorted waste collected, in kilograms per inhabitant*
$\text{Waste}_{compostable}$	Annual compostable waste collected, in kilograms per inhabitant*
$\text{Waste}_{recyclable}$	Annual recyclable waste (glass, paper and textiles) collected, in kilograms per inhabitant*
UBP_{weight}	Dummy = 1 if municipality has a weight-based pricing system
$\text{UBP}_{bagunscom}$	Dummy = 1 if municipality has a bag-based pricing system for both unsorted and compostable waste
UBP_{baguns}	Dummy = 1 if municipality has a bag-based pricing system for unsorted waste
UBP_{fre}	Dummy = 1 if municipality has a frequency-based pricing ystem
UBP_{vol}	Dummy = 1 if municipality has a volume-based pricing system
UBP_{oth}	Dummy = 1 if municipality has an unspecified type of pricing system
Retire	Percentage of inhabitants older than 65*
Fam size	Number of inhabitants per household*
Foreigner	Number of non-western foreigners per inhabitant*
City	Dummy = 1 if municipality has more than 100,000 inhabitants
Village	Dummy = 1 if municipality has less than 20,000 inhabitants
Density	Area of municipality, in hectares per inhabitant*
Ownhouse	Number of houses sold per 1000 inhabitants
Ownflat	Number of flats sold per 1000 inhabitants
Income	Percentage of inhabitants with income over 12,400 and under 21,400 euro*
IF_{weight}	Impact factor measuring surrounding municipalities with weight-based pricing
IF_{bag}	Impact factor measuring surrounding municipalities with bag-based pricing
IF_{fre}	Impact factor measuring surrounding municipalities with frequency-based pricing
IF_{vol}	Impact factor measuring surrounding municipalities with

	volume-based pricing
IF_{oth}	Impact factor measuring surrounding municipalities with unspecified type of pricing
Activism	Environmental activism dummy with value 1 for each municipality with a unit-based pricing system in one or more years of our sample and value 0 otherwise.

Note: variables with * are logged.

Chapter 3

Cost savings of contracting out refuse collection

3.1 Introduction

Contracting out tasks like refuse collection, building cleaning, catering and vehicle maintenance has become an important measure to improve efficiency within the public sector. There is much evidence that contracting out certain public services can imply an efficient provision of services well adapted to needs and reduces the costs to tax payers. In an overview article Domberger and Jensen (1997) show that contracting out suggests cost savings in order of twenty percent without sacrificing the quality of service provided for a number of government services.

In this chapter, we focus on the effects of contracting out refuse collection. A number of empirical studies are published on the effects of different institutional forms on performance in the waste collection market. The studies estimate the effects of private collection (or contracting out) by estimating a cost function. Generally, these studies show considerable cost savings, if refuse collection is contracted out.[1]

Kitchen (1976) estimates a cost decrease of $ 2.23 per capita when private firms collect household waste with data for 48 Canadian municipalities. Observations of 340

[1]Some studies only compare the average cost for private versus public collection on the basis of ratio analysis, see e.g. Savas (1977 and 1981) and McDavid (1985) or Data Envelopment Analysis, see e.g. Cubbin et al. (1987). However, these methods fail to account for the effects in changes of other variables. By estimating a cost function, institutional effects but also other factors as the frequency of collection and density of the infrastructur can be taken into account. Therefore, we rely on this method in this article.

public and private firms in the USA, Stevens (1978) indicate a cost decrease of 7% to 30% due to contracting out. The magnitude of the effect depends on the size of the municipality. Pommerehne and Frey (1977) study refuse collection in Switzerland and again the private sector comes up with lower costs that amounted to 20 percent. Domberger et al. (1986) published a study on the effects of contracting out household refuse collection in the United Kingdom. Making use of a data set with 610 observations for 305 municipalities, they concluded that there are cost savings of 22% for contracting out to private companies and 17% for contracting inside. Szymanski and Wilkins (1993) and Szymanski (1996) have confirmed the results, based on an extension (in years) of this database. Ohlsson (2003) reports comparable efficiency gains of contracting out for Sweden. Bosch et al. (2000) analyse Spanish data for 73 municipalities in Catalonia. They pointed out that the framework for which the service is provided is more relevant than the public private dichotomy. In a recent contribution Reeves and Barrow (2000) pointed out cost savings of around 45% in Ireland.

Though studies are performed for different countries, a study in the Netherlands is missing. We try to fill the gap and show that results of other studies are confirmed if we use comparable estimation techniques. Furthermore, we extend these studies in two directions. First, with the exception of Stevens (1978) all cited studies pool observations of waste collection units with respect to institutional forms to estimate the effects of contracting out. With this pooled data set a cost function is estimated and the coefficient of the included institutional dummy reveals the effect of different institutional forms. It is, however, questionable if this pooling is acceptable. Chow (1960) states that: "Often there is no economic rationale in assuming that two relationships are completely the same" (p. 591). In other areas of economics Chow stability tests are used frequently, see e.g. Apergis et al. (1997), Lai (1994) and Loomis (1989). The most important application of the Chow stability test is to check for the Lucas critique in time-series. However, checking for different types of models with cross-sectional databases can be important as well.

A priori it is not sure whether external refuse collection firms (outside firms) apply the same waste collection technology as internal municipal waste collection units (inside firms). Outside firms handle the collection process from a different perspective while organisational goals also differ. Moreover, differences in municipality size can lead to different collection techniques. For instance, bigger cities have more opportunities to make use of scale economies. If production techniques are not identical, pooling can lead to biased coefficients. Therefore, if pooling is not justified, different cost functions have to be estimated for each sub-sample. The omission of the checks on the validity of pooling in the mentioned studies may lead to biased estimated effects of contracting out on performance. From a policy perspective, it is important that estimations of possible cost savings are accurate.

Secondly, compared with previous studies more emphasis is put on the fiscal system. Due to the Dutch fiscal system there is a disincentive for contracting out. Even though we can estimate significant cost savings when waste collection is contracted out, households will not experience these cost savings on a one to one basis. In the Netherlands private collection firms have to pay VAT while public firms are exempt. Countries such as the United Kingdom and Denmark have a compensating system, in that local authorities are tax-neutral towards contracting outside or inside. Thus, the current fiscal system in the Netherlands renounces the role for private collection firms.

3.2 Effects of tendering

Although many foreign econometric studies on effects of contracting out refuse collection have been published, such estimations are not available in the Netherlands. This section is an attempt to fill this gap by estimating a cost function, making use of a representative data set for Dutch municipalities. To make the results comparable the applied technique in this section corresponds with the studies cited in the previous section. The Chow stability test is applied in the next section.

3.2.1 Method

The driving forces behind the total collection cost per municipality (C), include a number of variables. These variables are based on previous research (see e.g. Stevens, 1978).[2] First, the number of pick-up points (Q) is expected to determine part of the total cost. This reflects on the cost, which a collection unit has to make by the number of stops. Secondly, the time spent at the pick-up stop (more bags or bins) can determine total cost. The number of inhabitants per pick-up point (I) approximates these cost. A third driving force is the time to arrive at the different pick-up points. The density variable, surface per pick-up point (D) approximates this. Fourth, the frequency of collection

[2] No price variables for the different inputs are included, because no reason exists ex ante why factor prices would differ between municipalities.

(F) is expected to have influence on total collection cost and is therefore included. Furthermore, the percentage of glass (G), paper (P) and vegetable, fruit and garden waste (V) separately collected is included in the estimations.

Thus, the following standard equation is estimated:

$$C = f(Q, I, D, F, G, P, V, O) \tag{3.2.1}$$

Furthermore, we include a dummy for the institutional form in which waste is collected (O). Main difference of the institutional form is whether waste is collected by the municipality itself or outside. Within this category we can discriminate between two types on the basis of ownership, i.e. public and private. Public outside collectors are a combination of municipalities for which waste is collected by an other municipality and municipalities that formed an independent public organisation. Given the division of institutional forms, the basic model is tested whether the ownership of the outside collection service does matter.

Expected signs are positive for the number of pick-up points, inhabitants per pick-up point, surface per pick-up point and collection frequency and negative for the institutional dummy's, while signs of the coefficients for the percentage collected glass, paper and vegetable, fruit and garden waste are undetermined a priori.

The estimated functional form is based on a Cobb-Douglas production technique and minimisation of a total cost function.

3.2.2 Data

To collect data 120 municipalities were approached in the period November 1996-April 1997. These municipalities were selected at random from 646 Dutch municipalities. A total of 85 municipalities have responded to this inquiry, a response rate of 71%.[3] The 85 municipalities responded to an inquiry on the collection of waste in 1996. The resulting database was checked on consistency of answers and the reliability was checked by spot checks on key answers.

Of the 85 municipalities 41 collect their waste not inside, but trough an outside organisation (see Table 3.1). Of the 41 outside firms, 13 were public independent organisations

[3] In 1996 four municipalities were absorbed by another, 31 municipalities did not participate in this inquiry. See appendix A for details on the inquiry.

Table 3.1: Descriptive statistics

	Average	Maximum	Minimum	St. dev.
Total cost (mln. euro)	1.6	20.5	0.1	2.5
Pick-up points (number)	16386	267000	400	30618
Inhabitants (per pick-up point)	4.0	64.7	1.6	8.1
Density (km^2 per pick-up point)	11	93	1	15
Frequency (>1 per week, dummy)	0.19	1	0	0.39
Glass (%)	3.2	11.1	0	3.0
Paper (%)	6.6	29.7	0	7.5
VFG (%)	28.4	47.4	0	9.9
Outside (dummy)	0.48	1	0	0.50
Private outside (dummy)	0.29	1	0	0.46
Public outside (dummy)	0.19	1	0	0.39

VFG = vegetable, fruit and garden waste

while 3 municipalities collect the waste through an other municipality. The remaining 25 municipalities collected the waste through a private collection firm.

Total cost per municipality is measured by multiplying the refuse collection rate(s) by the total number of households. Total cost is diminished by handling cost by multiplying cost per ton with tons recycled (glass and paper), composted (vegetable, fruit and garden waste) and disposed (incineration and dumping).

3.2.3 Fiscal aspects

A lot of attention has been drawn to the distortionary aspects of taxation for all kind of commodities (see Atkinson and Stiglitz (1980)). For the central question in this article taxation can also be crucial. The fiscal regime distorts the decision process in the Netherlands with respect to public versus private waste collection (see Wassenaar and Gradus, 2004). Private refuse collection is faced with a VAT rate of 19%, while public organisations are exempted from VAT. Therefore, a municipality in the Netherlands is biased towards inside production, because then refuse collection is exempted from VAT.[4]

A possibility to resolve this inequality could be to assess public refuse collection as a business activity and thus tax them with VAT. This policy has been introduced to public companies such as telecommunications. However, taxing refuse collection by

[4] In the United States this distortion is not important, because the sales tax is usually levied at the retail level (Davis and Meyer, 1983). Therefore, this point has not been raised in the reviewed literature in the previous section.

municipalities is not allowed according to EU laws. The other extreme, introducing a VAT exemption for enterprises is also not allowed.

The ministry of Finance has been working on a system to create a VAT compensation fund for public waste collectors (Wassenaar and Gradus, 2004). In line with a system already working in United Kingdom, all VAT a municipality has to pay will be refunded. In that case a municipality that decides to contract out the waste collection to a VAT liable firm will be compensated for the VAT the firm has to pay. Thus, contracting out decisions by a municipality are no longer distorted by the VAT difference between public and private firms.

The difference in fiscal treatment can not be neglected for the Dutch data set for a proper analysis. The municipality cost for private companies are 19% higher compared to public companies. However, the costs for a private company are 19% lower and in this respect the cost data are corrected.[5] Thus, the VAT component is subtracted from the total cost for private firms.

3.2.4 Results

Results for the basic model are presented in the first column of Table 3.2. The F-statistic shows that the equation is significant, while the high (adjusted) R^2 indicate that the explained variation is high. All coefficients have the expected sign. T-statistics are not corrected for heteroscedasticity as the White test (White, 1980) could not reject the homoscedasticity hypothesis for all estimations with 95% confidence.

The number of pick-up points has a significant impact on the total collection cost. A Wald test of coefficient restrictions (Pindyck and Rubinfeld, 1991) does not falsify the constant returns to scale hypothesis. This result confirms earlier results from Reeves and Barrow (2000), Collins and Downes (1977) and Hirsch (1965), while Stevens (1978) found also constant returns to scale for the large cities. Decreasing returns to scale were found by Bosch et al. (2000) and Domberger et al. (1986) and increasing returns to scale in Szymanski and Wilkins (1993), but coefficients were very close to one. Kitchen's (1976) inverted U-shaped average cost curve result was not confirmed since inclusion of a quadratic term was falsified with an F-test on 95% confidence.

The number of inhabitants per pick-up point, the pick-up frequency and the percentage

[5] A the cost data are for the fiscal year 1996, the VAT correction is based on the tariff of that year (17.5%).

Table 3.2: Estimation results cost functions

		Outside collection		Private collection	
Pick-up points	ln	1.05	(20.90)	1.05	(20.81)
Inhabitants per point	ln	1.00	(12.34)	1.01	(12.29)
Density (km^2 per point)	ln	0.01	(0.23)	0.01	(0.24)
Frequency	dummy	0.17	(2.07)	0.18	(2.10)
Glass	%	0.02	(1.41)	0.02	(1.36)
Paper	%	-0.01	(-1.40)	-0.01	(-1.25)
VFG	%	-0.01	(-2.26)	-0.01	(-2.06)
Private and public outside	dummy	-0.16	(-2.18)	-0.13	(-1.44)
Private outside	dummy			-0.05	(-0.50)
Constant		4.13	(6.96)	4.10	(6.84)
R^2		0.93		0.93	
F-value		132.3		116.48	
Log likelihood		-11.36		-11.22	
Probability homoscedasticity		0.41		0.40	
Number of observations		85		85	

T-statistics between brackets

of collected vegetable, fruit and garden waste have a significant impact on total cost. If the number of inhabitants per pick-up points increases with 1%, the total cost will rise with the same percentage. A higher pick-up frequency leads to 19% higher cost. Total cost decrease if more vegetable, fruit and garden waste are collected. It may be due to a scale effect as vegetable, fruit and garden waste is collected on a one bin per household while the number of bins per household is fixed.

The dummy for outside collection is significant. On average outside provision leads to 15% lower total cost.[6] In the second column the hypothesis is tested whether private outside collection does have an effect on total cost above that of outside provision. The negative coefficient implies that on average private collection is 5% cheaper than public collection. However, the basic model, without an ownership dummy, is not rejected on the basis of a Log-likelihood-ratio test (test statistic is 0.28). Furthermore, the dummy for ownership is not significant, while the coefficient for outside provision in the extended model does not differ from the basic model (using a Wald-test). Thus, we can conclude that the choice between outside and inside provision is more important than the ownership of the collection service. Competition seems to have more effects than the ownership issue. This is consistent with the literature (see Domberger and Jensen, 1997).

[6] Calculated as e^x-1, where x is the absolute value of the estimation for the dummy for outside provision.

Compared to Domberger et al. (1986) and Szymanski (1996) effects of changing institutional forms are somewhat lower but of the same order. Maybe competition in the Netherlands is somewhat less stringent since the private firms are not numerous. Three firms with only some small local private collection firms dominate private collection in the Netherlands.

An important result from our findings is that the difference in fiscal treatment between private and public 'firms' hampers tendering on the waste collection market.[7] Tendering to a private firm will not result in significant effects on tariffs paid by households. Dutch local governments are free to decide either to collect the waste by themselves or to tender the job. However, from January 2003 a VAT compensation fund is present for public collectors. According to our results these initiative will lead to a decrease in social cost of waste collection.

3.3 Robustness of results

As Ganley and Grahl (1988) make clear the results for institutional dummies can be influence by specific observations that perform much better or worse than expected. Therefore, we tested whether our result for the outside dummy remains robust when we skip municipalities with much lower or higher cost than expected. By iteration we excluded municipalities with a higher deviation of predicted to real cost than 30%. The outside dummy remains significant (but now even at 99%), while the coefficient remains robust.

An other point to investigate is whether the estimations depend on extreme small or big municipalities. Therefore, we tested whether a dummy for very big or small municipalities should be added to our basic model. Using a Log-likelihood-ratio test the basic model is not rejected.

Szymanski and Wilkins (1993) test for sample selection bias. They have two reasons to

[7] The corrections made because of the difference in tax treatment (17.5%) could be too high as public collectors can not deduct paid VAT on inputs. This paid VAT is part of the price consumers pay for the collection of waste. However, inputs with a VAT obligation are very low in total cost. For example total cost for collection trucks are only about 10% of total collection cost. This would result in a 1.75% point lower difference in effective VAT rates between public and private waste collectors. Moreover, the obligation for private firms to pay profit tax would diminish this difference as capital cost rise. Regressions with a 1% point lower effective VAT rate for private firms show only very small differences in coefficients for the institutional dummy's. Even a 10% point lower effective VAT rate for private firms results in a significant cost decrease if waste is collected by an outside firm.

suspect that sample selection bias could be a problem for their estimations. First, they estimate a cost function for a data set including different years while the response rate in 1988 was significant lower than in other years. This may be due to the introduction of compulsory competitive tendering in that year. Moreover, they suspect that authorities which performed a successful competitive tender were certainly keen to report, whereas an inefficient controlled authority did not likely to report (p. 117). As we do not have an indication that comparable problems exist in the Netherlands, we assume that sample selection bias is not a crucial problem. Furthermore, Szymanski and Wilkins (1993) found that there model without corrections for sample selection bias is not rejected.

Stevens (1978) tested for the validity of pooling the different municipalities in one sample. She concludes that different estimations have to be made for a few municipality size classes, but that pooling of the private and public collection firms was valid. Also Ganley and Grahl (1988), in a reaction to Domberger et al. (1986), emphasise to make a difference between urban and rural municipalities. Domberger et al. (1988) state in their reply that the included dummy for rural versus urban municipalities solves this problem. However, they did not check explicitly the validity of pooling the observations.

Chow (1960) made clear that testing for the validity of pooling observations is possible (see also Fisher, 1970). As unjust pooling of observations can lead to biased estimated coefficients this validity check is also necessary. Therefore, we checked the validity of pooling the observations for the Dutch data set with respect to municipality size and the different institutional forms, making use of the Chow test.[8]

Testing for the hypothesis that breakpoints exist with respect to small, mid-size and large municipalities reveal that this hypothesis can not be rejected (see Table 3.3). The impossibility to reject the breakpoint hypothesis with respect to municipality size could be due to the relative inflexible Cobb-Douglas form of the production function. However, testing for size breakpoints with a more flexible translog form holds the same conclusions.[9] Moreover, a breakpoint hypothesis with respect to the different institutional forms can not be rejected. The probability that no breakpoints exist for all three organisation forms is less than 5%.[10] This means that different cost functions must be

[8] Toyoda (1974) and Schmidt and Sickles (1977) showed that the Chow test for equality of regression coefficients is not robust to heteroscedasticity. Then other tests can be applied (see e.g. Thursby, 1992) Fortunately, the homoscedasticity hypothesis is not rejected for o–ur estimations.

[9] The translog cost function has exactly the number of parameters required for a flexible functional form, see e.g. Diewert (1987).

[10] Although a breakpoint is rejected at the 95% level for private collection versus other institutional forms, a breakpoint between private outside collection, public outside collection and inside collection

Table 3.3: Chow breakpoint tests

	No breakpoint hypothesis		
Breakpoint between rest versus:	F-statistic	Prob.	Conclusion
Public and private outside collection	2.98	0.01	breakpoint
Private outside collection	1.93	0.07	no breakpoint
Public versus private outside collection	1.98	0.03	breakpoints
$<$ 20,000 inhabitants	3.58	0.00	breakpoint
$<$ 40,000 inhabitants	0.30	0.96	no breakpoint
$>$ 20,000 and $<$ 40,000 inhabitants	2.02	0.03	breakpoints

estimated for the three institutional forms. For reasons of both types of breakpoints, our estimates in the previous section could be biased.

Combination of the two different breakpoint tests results in 6 sub-sample estimations. As our sample includes only 85 municipalities the estimations would become meaningless. Therefore, we follow a three-step approach. First, we take into account the effects of pooling the three sub-samples related to institutional form by estimating three equations. Secondly, we test these equations for the validity of pooling the observations with respect to municipality size. Third, we make some calculations based on non-parametric methods to estimate the effect of institutional form on cost.

Table 3.4: Estimation results cost function, different institutional forms

	Inside collection		Outside collection			
			Private		Public	
Pick-up points (ln)	1.10	(15.86)	1.04	(8.28)	0.96	(12.21)
Inhabitants per point (ln)	1.10	(12.49)	-1.33	(-0.47)	-2.05	(-1.94)
Density (km^2/point, ln)	-0.00	(-0.00)	0.11	(0.87)	-0.02	(-0.16)
Frequency (dummy)	0.14	(1.50)	0.21	(1.03)	0.11	(0.34)
Glass (%)	0.01	(0.67)	-0.02	(-0.64)	0.02	(0.54)
Paper (%)	-0.00	(-0.49)	-0.01	(-0.96)	0.00	(0.28)
VFG (%)	-0.01	(-2.13)	-0.01	(-0.91)	0.00	(0.37)
Constant	3.59	(4.59)	5.27	(3.65)	7.26	(4.54)
R^2	0.91		0.80		0.98	
F-value	61.78		14.52		109.55	
Probability homoscedasticity	0.22		0.55		0.66	
Number of observations	44		25		16	

t-statistics between brackets

Table 3.4 reveals the effects of sub-sampling on the basis of the different institutional forms. Comparing the coefficients for the estimated equations clearly reveals that they

could not be rejected.

are significantly different. Apparently, inside, public and private outside waste collectors have a different production technology. These results give an indication that outside firms can make more use of economies of scale. This is not surprising as municipal waste collectors are bounded on their borders. Outside waste collection firms are more flexible as they can combine the collection of different municipalities. The number of inhabitants per pick-up point is significant in the 'inside' equation, while they have no significant effect on the cost of the different outside firms. This applies also for the relative part of vegetable, fruit and garden refuse in total waste.

Table 3.5: Chow breakpoint test cost function, institutional samples

Estimation:	Inhabitants	Maximum F-statistic	Prob. (no breakpoint)
Private outside	19000	2.17	0.13
Public outside	na	na	na
Inside	27500	1.70	0.14

na means that the breakpoint test is not available for public outside collection due to the low number of observations

We tested the three estimated equations for the validity of pooling the observations with respect to municipality size, again with a Chow test. Table 3.5 summarises the results. Each equation was tested for breakpoints, the number of tests only limited by the number of observations. Reported is the maximal F-statistic found per equation. For the equations for private outside and inside waste collectors the Chow breakpoint test reveals that the no-breakpoint hypothesis could not be rejected. Therefore, we conclude that pooling with respect to municipality size was valid for these cases. Due to the low number of observations, the equation for public outside collectors could not be tested for breakpoints.

While the samples are now homogenous for institutional form, it is not possible to include a dummy for this variable in the estimations. Nonparametric comparison however can give an indication of possible cost differences between the samples. The estimated equations can be used to predict the development of cost when the institutional form is changed. Total cost for municipal collectors if they are contracted out can be predicted with the estimated equation for private collectors, making use of the known variables for municipal collectors.

Predictions using the estimated equations based on sub-samples confirms the cost decrease effect of changing the institutional form to a more market related direction.

Table 3.6: Predicted cost increases sub-sample estimations (% total cost)

From outside collection	to	inside collection	17.2
From private outside collection	to	inside collection	19.3
From public outside collection	to	inside collection	14.0
From inside collection	to	private outside collection	-14.8
From public outside collection	to	private outside collection	3.4
From inside and public outside collection	to	private outside collection	-9.9
From inside collection	to	public outside collection	-13.9

Contracting out the inside collection to a private firm would yield an average cost decrease of 14.8% (see Table 3.6). This is almost exactly what we found with the pooled estimation for the basic model. If the institutional form of inside waste collectors is changed to public outside the estimated cost decrease is 13.9%, only 1% lower than we found earlier. Of interest is the prediction for bringing outside firms inside. Apparently municipalities that collect waste by means of contracting outside have a very good reason for doing that as the predicted average cost increase is large.

3.4 Conclusions

While empirical research on the effects of changes in institutional form on the waste collection market for the Netherlands is missing, this paper fills in the gap. Our results confirm the results of earlier studies, i.e. contracting out refuse collection results in lower cost of 15%. Moreover, we can conclude that the choice between outside and inside provision is more important than the ownership of the collection service. Competition seems to have more effects than the ownership issue.

The statistical analysis indicates that waste collectors in smaller, medium and big municipalities have different production technologies. This also applies for different institutional forms. As more flexibility exist with respect to combining the collection of different municipalities, outside firms can make more use of economies of scale.

The fiscal system in the Netherlands hinders a more profound role for private waste collection as households will not benefit of the possible cost decreases. The burden of higher taxes for private firms counteracts the efficiency improvements. A VAT compensation fund would further stimulate the role of private waste collection. The current actions taken by the Ministry of Finance to correct the VAT difference between public and private firms are necessary to stimulate a fair choice between the real advantages and disadvantages of contracting out.

Appendix A Inquiry

The following questions were asked in the inquiry:

1. What is the name of your municipality?
2. How much inhabitants does your municipality have?
3. How much housholds do live in your municipality?
4. What is the surface of you municipality in km^2?
5. Is the collection of waste done by:
   (a) an outside organization (continue with question 6)
   (b) a cooperative (together with some surrounding municiaplities) organization (continue with question 10)
   (c) the own municipal service (continue with question 11)
6. What is the company's name of the outside collection firm?
7. Is the outside collection firm chosen by contracting out in competition with other collection firms?
8. Is the outside collection firm a public or a private company?
   (a) public (100% of the shares are owned by municipalities) (continue with question 15)
   (b) private (100% of the shares are owned by one or more private firms) (continue with question 15)
   (c) public/private (part of the shares is owned by municipalities and part by private firms) (continue with question 9)
9. Which part of the shares is in private ownhership (in % of total shares) (continu with question 15)
10. Which municipalities are working together in the cooperation?
11. Is the own municipal service chosen by contracting out in competition with other collection firms?

12. What is the name of the municipal service that collects the waste?
13. Does this service has an independent bookkeeping system of costs and benefits?
14. How many persons (in full time equivalents) are collecting the waste?
15. How much were the labor costs for waste collection?
16. How much was the average labor costs per hour worked?
17. How much waste was collected in your municipality
18. What was the composition of the waste (glass, paper, vegetable, fruit and garden waste, unsorted waste) in percentage of the total quantity collected?
19. Which part of the collected waste is landfilled, incinerated, composted and recycled in percentage of the total quantity collected?
20. What were the costs of landfilling, incineration, composting and recycling?
21. How many customers does your municipality have (please split this in customers for municipal and commercial waste)
22. If applicable, does your collection service do other activities than the collection of household waste (like the collection of commercial waste and the collection of waste in other municipalities)?
23. How often (in number of collections per week) is household waste collected?
24. How much are the total collection costs (excluding cost of treatment)?
25. Do you have to pay VAT for the collection of waste (on the basis of collection costs excluding cost of treatment)?
26. If you have to pay VAT, how much VAT do you pay?
27. How much is the tariff household pay for waste collection?
28. Does your municipality has an exemption for people with a social security benefit?
29. Is this exemption only for part of the collection fee?
30. Is this exemption financed by raising the collection fee for other households?

In the inquiry it was clearly indicated that all questions relate to the year 1996.

Chapter 4

The institutional choice of refuse collection

4.1 Introduction

There seems to be evidence that contracting out government services saves taxpayers money, and sometimes a lot of money, compared to public provision. In an overview, Domberger and Jensen (1997) show that contracting out a broad field of government services might result in cost savings in the order of 20% without sacrificing the quality of services provided.

Also Tang (1997), in a critical assessment of several studies, comes to the conclusion that the private sector is found to be more efficient in refuse collection, fire protection, cleaning services, and capital intensive waste-water treatment, while in sectors as water supply and railways the results are more mixed.

Especially, the cost savings of private refuse collection have been discussed at length in the literature. Kitchen (1976) estimates a cost decrease of Canadian $ 2.23 per capita when private firms collect household waste. Stevens (1978) arrives at a cost decrease between 7% and 30% due to contracting out for the USA, where the magnitude of the effect depends on the size of the municipality. Based on UK-data Domberger et al. (1986) published a study on the effects of contracting out household refuse collection in the United Kingdom. They concluded that there are cost savings of 22% for contracting out to private companies. Szymanski and Wilkins (1993) and Szymanski (1996) have confirmed these results, based on an extension (in years) of this database. Chapter 3 shows similar cost savings between 15% and 20% for the Netherlands, in case Dutch municipalities are contracting out refuse collection. Moreover, Ohlsson (2003) reports almost the same estimations for Sweden. Bosch et al. (2000) presented Spanish data for 73 municipalities in Catalonia. They pointed out that the framework for which the service is provided is more relevant than the public private dichotomy. In a recent contribution Reeves and Barrow (2000) pointed out cost savings of around 45 % in Ireland.

Although the practice of contracting out refuse collection has become more popular, it is still less common than in-house provision. In the United Kingdom only 30% of the contracts for refuse collection is placed out-house (see Szymanski (1996)). According to Reeves and Barrow (2000), in Ireland in 39% of the studied cases private providers were contracted to provide refuse collection. In the Netherlands 40% of the municipalities use private collectors for refuse. However, due to the fact that private collectors are especially active in small villages, only 20% of total tonnage is in private hands (see chapter 3). Only Ohlsson (2003) finds for the Swedish case that private provision is slightly more common than public provision.

Furthermore, a recent study by López-de-Silanes et al. (1997) shows the reservations of local authorities towards contracting out. Based on data in 1987 and 1992 for 3042 counties for twelve services like water supply, landfills, libraries etc. only 25% of the services in 1987 and 35% in 1992 had been placed out-house. Moreover, in this article a nice empirical investigation of the mode of providing government services is given, where three leading theories (namely efficiency, political patronage, and ideology) are investigated. The evidence presented in this article indicates that clean government laws and state laws restricting county spending encourage privatisation, whereas strong public unions discourage it. This suggests an important role played by political patronage and taxpayer resistance to government spending in the privatisation decision.

In this article, we examine for the Netherlands the determinants of the provision mode of refuse collection. Data are available for 540 (i.e., almost all) Dutch municipalities. We find evidence for political patronage and the wealth of the local government as a ground for contracting out, but also the possible efficiency gain of contracting out plays a role. Moreover, we extend the existing literature by investigating more general specifications. Especially, the usually applied logit model seems too restrictive. Formal tests strongly reject the appropriateness of the logistic probability transformation. As alternative we use a semiparametric single index modelling approach, based on Ichimura (1993), where the probability transformation is left unrestricted. We find that the semiparametric single indices are comparable to the parametric analogues, but the probability transformations are quite different, implying that the logit specification might yield misleading predictions, particularly, when considering marginal effects.

The remainder of this chapter is organised as follows. In Section 2 we discuss the relevant theoretical issues. In Section 3 we describe the data we use. Section 4 contains the estimation results based on logit. In Section 5 we investigate the robustness of these results, by testing the logit specification, and by using a semiparametric alternative, based on Ichimura (1993). Section 6 concludes.

4.2 Theoretical issues

Before we specify the data and the empirical results, it is worthwhile to discuss some theoretical issues concerning the contracting out decision (see also López-de-Silanes et al. (1997), and Tang (1997)). As mentioned in the introduction, Chapter 3 shows that Dutch municipalities might achieve cost savings between 15% and 20% in case of contracting out refuse collection. With lower service costs, one would expect that municipalities favour private collection. Indeed, 40% of the Dutch municipalities chose for the option to collect waste by a private firm. The question arises: why did the other 60% not choose this option as well?

Hart et al. (1997) argue that private contractors might fail to pursue goals that politicians want to attain. Especially, in circumstances such as health care and prisons, where politicians cannot write a complete contract that specifies exactly what contractors are supposed to do in all circumstances, it may not be straightforward to contract out. The logic suggests some potential efficiency benefits of in-house government services to ensure quality. However, it is not clear how important such benefits are for refuse collection.

Hart et al. (1997, p. 1154) argue that in the case of refuse collection the damage to quality can be offset by a good contract, so that "private provision is superior". Nevertheless, according to a Dutch inquiry, such elements are still available and some municipalities put forward that quality is the reason for in-house provision (see NG magazine (1998)). A prediction following from this kind of reasoning is that the wealth of local government decreases the likelihood of contracting out. A poorer government is less likely to care about quality and is more interested in cost savings.

Related to these wealth arguments are the so-called output arguments. Some empirical insights suggests a linear relation between the cost of service and output (number of inhabitants, pick up points etc., see, for example, Domberger et al. (1986)). However, especially for small municipalities this may not be true. Kitchen (1976) finds that the maximum scale in refuse collection occurs in cities of about 324,000 inhabitants. Stevens (1978) divides the sample into several subsamples. For small municipalities there is less evidence for this linear relation. Therefore, she finds increasing returns to scale, if the city population is less than fifty thousand and constant returns to scale if the city population is larger than fifty thousand. A prediction following from this kind of reasoning is that the number of inhabitants decreases the likelihood of contracting out. However, this relation may not be linear. Above a certain level there is less evidence that private waste collectors have more opportunities to combine the collection of different municipalities and thus to use scale effects as a cost decreasing mechanism.

An alternative view of the contracting out decision focuses on public choice theory (see Buchanan (1987)). This approach explains social behaviour as the product of free choices of individuals. Self-interested politicians, bureaucrats and unions have a stake in in-house provision as they can use it as a status-enhancing feature.

López-de-Silanes et al. (1997) argue that in the United States the main political factor favouring in-house provision seems to be the public employee unions. Moreover, the role of unions becomes more important and, therefore, in-house provision becomes more beneficiary if unemployment in a municipality is high.

The third theory stresses the importance of voter ideology. To evaluate this view, one should take into account voting patterns in different municipalities. Hereby, it is assumed that the contracting out decision is simultaneously determined by the degree of voters' anti-government sentiment. This laissez-faire sentiment is most visible in right-wing parties.

Finally, it is possible that the privatization decision in a particular municipality is related to what happens in other municipalities. For instance, Bivand and Szymanski (2000) find evidence for the UK that in the period before Compulsory Competitive Tendering

(CCT) costs were spatially correlated across authorities, while following CCT this spatial correlation disappeared.

To account for this effect, Bivand and Szymanski suggest that before CCT most local authorities evaluated the service costs by comparison with their local neighbours. Municipalities with a higher than average cost compared with the neighbours would choose the option of privatization. In addition, the decision of contiguous municipalities might affect the decision of a municipality via scale economy, especially when the municipality under consideration is small. Alternatively, one could argue that municipalities might take into account the decisions in some kind of reference group of municipalities, where the reference group consists of municipalities which are, for instance, comparable in size or in number of inhabitants.

However, contrary to the first three points, this fourth issue, interdependence between municipalities is much harder to quantify. Without knowledge of which municipalities influence which municipalities, the researcher will have to model such interdependencies him- or herself by modelling reference groups. However, as argued by Manski (1993) in the context of a linear demand equation for consumers with interdependencies between consumers, it is impossible to infer unknown reference groups on the basis of observed behaviour: an informed specification of reference groups is a necessary prelude to an analysis of interdependent behaviour. As such information is not available for our case estimation of the effects of interdependencies is not possible.[1]

4.3 Data

To test the theories about contracting out, a database is constructed with data on the different institutional forms of waste collection and variables representing the theories. The data on the different institutional forms is based on a 1998 census of the Dutch Association for Refuse and Cleansing Management (NVRD). Moreover, municipalities' characteristics are available from Statistics Netherlands (CBS). For 540 of the Dutch municipalities (96% of all municipalities) figures are available, see Table 4.1.

[1] Moreover, since our model is of a binary choice type, we would also have to deal with the problem of "coherency", when modelling interdependencies, see, for example Schmidt (1981) or Gourieroux et al. (1980): the interdependency should be of a recursive type (one municipality may influence the other, but then not the other way around), since otherwise the model is not coherent, i.e., probabilities do not sum to one.

Table 4.1: Descriptive statistics

Variables	Average	Max.	Min.	St. dev.
Private provision (%)	42	100	0	49
In-house provision (%)	28	100	0	45
Inhabitants (x 1000)	26	722	1	45
Inhabitants per hectare	6	63	0	8
Transfer from state (euro per inhabitant)	442	1727	118	113
Income per inhabitant (1000 Euro)	9	13	6	1
Unemployed per 100 inhabitant	3	6	1	1
Local civil servants per 100 inhabitant	11	16	8	3
Conservative Liberals (%)	16	52	0	9
Social Democrats (%)	16	49	0	9
Progressive Liberals (%)	8	34	0	7
Orthodox Protestants (%)	6	67	0	10
Green Left (%)	4	34	0	6
Extreme Right (%)	0	11	0	2
Local parties (%)	25	100	0	20

- Institutional forms

  In general, three modes of provision are used in this dataset. The first mode is provision by a private firm (42%). The second and third mode are both by public ownership but differ with respect to autonomy of the collection service. The second mode occurs when municipalities collect the waste of their own citizens (28%). The waste collection service is in this case under direct control of the municipality council. The third mode occurs when another municipality or an external public organisation (30%) collects the waste, so that the municipality council has less direct control on the waste collection service.

- Output variables

  To check for the output arguments the number of inhabitants and population density (number of inhabitants per hectare) are included in the empirical setting. On average a Dutch municipality has 26 thousand inhabitants, while the largest city (Amsterdam) has 722 thousand inhabitants and the smallest municipality only 1 thousand. To check for scale economy the number of inhabitants squared is included as well. Moreover, the population density shows a high variation between municipalities, indicating that the transport distance between individual pick-up points varies.

- Wealth variables

  The theory about the influence of wealth on contracting out suggests that budget constraints influence the trade-off between efficiency and social arguments. Hard budget constraints increase the likelihood of privatisation. In the Netherlands the income of local government depends almost totally on the transfers by the central government. The freedom of Dutch municipalities to collect their own taxes is quite restricted. Therefore, we include as an explaining variable the transfer from central to local government per inhabitant. As the trade-off between efficiency and social arguments depends on the social characteristics of the inhabitants we include the average personal income in a municipality as a wealth variable as well. The hypothesis is that a municipality will weigh cost savings more if the inhabitants are poor.

- Interest group variables

  In the López-de-Silanes et al.-study interest group variables are included for the number of public employee's or union membership and for the opportunity to purchase supplies from political allies (the so-called clean government variables). However, for the Netherlands clean government laws are dictated at a national level and, therefore, these data cannot be included. No data are available for the number of public employee's in a municipality. However, these data are available at a regional level and are, therefore, included.[2] Similar to López-de-Silanes et al., it is possible to include labour-market conditions as an approximation of interest group variables. In general, we should expect that in-house provision becomes more beneficiary if unemployment in a municipality is high. Therefore, the unemployment level is included in our estimations.

- Political variables

  We include the fractions of the following parties, based on the local elections of May 1994[3]: green left, social democrats, conservative liberals, progressive liberals, orthodox Protestants, extreme right and local parties.[4] In the estimations the Christian democrats, who are in the middle of the political spectrum, are excluded.[5]

[2] There are twelve provinces or regions in the Netherlands.

[3] There were new elections in May 1998.

[4] Green left: Groen Links + SP, social democrats: PvdA, conservative liberals: VVD, progressive liberals: D66, Christian democrats: CDA, orthodox Protestant: SGP + RPF + GPV, ultra right: CD and local parties: other parties. Combination of the parties is tested using a Log Likelihood test.

[5] In addition, we looked at municipality-level voting in the 1994-election for Parliament as alternative indicator of the electorate's ideological orientation. However, local elections seem to be the best means of predicting the probability of private contracting.

4.4 Estimation results: logit

We start our estimations with a standard logit analysis for two models.[6] In the first model, the choice between public and private provisions is estimated as dependent on a number of explaining variables. In the second model the choice between in-house and out-house provision is the dependent variable. In both models, all explaining variables are initially the same.[7] Thus, the basic model is:

$$P(Dep = 1 \mid x) = \wedge(\beta^T x), \tag{4.4.1}$$

where Dep is the dependent variable (for model 1 this is a dummy with value 1 for municipalities with no private collection, for model 2 this is a dummy with value 1 for municipalities with collection in-house) and where x contains the following explanatory variables (next to a constant term):

Inhabitants	number of inhabitants (*10000)
Funds	transfers from central government (euro per inhabitant)
Income	personal income (Euro per inhabitant)
Civil servants	number of civil servants (per 100 inhabitants)
Unemployment	persons with unemployment benefit (per 100 inhabitants)
Conservative Liberals	percentage of total votes in a municipality
Orthodox Protestants	percentage of total votes in a municipality
Social Democrats	percentage of total votes in a municipality
Progressive Liberals	percentage of total votes in a municipality
Green Left	percentage of total votes in a municipality
Extreme Right	percentage of total votes in a municipality
Local Parties	percentage of total votes in a municipality

To account for sufficient flexibility in terms of the number of inhabitants, we also decided to include the number of inhabitants squared (/1000). The parameter vector β contains the unknown parameters, and $\wedge$ represents the logit-transformation.

Results are given in Table 4.2. First, we discuss the no-private provision case.

- Output variables

  It shows that scale effects are present. The estimated second order polynomial in terms of inhabitants is increasing up to its maximum at around 312,500 inhabitants, so that with an increasing number of inhabitants (up to this maximum)

[6] The probit and the OLS results are extremely similar.

[7] An interesting extension would be to include the previous state of the dependent variable as an explanatory variable. However, such data are not available.

Table 4.2: Estimation results

Variables	No-private Logit	No-private Ichimura	In-house Logit	In-house Ichimura
Constant	-1.72	-	-4.51	-
	(1.88)	-	(2.18)	-
Inhabitants	0.26	0.26	0.18	0.18
	(0.11)	-	(0.08)	-
Inhabitants squared	-4.16	-3.94	-2.45	-2.57
	(1.73)	(0.30)	(1.13)	(0.38)
Population density	0.08	0.07	0.03	0.05
	(0.03)	(0.03)	(0.02)	(0.01)
Fund	1.38	1.11	1.36	1.34
	(0.66)	(0.55)	(0.73)	(0.33)
Income	-1.61	-1.24	-0.67	-1.17
	(0.93)	(0.73)	(1.05)	(0.43)
Unemployment	0.02	0.02	0.10	0.04
	(0.02)	(0.02)	(0.03)	(0.01)
Civil servants	0.27	0.19	0.10	0.04
	(0.05)	(0.05)	(0.05)	(0.02)
Conservative Liberals (%)	-0.004	-0.01	0.0008	0.009
	(0.02)	(0.02)	(0.02)	(0.01)
Social Democrats (%)	-0.0003	-0.02	-0.04	-0.03
	(0.02)	(0.01)	(0.02)	(0.01)
Progressive Liberals (%)	0.008	-0.02	0.03	0.01
	(0.02)	(0.01)	(0.02)	(0.01)
Orthodox Protestants (%)	0.03	0.01	0.02	0.009
	(0.01)	(0.01)	(0.02)	(0.01)
Green Left (%)	-0.02	-0.04	-0.01	-0.01
	(0.02)	(0.02)	(0.02)	(0.01)
Extreme Right (%)	-0.05	-0.08	-0.18	-0.29
	(0.13)	(0.06)	(0.08)	(0.04)
Local party (%)	-0.02	-0.03	-0.03	-0.008
	(0.01)	(0.01)	(0.01)	(0.01)
Log likelihood	-281.52		-256.40	

Estimated standard errors in brackets.

the probability of public provision (i.e., no private provision) increases.[8] The occurrence of scale effects makes public provision more likely. Furthermore, if the number of inhabitants per hectare increases the probability of public provision increases. Again scale effects are present.

- Wealth variables

  As we expected, more transfers by the central government favours public provision, because less emphasis has to be given to cost savings investigations. Contrary to our prior, a higher income level in a municipality lowers the probability of public

[8] In the Netherlands only three cities have more inhabitants than this maximum, namely The Hague, Rotterdam, and Amsterdam.

provision. However, the estimated coefficient is not significant.

- Interest group variables

  Interesting are the results with respect to the interest group variables. The data give evidence for the prior that the number of public employees raises the probability of public provision. Also the number of unemployed persons raises the probability of public provision, although the coefficient estimate is not significant.

- Political variables

  The results with respect to the political variables are much weaker.[9] Only local parties are against public provision in a significant way (compared to the Christian Democrats, who are the reference group). Probably, this can be explained by the anti-government sentiment by some of these local parties. From the other parties only the Orthodox Protestants are in favour of public provision in a significant way. This can probably be explained by the reserved attitude towards the role of market forces in these parties.

For in-house provision the over-all results are in line with no-private provision. The top of the polynomial in terms of inhabitants is now at around 367,000 inhabitants: thus, if the number of habitants increases (up to this maximum) then the probability of in-house provision increases. The effect of inhabitants per hectare, however, becomes insignificant. This also applies to the effect of transfers by the central government. For the number of unemployed persons the effect of in-house provision seems somewhat stronger and quite significant. The effect of income per inhabitant remains contrary to our prior, but the estimated coefficient is again not significant. The effect of the number of public employees is again positive and significant. This seems to be in line with the theory that interest group considerations are an important obstacle to out-house provision. In addition, the results for political variables are also here suggestive. The attitude of the social democrats and extreme right towards in-house provision turns

[9] The insignificance of political variables may be sensitive to specification of these variables. Therefore, we experimented with a "left/right" variable. This "left/right" variable is constructed as follows: 8*Green Left + 7*Social Democrats + 6*Progressive Liberals + 5*Christen Democrats + 4*Local Parties + 3*Orthodox Protestant + 2*Conservative Liberals + 1*Extreme Right. A loglikelihood test was used to investigate whether the model with individual parties is preferred. We obtain as test result for the no-privatisation case a value of 25.86 and for the in-house modelling a value of 27.52; since the test is asymptotically chi-squared-distributed with 6 degrees of freedom, the tests lead to rejection at the 1%-level, suggesting that the specification including different parties is preferred. Furthermore, the left/right variable is not significant at 10%.

out to be significant, whereas the effect of the other parties is insignificant (compared to the Christian Democrats).

4.5 Robustness of results

The basic logit model presented in the previous section requires strong distributional assumptions to be valid. In particular, the assumption that the probability transformation is given by the logistic probability distribution L may be questioned. To investigate the validity of this assumption we tested it against a more general specification as proposed by Ruud (1984), and as used by Newey (1985) for constructing conditional moment tests.[10] Thus we test $H_0 : \gamma_1 = \gamma_2 = 0$ in

$$P(Dep = 1 \mid x) = \Lambda(\beta^T x + \gamma_1(\beta^T x)^2 + \gamma_2(\beta^T x)^3)$$

using the test statistic proposed by Newey (1985), adapted to the logit specification. We obtain as test result for the no-privatisation case a value of 6.16; since this test is asymptotically chi-squared-distributed with 2 degrees of freedom, the test leads to rejection of the logit specification at the 5%-level. In case of the in-house modelling the test result becomes much higher: 33.95; this means rejection of the logit specification at all usual significance levels.

Consequently, it makes sense to investigate alternative specifications, which require less severe distributional assumptions. One possibility is a fully nonparametric approach, but due to the curse of dimensionality this will not work in our case with only 540 observations. So, we restrict attention to semiparametric models. There are several possibilities available in the literature for application to the binary choice case. One possibility is the Maximum Score estimator proposed by Manski (1985), and turned into smoothed Maximum Score by Horowitz (1992). Although (Smoothed) Maximum Score requires very weak distributional assumptions it has some drawbacks: it has a lower rate of convergence than ordinary parametric estimators and it only allows one to estimate the index, but not the probability transformation. Another possibility are the single index models in which the probability that the binary dependent variable equals one given the covariates is equal to a single index of the covariates evaluated in an unrestricted (nonparametric) probability transformation:

[10] Newey (1985) considers the probit specification; however, the adaptation to the logit model is straightforward.

$P(Dep = 1 \mid x) = H(\beta^T x)$

where H is an unknown function that has to be estimated as well. There are several estimators available to estimate such single index models. For instance, Klein and Spady (1993) provide a semiparametric efficient one. However, this estimator is quite hard to calculate in practice. We decided to use Ichimura (1993).[11] The estimator for β consists of solving the minimisation problem

$\hat{\beta} = Arg\min_b \sum_i (Dep_i - \hat{H}(b^T x_i))^2$,

where H represents a nonparametric estimator for

$P(Dep = 1 \mid x) = E(Dep \mid x) = H(\beta^T x)$

We estimate this latter conditional expectation using a kernel estimator with a standard normal Gaussian kernel. Since there is no optimality theory for the corresponding bandwidth, we have set it equal to the familiar rule of thumb $\hat{\sigma} n^{-1/5}$, with $\hat{\sigma}$ an estimate for the standard deviation of $\hat{\beta}^T x$.[12] The resulting estimator for β has a normal limiting distribution whose asymptotic covariance matrix can straightforwardly be estimated. See Ichimura (1993) for further details.

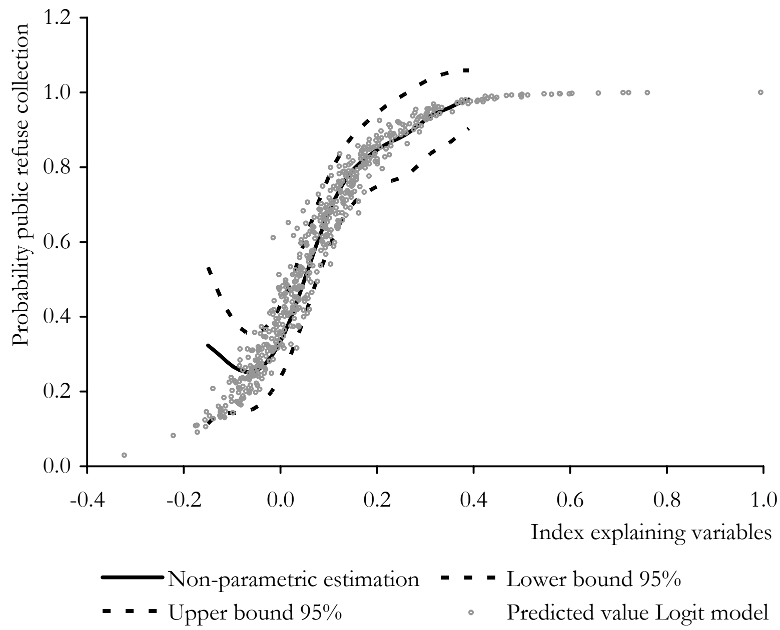
Table 4.2 contains the estimation results for β, and Figures 4.1 and 4.2 present the estimates for H, for the no-privatisation and in-house case, respectively. Notice that in the single index model the constant term is not identified (therefore, set equal to 0). Also the scale is not identified; we have fixed the scale by normalising the coefficient of the variable Inhabitants, equal to the corresponding estimated coefficient in the Logit model.

The estimation results in terms of β according to Ichimura are, at least qualitatively, quite comparable with those according to the logit specification. To investigate whether the results are also quantitatively the same, we considered the hypothesis that the coefficients of logit are (simultaneously) equal to the corresponding single-index coefficients of the Ichimura-specification. We tested this hypothesis by a Hausman-type test by using the difference of the vector of logit estimates and the corresponding Ichimura-estimates. The limit distribution of this difference can easily be obtained under the null hypothesis. The value of the resulting chi-square test statistic turned out to be 1.75 in case of no-private collection, and 3.69 in case of in-house provision. Sin-

[11] For other possibilities, see, for instance, Horowitz (1998).

[12] As a starting value for the iteration procedure we used the OLS-estimate for β, from which we also constructed the estimate for σ.

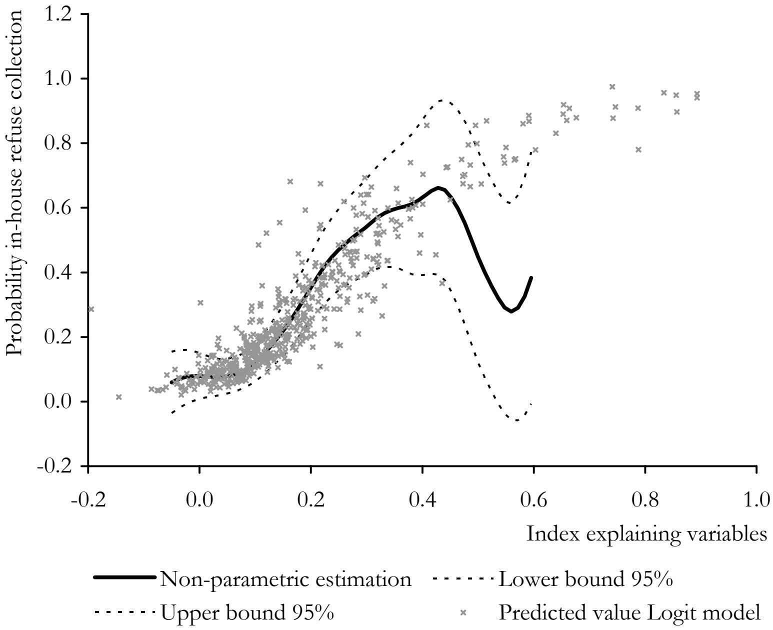
Figure 4.1: Non-parametric (solid lines) and logit (dots) estimation of choice between public and private refuse collection



ce, under the null hypothesis, the test statistic is asymptotically chi-square-distributed with 13 degrees of freedom (the number of coefficients, except the constant term and the normalised coefficient of inhabitants), we conclude that the results in terms of the single-index coefficients are also quantitatively the same.

Next, we turn to the estimated probability transformations. In Figure 4.1 we plot the nonparametric estimate of the probability transformation in case of no-private-collection, together with 95% confidence intervals. In addition, we plot in the figure the corresponding predictions according to the logit model. From this figure we can conclude that the logit- and the Ichimura-specifications for most observations are not too far apart from each other. However, a non-negligible part of the predictions according to logit fall outside the 95% confidence interval, which can be seen as evidence that the logit model is misspecified, in line with the earlier rejection of the logit probability transformation. Moreover, for the lowest values of the single-index the results of Ichimura differ substantially from logit, although not significantly so. It seems that the probability transformation is not increasing over the whole range, a feature that cannot

Figure 4.2: Non-parametric (solid lines) and logit (dots) estimation of choice between in-house and out-house refuse collection



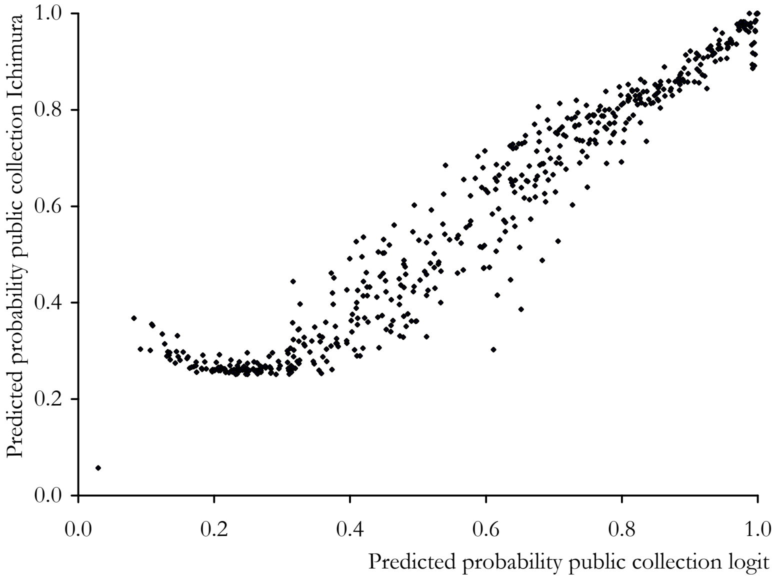
be captured by the logit-specification. In Figure 4.2 we present the corresponding plot in case of in-house-provision. Again, we see that for many observations the logit- and Ichimura-specifications are reasonably close, but not as close as in case of no-private-collection: Over the whole range we see predictions according to logit falling outside the 95%-confidence band.[13] Moreover, for larger values of the single index, the Ichimura probability transformation is not increasing, but inversely hump shaped, a pattern that clearly cannot be captured by the logit probability transformation. Concluding, based on the overall evidence, the difference between logit and Ichimura is significant, in line with the earlier reported rejection of the logit probability transformation.

Table 4.3: Percentage correct predictions of three models

	No-private collection	In-house collection
Naïve prediction	0.58	0.72
Logit	0.7574	0.7685
Ichimura	0.7593	0.7796

[13] The number of inhabitants, population density and the share of local parties deviate for the municipalities outside the 95%-confidence band. Probably the Ichimura specification is especially superior for observations with special characteristics as this specification allows more flexibility.

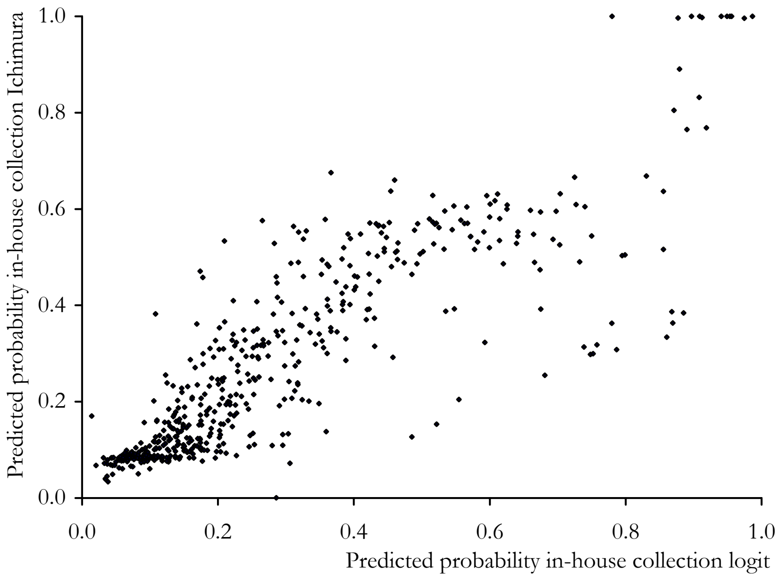
Figure 4.3: Comparison predictions logit versus Ichimura (public versus private refuse collection)



To investigate the consequences of the misfit by logit for a substantial part of our sample, we compare the prediction performances of the models, as well as the estimated marginal effects of changes in the covariates on the probabilities. First, Table 4.3 contains the prediction performances. For the sake of comparison, we also include in this table the naive predictions without using any covariates. We predict the endogenous variable to be equal to one, if the predicted probability is at least a half; otherwise, we predict the endogenous variable as zero.

From this table we conclude that the prediction capabilities of both logit and Ichimura are quite comparable, and that Ichimura only slightly outperforms logit in both the no-private-collection and the in-house-collection cases. Of course, this is only a very rough comparison. To further illustrate how the predictions of both specifications are in line with each other we plot in Figures 4.3 and 4.4 the predictions according to logit against those according to the Ichimura specification. Figure 4.3 contains the comparison for the no-private-collection case. The correlation coefficient between the predictions equals 0.97, but, particularly, at lower values of the single indices, we see a clear difference between both specifications, as already suggested by Figure 4.1, but not reflected in Table 4.3. In Figure 4.4 we consider the in-house-collection case. Here

Figure 4.4: Comparison predictions logit versus Ichimura (in-house versus out-house)



the correlation is much weaker than in Figure 4.3. Indeed, the correlation coefficient is only 0.87, indicating that a blind use of logit may be misleading.

Thus, although the single index of logit corresponds quite closely to the single index according to Ichimura, the logit probability transformation is likely to be misspecified, due to its inflexibility, preventing it from fitting non-monotonic patterns. This might have implications for the quantification of the marginal effects of the covariates on the probabilities of no-private collection and in-house provision. To investigate this, we compare the estimated marginal effects of changes in the covariates on the predicted probabilities. We calculate these effects for each municipality in our sample, and then we average them over the sample. In this way we are measuring the (average) macro-effect of a marginal change in the covariates. Notice that our sample contains almost all Dutch municipalities, so that we are more or less dealing with the whole population. We include the standard deviations of the means to give an indication of the variability of the calculated effects, and we calculate the average of the absolute differences per municipality between the two models, to see how close the effects are. Table 4.4 contains the results for no-private-collection and Table 4.5 presents the results for in-house-collection.

Table 4.4: Marginal effects no-private-collection

Variable	Logit	Ichimura	Abs. Difference
Inhabitants	0.0437	0.0484	0.0269
	(0.0182)	(0.0425)	
Population density	0.0133	0.0135	0.0072
	(0.0053)	(0.0117)	
Funds	0.2413	0.2165	0.1101
	(0.0967)	(0.1871)	
Income	- 0.2815	- 0.2420	0.1223
	(0.1127)	(0.2091)	
Unemployment	0.0037	0.0035	0.0018
	(0.0015)	(0.0030)	
Civil servants	0.0471	0.0377	0.0191
	(0.0189)	(0.0325)	
Conservative Liberals	- 0.0007	- 0.0023	0.0020
	(0.0003)	(0.0020)	
Social Democrats	- 0.0001	- 0.0035	0.0038
	(0.0000)	(0.0030)	
Progressive Liberals	0.0014	- 0.0030	0.0045
	(0.0006)	(0.0026)	
Orthodox Protestants	0.0047	0.0022	0.0026
	(0.0019)	(0.0029)	
Green Left	- 0.0033	- 0.0082	0.0065
	(0.0013)	(0.0071)	
Extreme Right	- 0.0081	- 0.0172	0.0131
	(0.0032)	(0.0148)	
Local Parties	- 0.0041	- 0.0062	0.0040
	(0.0016)	(0.0053)	

Standard deviation in brackets

Looking first at Table 4.4 (no-private-collection), we see in case of, for instance, the output variables (inhabitants or inhabitants per hectare) that the calculated average macro effects are quite comparable between the two specifications. However, in both cases, the average absolute differences are quite large compared to the average marginal effects, indicating that on the individual municipality level the models yield substantial differences, which, on an aggregate level, are averaged out. We also see that the variability in the logit marginal effects is much smaller than the variability in the Ichimura marginal effects, which, of course, is a consequence of the imposed monotonic logit probability transformation, as opposed to the non-monotonic Ichimura probability transformation. In case of the wealth variables and the interest group variables we have a similar story. Looking at the political variables, we see that in some cases the differences between both models are, at least qualitatively, substantial, although the magnitudes of the marginal effects are quite small.

Turning next to Table 4.5 (in-house-provision), we see that the differences are now more

Table 4.5: Marginal effects in-house-collection

Variable	Logit	Ichimura	Abs. Difference
Inhabitants	0.0257	0.0426	0.0314
	(0.0110)	(0.0432)	
Population density	0.0051	0.0136	0.0121
	(0.0022)	(0.0149)	
Funds	0.2092	0.3378	0.2596
	(0.0890)	(0.3702)	
Income	-0.1029	-0.2934	0.2659
	(0.0438)	(0.3215)	
Unemployment	0.0258	0.0109	0.0083
	(0.0067)	(0.0120)	
Civil servants	0.0157	0.0122	0.0084
	(0.0067)	(0.0123)	
Conservative Liberals	0.0001	0.0022	0.0025
	(0.0001)	(0.0025)	
Social Democrats	-0.0063	-0.0066	0.0045
	(0.0027)	(0.0072)	
Progressive Liberals	0.0042	0.0029	0.0022
	(0.0018)	(0.0032)	
Orthodox Protestants	0.0037	0.0024	0.0020
	(0.0016)	(0.0026)	
Green Left	-0.0018	-0.0034	0.0028
	(0.0008)	(0.0038)	
Extreme Right	-0.0270	-0.0721	0.0642
	(0.0115)	(0.0790)	
Local Parties	-0.0041	-0.0020	0.0024
	(0.0018)	(0.0022)	

Standard deviation in brackets

substantial. For instance, in case of the output variables the estimated marginal effect according to Ichimura is between 1.6 (inhabitants) and 2.6 (inhabitants per hectare) times as large as the corresponding effect according to logit.

In case of the wealth variable income per inhabitant the estimated negative marginal effect of income per inhabitant in case of Ichimura is even almost three times as large as in case of logit. The corresponding average absolute differences are also quite substantial. Similarly to the no-private-collection case, the logit marginal effects again show much less variability than the Ichimura marginal effects.

Concluding, we can state that, although the logit single index seems to be appropriate, the logit probability transformation seems to be too inflexible, producing, at least in the in-house provision case, average marginal effects whose magnitudes may be quite incorrect, and resulting in both the no-private collection and the in-house provision cases in an accuracy which may be quite misleading. By applying a semiparametric specification, this inflexibility of the logit probability transformation can easily be circumvented.

So far, we considered no-private-provision and in-house-collection separately. However, one might argue that there may be some ordering present: at level 0 one can consider full privatisation; at level 1 there is public provision, but not in-house; and at level 2 there is full in-house collection. Such an ordering may be modelled by a single index model as well. However, this only makes sense if the two indices, when estimating the choices no-private-collection and in-house-provision separately, are (more or less) the same. Therefore, we also considered the hypothesis that the vectors of coefficients of these two indices are equal. We tested this hypothesis by means of a Hausman-type test based on the difference between the two Ichimura-estimators, after appropriate scaling.[14] The resulting chi-square test statistic yielded as value 36.5, which results in strong rejection of the hypothesis of equal indices, since the critical value of a chi-square distribution with 13 degrees of freedom equals 22.36 (at 5%). We concluded that the modelling of the mentioned ordering by means of a single index is likely to yield a misspecified model, even if modelled semiparametrically. Therefore, we did not investigate this possibility further.

4.6 Conclusions

In this chapter we try to explain the reasons why contracting out refuse collection is less common than in-house provision, although considerable efficiency improvements by contracting out seem achievable. We present an empirical investigation motivated by output arguments, interest group theory, and ideology arguments.

We used both a parametric (logit) and a semiparametric (Ichimura (1993)) modelling approach, which correspond in the use of a single index, but which differ in terms of the flexibility of the probability transformations employed. The estimated single indices are quite similar, so that both yield the same conclusions, when investigating the direction and statistical significance of the various effects. In both models we find evidence for the hypothesis that a high level of transfers by the central government (the wealth argument) or a high level of unemployment (the interest group argument) raises the probability of public and in-house provision. We also find evidence for the assumed relation between the size of municipalities and private collection. In all cases a

[14] The limit distribution of this difference can easily be obtained under the null hypothesis.

smaller municipality is more likely to have private collection. Therefore, scale effects are important for the choice between public and private provision. For the choice between out-house and in-house collection in relation to scale lesser evidence exists. Weak evidence is found for an ideological motivation of this choice.

However, when explicitly quantifying the size of these effects, one also needs the probability transformation, transforming the single index into the probability that the dependent variables equals one. Here, we find strong statistical evidence that the parametric specification is far too inflexible, with the danger that the corresponding estimated marginal effects might be misleading. Indeed, in a number of cases, we find serious differences between the parametric and the semiparametric marginal effects, implying that one should be very cautious, when using parametric models.

Chapter 5

Burn or bury?

5.1 Introduction

Most developed countries, in particular European countries and Japan, have adopted a hierarchical approach to solid waste management, including final waste disposal options. First of all, waste should be reduced, otherwise recycled or reused, next incinerated with energy recovery and, only if nothing else works, landfilled. Landfilling is often considered to be the worst option because it consumes a lot of space, runs a high risk of leakages to air, water and soil and makes less use of the energy content of waste compared with incineration. Incineration is generally thought to produce fewer externalities, in particular in so-called waste-to-energy (WTE) facilities (Miranda and Hale, 1997). These facilities not only reduce final disposal of waste, but also produce electricity and/or heat, saving (energy) resources elsewhere.

Incineration plants, however, also contribute to externalities, such as emissions to air and chemical waste residuals. In addition, they are expensive to build even compared with modern landfills with appropriate prevention of leaking. In fact, the relative performance of incineration depends not only on its own emissions profile but also on the different technological options for landfilling, and all of their associated private and environmental costs, including their recovery characteristics. For instance, methane emissions of landfills, the main source of emissions to air, can be reduced by flaring, and can be used to produce energy as well.

Ideally, the choice between final waste disposal methods requires a systematic comparison of all costs and benefits involved, i.e. a proper social cost-benefit analysis. Obtaining information on individual preferences for final waste disposal facilities, however, is surrounded by difficulties, especially if social costs are included (Miranda and Hale, 1997). Moreover, these individual preferences do not necessarily provide useful information due to free-rider effects such as Not In My BackYard (NIMBY) problems. Consequently, the focus of recent analytical and empirical studies has been on (social) cost comparisons, while (social) cost minimization is a necessary condition for welfare maximization anyway.

For instance, Keeler and Renkow's interesting analytical contribution (1994) analyzes the desirability of incineration relative to recycling and landfilling based on their differences in (marginal) cost. However, this study only includes energy production by incineration, and it neglects the role of environmental externalities. Brisson (1997) is one of the first to extend this framework of analysis to the role of (marginal) social cost, building on a simple linear model of social cost minimization by a government.[1] Empirical work on the performance evaluation of final waste disposal methods, however, is not always adequate from the social cost perspective, and is often disfigured by lack of data. Earlier attempts focus only on environmental risks (e.g. OME, 1999), lack private cost data, apply an asymmetric (extended) private cost analysis only (Keeler and Renkow, 1994), use very rough or incomplete data on (indirect) environmental impacts, or exclude the recovery functions or include them inadequately (Brisson, 1997).

To our knowledge this study is the first to present an encompassing empirical analysis of the incineration versus landfilling including the effects of their recovery functions from a social cost perspective. We have data describing a reasonable set of available (technical) options for each disposal method, as well as on their associated private costs and cost performance in terms of environmental externalities and energy and materials recovery. We present the results from a comprehensive data-set on the average social cost of two 'best-practice' technologies for incineration and landfilling. The data are taken entirely from the Netherlands because they reflect cost estimates of technologies that comply with the strictest waste disposal regulation in the world (see section 5.6).

[1] Unfortunately, her model allows only for interior solutions, implying that a government should always choose a mix of both options. In our opinion, there are no ex-ante reasons for such a limitation. Social cost might also be minimized by choosing one option only. In fact, corner solutions are paramount, given the large number of technical options available, such as including or excluding energy and materials recovery.

Indeed, environmental conditions for final waste disposal are rather poor because the Netherlands is not only densely populated, like Singapore, Japan and some areas in the USA, but also faces pretty bad soil conditions for landfills.[2] If WTE plants were to signal lower social cost than landfilling anywhere, one would expect them to do so in the Netherlands.

This chapter is organized as follows. We first illustrate the existing policy preferences for final waste disposal options in several developed, particularly European, countries (Section 5.2). Next we discuss the characteristics of our social cost-benefit analysis of the choice between landfilling and incineration (Section 5.3). Then we present our results for the best available techniques for the Netherlands (Section 5.4), and we explore the sensitivity of our results in Section 5.5. Section 5.6 analyzes the different arguments in the current policy debate on waste disposal options and how they relate to the choice of technological options, in particular in the European Union (EU). The last section presents some conclusions and discusses further research.

Note that we do not consider the issue of illegal dumping or other forms of non-compliance behavior (see Fullerton, forthcoming), nor do we evaluate issues related to the interaction between the choice of final waste disposal methods and recycling (e.g. Huhtala, 1999). By analyzing the choice between landfilling and incineration, we implicitly restrict ourselves to that part of the overall waste stream where both options are available, namely burnable waste. Therefore, we also neglect issues related to specific waste streams, such as hazardous waste (including radioactive waste).

5.2 Existing waste hierarchies

In most developed countries, in particular within the EU and Japan, there is widespread belief in the previously mentioned hierarchy for waste disposal options. For instance, the EU confirmed this hierarchy in preparing its directives on landfilling and incineration:

> "The 1996 Commission Communication on the review of the Community Strategy for Waste Management confirmed the hierarchy of waste principles established by the Communication of 1989. The principle of prevention of waste generation remains the first priority, followed by recovery and finally

[2] Indeed, Dutch final waste disposal policy has almost exclusively focused on expanding incineration, in particular in WTE plants, over the last decade (see Vollebergh, 1997).

> by the safe disposal of waste i.e. landfilling. In the Community Waste Strategy landfilling represents the option of last resort because it can have substantial negative impacts on the environment. (..) Landfilling as a waste management method has no effect on the prevention of waste and does not make use of waste as a resource, which has a higher priority in the Community Waste Strategy." (COM(97), p.3).

Apparently, the hierarchical approach of the European Commission relies on a rather asymmetric judgment in comparing the different waste strategies. It does include environmental costs for landfilling, but only stresses environmental benefits for the other options. Also, its recent directive on landfilling prohibits flammable waste being landfilled (see COM(99)31), while it remains unclear whether private costs have to play a role in decisions over waste disposal options by Member States.

In contrast, the USA has not had a clear preference for incineration over landfilling for a long time. The Environmental Protection Agency (EPA) also follows a hierarchical approach in its waste policy, with a preference for source reduction, followed by recycling (including composting) and finally by disposal in combustion facilities and landfills. However, the EPA explicitly mentions indifference between the final waste disposal methods:

> "Waste combustion and landfilling are at the bottom of the hierarchy - USEPA does not rank one of these options higher than the other, as both are viable components of an integrated system." (EPA, 1995, p. xxvii)

The use of energy and residual materials, however, is unquestioned as well:[3]

> "When waste generation is unavoidable, the materials can be viewed as a resource from which reusable materials, raw feedstock, minerals, organic matters, nutrients, and energy can be recovered for beneficial uses." (EPA, 1995, p. iii)

Again, however, there is no explicit reference to private cost.

Existing final waste disposal is still dominated by landfilling, even if stated policy preferences suggest otherwise. The first column of Table 5.1 shows that fewer than half of

[3] This view has not changed since (see EPA, 2002).

the EU countries plus Japan incinerate over 50% of their domestic waste. In contrast, Finland, Italy, Spain, the UK and the USA have a very low percentage of incinerated waste, while Greece, Ireland and Portugal incinerate no waste at al. Nevertheless, waste incineration has become more popular in most EU countries, in particular because of its alleged benefit of energy recovery. Over the last decade, newly built waste incineration plants have always included energy recovery facilities, and older plants without energy recovery have been closed. In the early 1990s, on average only a small part of total waste incinerated was with energy recovery, but this percentage is almost 100% in most countries today.[4]

Indeed, WTE plants may appear preferable from a social cost perspective. First, it is often assumed that additional private costs are (very) small compared with the setup cost of an incineration plant without WTE. Second, assuming electricity and/or heat is appropriately priced, selling electricity or heat generates resources for the incinerating company, and therefore lowers the net cost of waste disposal. Third, WTE saves on electricity and/or heat production and its associated externalities.

It is widely thought that one of the major reasons for the dominance of waste incineration in general, and WTE in particular, is scarcity of land in some countries. Table 5.1, however, shows no clear correlation between population density or cultivated land and the percentage of domestic waste incinerated. For instance, the UK and Italy incinerate only small amounts of waste with relatively high population densities and levels of cultivated land.

On the other hand, Sweden, France and Belgium incinerate a lot of domestic waste with much lower land scarcity indicators. Indeed, other environmental concerns, such as leakage to air, water and soil of particular contaminants, including hazardous waste from incineration (dioxins, ash), also influence preferences over waste disposal methods (Menell, 1990; Miranda and Hale, 1997).

In general, incinerating waste is thought to provide the best solution to these problems, in particular if the energy released from the burning of waste is used for electricity production and heating.[5]

[4]The older incineration plants have been closed over the last decade in countries such as Belgium and France due to stricter environmental regulation. Newly built plants are always WTE plants, in line with the EU directive on waste incineration (COM(00)76): 'Any heat generated by the incineration or the co-incineration process shall be recovered as far as practicable'.

[5]We assume that the energy function includes the production of electricity, heat or both.

Table 5.1: Waste incineration characteristics in some developed countries, 2001

	Incinerated waste as a % of total municipal waste	Domesticated land as a % of total land[a]	Population density in people per square km
Europe	33	22	122
Austria	20	43	98
Belgium	55	45	312
Denmark	100	63	126
Germany	72	50	235
Finland	5	9	17
Greece	0	68	82
France	63	55	107
Ireland	0	64	54
Italy	13	53	196
Luxembourg	47	na	166
Netherlands	113[b]	58	466
Portugal	0	43	109
Spain	9	62	79
Sweden	56	8	22
UK	5	71	246
Japan	75	13	336
USA	16	47	30

Sources: Ministry of Finance (2002), World Bank (2002), EPA (2002), WRI (1998).

[a] Figures for 1994.

[b] More than 100% is possible as incinerated waste includes also firm waste, whereas - due to statistical reasons - this waste is not included in municipal waste.

It is still an open question whether modern WTE plants are the best solution ('global optimum') or only provide a local optimum, i.e. only provide a better solution compared to the status quo. That is, given a suboptimal status quo with waste incineration plants without energy recovery, WTE plants probably reduce the social costs of incineration, in particular if additional private costs are small. For example, the prevalence of incineration in Sweden might be explained due to the available network for district heating. However, the overall social cost of final waste disposal may not be minimized if all other options for final waste disposal are considered.

Whether WTE plants are a local or a global optimum depends on the relative performance of those plants compared with, for instance, landfills with energy recovery. To answer this question, it is important to incorporate all dimensions of the different waste disposal options systematically in a social cost-benefit analysis including joint production characteristics of energy and materials recovery as well as the environmental effects associated with all technologies (including landfills with energy recovery).

5.3 The choice between waste disposal options

Our goal is to evaluate existing policy preferences in a general framework that compares the (net) social cost of different incineration techniques with different landfilling techniques. The best option is simply the final waste disposal technology that minimizes net social cost at the margin. Obviously, how much waste should be incinerated and/or landfilled depends on the overall net social cost function, viz. on the marginal cost of landfilling and incineration together (Brisson, 1997). If the marginal social cost of landfilling exceeds the marginal social cost of incineration for a given range of waste to be processed, it would be optimal for the government to incinerate all that waste from a (net) social cost perspective, and vice versa.

The net social costs for final waste disposal methods is the difference between the gross social costs, comprising gross private and environmental cost of a particular disposal option, and the social cost savings associated with their jointness characteristics, i.e the private and environmental cost associated with the energy and material recovery function. First of all, gross private costs split into labor and capital cost for operation and maintenance. Gross environmental costs are restricted to the set of environmental externalities for a specific technology. Both landfills and incineration plants cause substantial negative environmental externalities. In general, landfills are mainly a threat to water and soil systems, whereas incineration contributes to air emissions and chemical waste from burned ashes. Indeed, the set of externalities differs widely across specific technologies.[6] In order to obtain the environmental cost estimates the physical externalities are weighted using monetary estimates of their relative importance or shadow prices (see next section for details).

The social costs associated to the jointness characteristic of the waste disposal methods depend on the amount of energy and material recovery of the final waste disposal technologies taken into consideration. At some additional gross private costs both incineration plants and landfills may recover energy and materials from final waste disposal streams. By choosing a specific technology, the government therefore not only decides on final waste reduction and its environmental cost, but also on the composition of energy production and the amount of materials recycling in the economy. For instance, if a specific technology allows for material recovery, this not only saves private costs

[6] Note that the environmental and private costs are closely linked. For instance, with better abatement equipment (which prevents emissions to air, water and soil), private costs are (usually) higher and environmental costs lower.

in producing these material outputs in the economy but also reduces the gross environmental cost. To what extent investments in the energy and/or material recovery function improve net social cost finally depends on the additional gross private cost involved on the one hand and the private and environmental cost savings elsewhere in the economy on the other hand.

Note that both disposal options may contribute to these joint outputs. Also landfilling is possible with energy and material production, although it is extremely costly to recover materials like aluminum from a landfill. Accordingly, the government faces a wide variety of final waste disposal technologies. At one extreme, there is a modern, best-practice landfill site, which not only generates electricity but also runs a very small risk of leakage. At the other extreme, there is an old-fashioned incineration plant without electricity production and lacking flue gas abatement technologies to prevent air emissions (including dioxin).[7]

As we argued in the introduction, our approach reflects a much wider notion as to how to choose between landfilling and waste incineration than is usually perceived in both theory and policy. For instance, Keeler and Renkow's suggestion (1994, p. 210) that waste incineration has become more favorable in recent years due to the growing attention given to energy recovery is not necessarily true in the more general framework we address.[8] Their claim can be valid only if this option is compared with the social cost characteristics of other technologies, such as a modern landfill site with energy recovery.

Also the view that incineration costs should be measured after subtracting the revenues from energy recovery earned in the electricity market (e.g. Turner, 1992; Brisson, 1997) has to be qualified. First, the WTE technology tends to have an upward, though admittedly small, effect on gross private incineration costs due to the need for some additional capital equipment. Second, and much more important, these private revenues only reflect a net transfer between different consumers, viz. between consumers of electricity and final waste disposal, and this may reflect a highly distorted value (Vol-

[7] In fact, for a given waste facility, we assume the very general production function denoted as f(W, E, G, X(W, E, G), L, K) = 0, with W, E and G representing the useful outputs waste reduction W, energy production E and materials recovery G, X(W, E, G) as the overall environmental externalities produced or saved by each of the useful outputs, and L and K as the usual labor and capital inputs.

[8] If the government selects another waste technology, this not only affects its energy policy indirectly but also its environmental policy. Even if both energy and environmental policies in the energy sector are optimal in the status quo, a shift towards WTE plants would render the environmental policy towards the energy sector suboptimal because emissions may fall below or exceed the existing optimal emissions in this sector.

lebergh, 1997). Therefore, it is more appropriate to measure this contribution by its social value in the energy (and materials) system (including savings on both private and environmental costs). Moreover, the energy recovery potential of landfills should be included in the comparative analysis because energy recovery can be substantial in this case as well (see also the next section).

5.4 Dutch social cost of waste disposal

This section presents a point estimate of the net social cost of two 'best-practice' final waste disposal technologies for incineration and landfilling based on estimates from the Netherlands in 2000. As noted before, the Netherlands is densely populated, providing a prima facie reason for incineration. Indeed, Dutch final waste disposal policy has focused entirely on expanding incineration, in particular in WTE plants, over the last decade.[9] Almost 40% of the waste that could be incinerated, or 2.8 million tonnes of waste, was actually burned in 11 incineration plants in 1995. This percentage has increased to 70% in 2000. In addition, electricity and heat production from waste incineration have grown by 150% and 250% respectively between 1995 and 2000.

Our methodology is to put comparable values for both gross and net private and environmental costs involved. The question as to how different technologies are priced relative to both their gross and net private and environmental costs is operationalized as follows. Our (gross) private cost estimates for waste incineration at a state-of-the-art incineration plant are obtained from a newly built waste incineration plant.[10] These numbers include (large) capital investments in both capacity expansion, with a normal rate of return to capital, and abatement technologies (mainly flue gas scrubbers) required by very strict environmental regulation (see also Section 5.6). The calculations use real figures for the best-practice incineration plant, accounting for cost savings that would apply if the plant were built in a market environment.[11] Based on these assumptions, total private costs equal Euro 103 per tonne for an incineration plant burning about 650 ktonne per annum.[12] At this scale, the installation produces 580 kWh of

[9] Vollebergh (1997) describes institutional details on final waste management in the Netherlands, with special emphasis on the energy potential of WTE plants.

[10] See chapter 6, page 158.

[11] Dutch waste incineration plants currently compete at a national level, and in 2006 even at an international level. The figures have been checked in detail with regulators and plant managers in the Netherlands.

[12] The figure is based on constant total cost per tonne of waste during the operational years. Profit margins are allowed to increase over time when (average) capital costs decrease. We assume a net

electricity and 299 kWh of thermal heat per tonne of waste. Furthermore, 1.6 kg of aluminum and 34 kg of iron per tonne of waste is recovered from the ash.

Gross private cost data for landfilling are much harder to obtain. No reliable figures are available for disposal sites in the Netherlands. Therefore we use estimates from an engineering study on the private cost of a landfill with best available technology (VROM, 1992).[13] The technology considered reflects the private cost of measures against leakage required by Dutch legislation, which is one of the strictest in the world (see also Section 5.6), and includes energy recovery investments to generate 122 kWh of electricity per tonne of waste from landfill gas. Furthermore, the costs for after closure care as well as an insurance against after closure risks are included. We also use the 'worst-case' estimate of Euro 36 per tonne allowing for relative price differentials across the country, such as differences in scale, prices for land use, etc. According to a recent study by the Dutch Ministry of Finance (2002) these figures still reflect the costs of landfilling in the Netherlands accurately.[14]

The savings on the private costs of generating electricity and materials production elsewhere in the economy are simply obtained from private cost estimates for gas-fired power plants in the Netherlands. Electricity production costs were, on average, Euro 0.036 per kWh in the Netherlands in 2000. This reduces the cost by Euro 21 per tonne of incinerated waste (assuming current electricity productivity of the average WTE plant), but only by Euro 4 per tonne of waste for the less energy-productive landfill. Finally, the private (opportunity) costs of materials production are obtained from world market prices for those materials having a positive recycling price, viz. Euro 1.23 per kg of aluminum and Euro 0.02 per kg of iron (Eurostat, 1992).

Table 5.2 summarizes the gross private cost estimates for both final disposal methods and the net private cost including savings from the joint outputs, i.e. recovery of energy and/or materials. The result is straightforward: incineration is much more expensive than landfilling. Although waste incineration contributes to considerable private cost savings elsewhere, these savings do not by any means offset the much higher private

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profit of 12% on invested capital (40% of total capital) for the owner of the plant over the entire time horizon (25 years). The scale size of 650 ktonne is the minimum efficient scale, according to expert opinion.

[13] This somewhat older calculation still provides a reasonable approximation for overall average cost because the same (environmental) regulation applies in 2000 as in 1992 and inflation has been modest. Note also that the impact of specific price changes, such as the much higher farmland price in 2000, is still small and falls within the 'worst case' on which our figures are based.

[14] Note that these costs are somewhat lower than the tariff used in chapter 6 due to the private cost savings which apply only for the most modern landfills.

Table 5.2: Private cost estimates (euro per ton)

	Landfilling	Incineration
Gross Private Costs	40	103
Private Costs Savings:		
- Energy function	-4	-21
- Materials function	0	-3
Net Private Costs	36	79

cost of the incineration plant.

We have also been able to include both gross and net environmental costs using a well-documented comparative engineering study for both waste disposal options, including their energy and materials productivity (CE, 1996). This study produces physical and monetary values for the environmental externalities. First, physical environmental impacts are available for a whole set of life-cycle emissions related to landfilling, incineration, energy and materials production.[15] Second, environmental costs are calculated by weighting these impacts by a set of shadow prices for environmental damage.

Given the strong regulatory constraints on both landfills and incineration in the Netherlands, these estimates (CE, 1996) reflect (expected) emissions of the best final waste disposal technologies currently available.[16] The physical environmental impacts include emissions to air for 47 different substances, such as climate change emissions carbon dioxide (CO_2) and methane (CH_4), and acid rain emissions sulfur dioxide (SO_2) and nitrogen oxides (NO_x). Furthermore, 29 water-polluting substances are included, as well as the amount of chemical waste (such as fly-ash) produced and land use.[17] The estimates for incineration, materials and energy externalities are based on averages for existing plants (CE, 1996). For the estimation of energy externalities, not only is the direct generation of emissions included, but also emissions related to extraction and transport of fuels.

The shadow prices used to calculate environmental costs reflect marginal abatement

[15] Figures for landfilling and incineration are based on the average waste composition in the Netherlands (31% organic. 23% paper, 14% plastics, 7% wood, 6% metals, 3% glass and 16% otherwise).

[16] CE (1996) provides a very detailed analysis of the physical externalities associated with final waste disposal technologies. It still provides up-to-date material because the estimates were based on 'state-of-the-art technology', which - according to expert opinion in the field - has not changed much since 1996 (see also Section 5.5).

[17] According to EU-regulation (EURAL) residues of flue gas treatment waste have to be defined as chemical waste as well as all residues that contain chemical substances. For our calculations chemical waste is defined as the flue gas residue for WTE-plants and as part of the bottom ash and nuclear waste for the energy production plants.

Table 5.3: Environmental cost (euro per ton)

	Landfilling	Incineration
Environmental costs:		
- emissions to air	5.84	17.26
- emissions to water	0.00	0.00
- chemical waste	2.63	28.69
- land use	17.88	0.00
Gross environmental costs	26.35	45.95
Environmental costs savings		
- energy function	-4.21	-22.55
- materials function	-0.00	-5.76
Net environmental costs	22.14	17.64

cost estimates by the Dutch government for physical aspiration levels for emissions to air and water, land use, waste, etc. (measured in emission volumes) for 2010 (relative to 1996). These cost estimates are assumed to reflect current minimum willingness to pay for emissions reduction in the Netherlands. Moreover, they are available for a fairly large set of environmental impacts. The only shadow price we changed was that for land use. CE (1996) uses the (estimated) price of farmland in 1997 (Euro 2 per m^2), but it is hard to see why this price reflects any valuation of externalities related to the siting of a landfill or incineration plant. Instead, we use the price people are willing to pay for the most expensive opportunity forgone, which is the average price of house building land (Euro 227 per m^2) in 2000. Finally, we use market prices for recycled materials such as aluminum and iron.

Both the gross and net environmental costs of the different functions of the waste disposal options are summarized in Table 5.3. Surprisingly, the gross environmental costs for incineration are considerably higher than those for landfilling per tonne of waste. Although land use causes quite high gross environmental costs for landfilling, the much larger emissions to air and disposal of chemical waste for incineration result in even higher environmental costs. In both cases, emissions to water are negligible. The best available technology for landfilling performs quite well, even in the Netherlands, where local conditions are not favorable to this type of technology.

Including environmental cost savings from the energy and materials functions, however, changes the picture considerably. The low efficiency in energy recovery and the absence of materials recycling result in under Euro 5 externality savings per tonne of waste for landfilling. In turn, the typical energy-efficient WTE plant saves over Euro 22 per tonne in the energy production system, and nearly another Euro 6 per tonne in the materials production system. Therefore, the net environmental costs are indeed somewhat lower for incineration than for landfilling.

Together, Tables 5.2 and 5.3 provide the overall social costs of both 'best-practice' technologies in the Netherlands. Social costs for waste incineration amount to Euro 97 per tonne of waste compared with only Euro 58 per tonne of landfilled waste. Even though the environmental cost of incineration is somewhat lower than that of landfilling, it does not outweigh the much larger private cost difference.

In other words, even in a densely populated country such as the Netherlands, incineration seems to be a rather expensive option for disposing of waste. This remains true even if one allows for the joint production of energy and materials. Thus, our point estimate clearly rejects the hypothesis that Dutch WTE plants signal lower social cost than landfilling. In other words, the current policy preference for incineration in the Netherlands is not supported by social cost data in a country where this support is most likely.

The policy preference for incineration probably originates in the overall environmental cost savings, because only incineration with recovery generates less environmental costs than the modern landfill. Net savings are far from substantial and only exist for best-performing WTE plants that also recover materials on a considerable scale. Traditional incineration plants without energy and materials recovery are strongly outperformed by modern landfills.

5.5 Sensitivity analysis

This section discusses the extent to which our conclusions depend on certain assumptions used to calculate our (average) social cost estimate. In other words, we test whether this cost estimate holds within a (much) wider confidence margin that accounts for (marginal) cost differences in either private or environmental cost, or both.

First of all, our private cost estimates depend on the specific regulation required by the Dutch government against emissions and leakage as well as on local siting conditions. The estimates reflect land, labor and capital costs of a completely new landfill or WTE plant with an emission and leakage risk profile that complies with the very strict standards set by the Dutch authorities (see also section 5.6). Sunk costs related to existing installations are therefore assumed away. Obviously, actual siting costs might differ across locations due to differences in local circumstances (e.g. land prices are not uniform). However, we expect differences in scale effects between the options to be limited. Indeed, efficient operation of an incineration plant requires a minimum base load, which is responsible for some lumpiness in the planning of a new facility, but considerable degrees of freedom exist in capacity planning beyond the minimum level. Interestingly, scale effects of landfills are also likely to exist and mainly relate to the increasing cost of siting (Fullerton, forthcoming). Planning just one large landfill saves on siting costs caused by NIMBY problems. Therefore we expect little difference in siting costs between planning a landfill or an incineration plant at a specific location.[18]

Another crucial issue is that the environmental cost estimates are highly sensitive to the shadow prices used. A first observation is that only some emissions to air contribute significantly to the environmental cost in money terms. The main reason is that many emissions to air and soil are small and/or their shadow prices are (very) low. Furthermore, different shadow prices for the environmental costs change our policy conclusion only if they more than compensate the much higher (private) cost of waste incineration. To affect the conclusion, the shadow price for the externalities needs to be more than 10 times higher on average. In other words, given the relatively small absolute difference in environmental cost between incineration and landfilling, shadow prices have to reach a very high level in order to compensate for the high difference in private cost. These levels are unlikely because they are beyond the upper bound of current, alternative estimates.

Restricting our sensitivity analysis to the shadow prices that significantly affect the environmental cost estimates, we observe that our initial prices are within the margins reported in other recent studies (see Table 5.4). Individual levels of the shadow prices responsible for indifference at the margin between incineration and landfilling fall outside the scope of these studies. For instance, a so-called turning point (TP), i.e. the price level that would shift our conclusion, could only be obtained with a shadow price for methane (CH_4) that is 7.8 times higher than currently assumed, and the shadow price for NO_x would have to be even 31.3 times higher than assumed currently. Note also that the shadow price for CO_2 should be lower than currently assumed in order to

[18]Assuming the (gross) local environmental cost estimates have some value in signaling potential NIMBY problems, siting costs for incineration and landfilling are indeed quite close. Chemical waste of incineration plants is usually not a local problem, but its final disposal (at landfills or in closed salt mines) causes potential NIMBY problems elsewhere.

Table 5.4: Shadow prices (euro per ton emissions)

	CO_2	CH_4	SO_2	NO_x
This study				
- initial	34	379	4,701	3,291
- turning point	-493	2,965	54,109	103,119
Fankhauser (1992)	138	617		
CSERGE (1993)			592	456
ORNL (1995)			1,191	5,162
Externe (1997)	4-140	45-1,583		1,494-15,731
EU (2000)	4-42	53-2,223	2,100-12,200	4,300-18,340
IPCC (2001)	20-135			

produce a TP.

Using the shadow prices of the six different studies presented in EU (2000), we only find significant changes for one study. This study exploits extremely high shadow prices for all emissions, and, more important, a shadow price for CH_4 relative to CO_2 that is 6 times higher than is common in the literature. Because higher shadow prices for CH_4 are unfavorable for landfilling and higher prices for CO_2 are unfavorable for incineration, the environmental cost difference is much smaller. A TP, however, is not even reached in this case either.

Furthermore, our social cost estimate is quite insensitive to the shadow price for chemical waste and dioxins. Note, first of all, that a higher price for these externalities would only strengthen our conclusion that incineration is unattractive. The disposal of chemical waste produced by incineration plants dominates the environmental cost estimate of incineration.[19] A slightly higher level of the shadow price of chemical waste (15% higher) renders incineration unattractive even on environmental cost alone. The external costs of dioxin emissions under the Dutch regulatory regime are now nearly zero. Using the highest shadow price for dioxins reported in EU (2000) - Euro 713 million per TEQ - results in total environmental costs of only Eurocent 21 per tonne of incinerated waste. Neither is a TP reached using a lower price for chemical waste. Even with a zero price landfilling is still the preferred option because less environmental costs are saved by the energy function of the WTE plant with a lower price for chemical waste.

The shadow price for land is mainly responsible for the environmental costs of landfilling.

[19] Note that in the Netherlands fly ash is 100% reused as building material for roads. Without this option the preference for landfilling is even further increased.

As noted before, our price is based on a relatively high opportunity cost estimate in order to reflect scarcity-of-land considerations (Euro 227 per m^2). Less densely populated countries with more space available more likely face a lower shadow price for land. If this price falls below Euro 176 per m^2 or 22%, landfilling is also the best option on environmental costs alone. Growing land scarcity, however, may increase the price of land even beyond our estimate - that already accounts for highly populated areas -, but a TP is reached only with a landprice of Euro 716 per m^2 or 315% above its current price.

Our calculations are also quite sensitive to the computation of the cost savings of the energy function in particular. First of all, private cost savings from energy recovery largely depend on the level of the energy price, in particular the electricity price. Higher electricity prices would render waste incineration more profitable at the social cost margin (ceteris paribus) than landfilling if the electricity price were 3.3 times the current price (or Eurocent 12.2 per kWh). However, such a price is unlikely because competition in the EU electricity market will force electricity producers to choose the cheapest production facilities (see Paffenbarger and Bertel, 1998, p. 7). According to this study, Dutch (expected) prices are also close to those in the most expensive EU country.

Furthermore, our calculation is based on a gas-using electricity plant because this type of plant delivers the marginal unit of electricity in the Netherlands.[20] Other technologies would affect savings on both private and environmental costs. For instance, a more expensive marginal electricity plant producing fewer emissions, such as a wind farm or hydroelectric plant, would strengthen the case for landfilling if the additional environmental cost savings were smaller than the additional private cost savings. Calculations with a coal-fired electricity plant as the reference technology (lower private, but higher environmental, cost savings) reduce the difference in social cost between the options, but still leaves landfilling cheaper (Euro 54 for landfilling and Euro 81 for incineration). Indeed, the net environmental costs only fall to Euro 18 for landfilling, but they decline to Euro 2 for incineration. Current climate change policy within the EU, however, weakens the case for using coal-fired power plants as the reference technology in the near future.

Finally, the cost savings of the energy function might be higher in countries that could

[20] Paffenbarger and Bertel (1998) show that this is a reasonable assumption for the long term. While in a deregulated market a wide set of other (cheaper) options is available on short term, this is not the case on long term when supply equals demand.

exploit the heat produced by waste incineration on a larger scale. Our estimates reflect the fact that in the Netherlands only a small part of the heat can be used, as nearly no infrastructure is available to deliver the heat to consumers. For countries that may exploit an existing infrastructure for heat delivery, heat production might be six times as high as assumed for the Netherlands.[21] Accordingly, environmental cost would decrease nearly 10 euro per tonne of waste rendering waste incineration preferable from an environmental point of view. This remains true even if we account for heat production by landfills, which is only 2 euro per tonne due to the lower heat contents. Even with this additional improvement of the energy function of WTE plants, this is insufficient to bridge the gap in private cost between landfilling and incineration.

Summarizing, our conclusion that landfilling is preferable at the margin is fairly robust. Although WTE plants are certainly preferable to the older incineration plants, the strong emphasis on incineration in Dutch waste policy is not supported by our calculations. We conclude that the social cost estimates suggest that the Dutch government could reduce the social cost of waste disposal at the margin by expanding landfills.

5.6 Policy choices and consequences

Our analysis started from the observation that waste policy within the EU is strongly founded on the perception that WTE is preferred to landfilling. The data on the Dutch case only provide some support for this policy preference: WTE plants perform better than modern landfills only if one restricts the analysis to net environmental cost, and the difference is small. The net private costs, however, are so much higher for incineration that landfilling is the social cost minimizing option at the margin. This is a surprising conclusion and challenges current policy preferences. This section evaluates these preferences and their implications in more detail.

One important issue is whether our estimates for the best-practice technologies complying with Dutch final waste disposal standards have any wider applicability, in particular within the EU. Wider applicability depends on i) the regulation by the government, and ii) local siting conditions. The regulatory conditions imposed by the Netherlands have been an important focal point in drafting the EU directives on incineration and

[21] Dutch WTE plants do not operate at full energetic capacity because the main part of heat production is not used. Note that in this particular case expanding heat production is possible without reducing the production of electricity.

Table 5.5: Current environmental regulation in EU-countries

	Dioxine (ng/m^3)	*Dust* (g/m^3)	*CO* (g/m^3)	*HCl* (g/m^3)	SO_2 (g/m^3)	NO_x (g/m^3)
Ireland	A	A	A	A	A	No
Italy	A	A	A	A	A	No
Sweden	A	A	A	A	A	A
Belgium	B	A	A	A	A	Na
Denmark	B	A	A	A	A	No
Spain	B	A	A	A	B	B - 120
UK	B + 0.9	B + 15	B	B	A	B - 50
Austria	B	B + 5	B	B	B	B - 100
Germany	B	B	B	B	B	B
France	B	B - 5	B	B	B - 10	B - 130
Netherlands	B	B - 5	B	B	B - 10	B - 130
A=COM(89)369	No limit	30	100	50	300	No limit
B=COM(00)76	0.1	10	50	10	50	200

Source: Chapter 6

landfills.[22] For instance, Table 5.5 shows that the recent European standards for incineration plants are close to the Dutch ones, except for dust and acid rain emissions, in particular for NO_x. As noted before, our estimates of the social cost characteristics of both disposal options reflect compliance with these strict rules. According to industry experts, only the stricter NO_x limit increases private costs (by Euro 5 per tonne of incinerated waste, see Ministry of Finance, 2002). Without this limit environmental costs are raised by almost Euro 3 per tonne. In other words, the difference in standards only slightly influences the general preference for landfilling. So we conclude that the emissions profile of the technologies we consider is such that they are properly called 'best practice', given the current EU standards, and might be applied everywhere in the EU.

A second issue is that the estimated difference in social costs would certainly be different in other countries due to differences in siting conditions (assuming similar standards). Note, first of all, that local circumstances are particularly unfavorable to landfills in the Netherlands. Indeed, the included strict measures against leaking (such as installing thick plastic linings along the base, collecting and treating leachate and monitoring groundwater) raise the private costs of landfills considerably. Other siting locations with a much lower risk of leaking (e.g. on hard rock) might be exploited at much lower social

[22] These directives are published as COM(00)76 and COM(99)31. Countries should have started to comply with the directives in 2000, but they are allowed a transition period of five years, i.e. until 2005 to reach full compliance.

cost per tonne of landfilled waste, which would further strengthen the case for landfilling. Private cost differences for incineration with similar standards are much less likely to be affected by local ecological circumstances, and amount to differences in wages and user cost of capital across countries.[23] Note, furthermore, that our environmental cost estimate for landfills is likely to be high judged by European standards. This is not because the Netherlands would apply stricter requirements than the EU-regulation for landfilling - they are quite similar - but because of the high shadow price for land use. This price would be (much) lower in less populated areas in Europe and as noted in the previous section, a 22% lower shadow price for land would render landfilling cost-efficient from the environmental perspective alone.

What lessons can be drawn from our estimates for the Netherlands given their wider applicability? First of all, we find strong support for both recent EU and US policy to focus on energy recovery in improving the environmental performance of final waste disposal. This policy should certainly also include landfills. In most countries, energy production from landfill gas is not yet common. It has been estimated that the proportion of waste in sites with gas control ranges from as little as 10% in Greece to 90% in the UK and Germany (see Smith et al., 2001, p. 98). Also, in the USA, only 10% of landfill sites produced electricity in 1997 (EPA, 1999). Methane recovery from a landfill is likely to save on overall private cost and certain to save on environmental costs. Without gas recovery and flaring, the net environmental costs of a landfill nearly double, to Euro 43 per tonne.[24] Although energy recovery from landfills does not reduce gross environmental costs, it is certainly important in reducing net environmental cost at a relatively small investment. According to Miranda and Hale (1999), the private costs of introducing gas recovery at a landfill are less than Euro 3 per tonne of waste, while the private cost savings are equal to Euro 4 per tonne (see Table 5.2). Further improvements in the energy efficiency of waste incineration, however, are not supported by our data. Using a larger boiler with higher steam pressure and machinery for reheating the steam in a so-called high-efficiency incineration plant would improve electrical

[23] Comparing private cost differences between landfilling (based on observed prices including tax) and incineration (based on assumed differences in (average) wages and the corporate income tax) across the EU reveals that all EU countries except Austria face a larger wedge than the Netherlands (see Chapter 6).

[24] For our landfill 78 m^3 landfill gas per ton of waste is exploited, 36 m^3 is flared and 34 m^3 is emitted. A landfill site with 100% gas emission (without energy recovery and flaring) would result in 4.3 times higher emissions to air. Note that we exclude externalities such as stench. Interestingly, stench associated with landfills is correlated to methane emissions, and therefore its nuisance is also strongly reduced by energy recovery on landfills.

efficiency from the conventional 20% to 30%. The additional capital cost (including the lower availability of the plant due to more maintenance) would increase net private cost by Euro 15 per tonne of waste in the Netherlands, but the net environmental cost would be only Euro 11 per tonne of waste lower (Dijkgraaf, 2002).[25]

Second, our social cost evaluation casts serious doubts on the hierarchical preference with respect to incineration over landfilling. According to our social cost evaluation, this hierarchical approach would only be justified if the marginal social cost of incineration were smaller than that of landfilling across the whole range of disposable (and flammable) waste. Our results do not provide any evidence for this preference; rather, the opposite is true, i.e. landfilling should be the preferred option. However, a point estimate is not entirely suited to being applied across the whole range of disposable waste, although our sensitivity analysis supports wider applicability. The suboptimality of the hierarchy at the margin is based on the assumptions that both existing and new facilities comply with the strong EU regulation, and that shadow prices remain constant over the entire waste disposal stream. This is a useful starting point for the Netherlands today, but Table 5.5 also shows that many countries within the EU apply laxer standards. Moreover, even for the Netherlands, it would be rather naive to expect that shadow prices would remain constant if all waste were landfilled in the future.

Indeed, current standards differ considerably across EU Member States, and this also affects both private and environmental costs of existing facilities. For instance, in 1999, according to the French Ministry of Environment all 72 French incinerators for which data were available produced more dioxins per tonne of waste than our Dutch plant, while 26 even produced 10 times as much.[26] One expects not only lower private costs, but also much higher environmental costs, for these plants. Assuming private costs are lower for these French incineration plants than for the best available incineration technology, one also expects waste migration towards this less costly alternative.

Evaluating optimal standard setting within the EU is beyond the scope of this chapter, but this example clearly illustrates that our general findings only hold for best available technologies complying with a particular set of standards.[27] Indeed, a large number

[25] Not surprisingly, the Dutch Waste Processing Association (VVAV) prefers an energy efficiency of 20% for new waste incineration plants in the case of commercial operation.

[26] www.environnement.gouv.fr/actua/cominfos/dosdir/DIRPPR/dioxine/tablaitdioxines.htm

[27] Note that we do not have to assume optimal standard setting. Such standards imply that the higher private cost of abatement equipment is outweighed by the savings on additional environmental costs. Our social cost analysis, however, also applies if the government selects suboptimal standards.

of French incinerators will be closed, given the newly established legislation in France, which is similar to Dutch standards (see Table 5.5). However, several countries across Europe only comply with the 1989 directive, and still have to adapt to the latest directive. Similar observations hold for landfills. Indeed, the current EU efforts to harmonize emission standards for both incineration plants and landfills may be considered essential in creating a fair level playing field.[28]

It is essential to separate carefully issues of (optimal) standard setting from the (optimal) final waste disposal strategy, even though these issues are closely connected. Our findings are valid for the strong Dutch incineration and landfill standards. Compliance with these standards might impose high costs on non-complying countries. These costs may even be excessive if the focus is on expanding incineration based on these new standards, because a modern landfill provides a serious, much cheaper alternative. To use the French example again, with a shadow price for dioxins of Euro 713 million per TEQ, 20 incineration plants produce even higher environmental costs than our landfill. Apparently, waste incineration plants without dioxin abatement equipment are outperformed by our modern landfill from an environmental cost perspective alone. The choice of the French government to invest in incineration conforming to EU policy instead of investing in modern landfills imposes a huge burden on French society.

The (implicit) assumption of constant marginal cost behind our point estimate is rather strong, and implies that landfilling is the preferred solution for all disposable waste now and in the future. However, it is unlikely that consuming more land for the purpose of landfills would not change shadow prices (but note that a continuing stream of the much more polluting chemical waste from incineration is not likely to keep shadow prices constant either). Nevertheless, we think that a less rigid approach, such as that in the USA, also seems optimal for the EU. Many useful spots for landfills are still available, while standards are already set and only require appropriate monitoring. A common waste market without further restrictions is likely to generate a more efficient solution for waste disposal.

[28] Note that a set of minimum standards would be equally justifiable. Accordingly, Member States might impose stricter local standards allowing for differences in preferences across the Union.

5.7 Conclusions

From a social cost minimizing perspective, we find little support for a hierarchical approach towards final waste disposal methods. Our average cost estimate of the two best available options in the Netherlands indicates much higher gross environmental cost for a WTE plant than for a modern landfill that also produces energy. Only if the current energy system is rather polluting are WTE plants attractive relative to landfills from an environmental cost perspective alone. Certainly in countries such as France and Belgium, with their much lower climate change emissions from the electricity sector, WTE plants are a very expensive climate change abatement option. The net private cost of WTE plants is so much higher than that for landfilling that it is hard to understand the rationale behind the current hierarchical approach towards final waste disposal methods in the EU. Landfilling with energy recovery is much cheaper, even though its energy efficiency is considerably lower than that of a WTE plant.

To illustrate, according to our calculations, switching from a modern landfill to a WTE plant as required by the hierarchical approach would save 508 kg of CO_2-equivalent, or Euro 17 per tonne of waste, while the social cost difference between incineration and landfilling is Euro 39. This is equal to an abatement cost of Euro 77 per tonne of CO_2-equivalent, which is over 3 times higher than other options available (see Hendriks et al., 2001). Even slightly higher savings can be reached by switching from a landfill without flaring or energy recovery to the best available landfill technique including energy production. This saves almost 40 kg of CH_4 or 585 kg of CO_2-equivalent or Euro 20 per tonne of waste, at a modest additional private cost.

Thus our analysis casts serious doubts on the current policy preference for incineration over landfilling. WTE plants reduce the net social costs of final waste disposal only if waste incineration without energy recovery is applied already or infrastructures for the use of heat exist in the status quo. Because energy recovery is a joint product obtainable with small additional investments in technology, not only will the net private cost be lower due to revenues raised from selling electricity or heat, but also the environmental cost of the energy system. Our calculations however, also suggest that, within a reasonable confidence margin for the shadow prices used, the overall environmental cost savings from WTE compared with landfilling are small in a proper environmental impact analysis, and they are certainly unlikely to compensate for the large differences in private costs.

One important caveat remains. This study extensively uses the basic idea of putting a price on the environment by using shadow prices that depend on revealed policy preferences of the Dutch government with respect to different environmental issues. Although this yields prices within the margins used in other studies, there is always the risk of not using appropriate estimates of either the physical or monetary impact of the environmental externalities to date. Indeed, our calculations depend on a number of other assumptions as well, including the waste composition of Dutch final waste stream and existing knowledge with respect to damage of chemical composites and (potential) leakage effects. Also issues related to (social) discounting are not discussed. One has to realize, however, that different prices always affect both options because the comparison of the environmental costs of different options follows from their relative environmental performance for a given set of shadow prices.[29]

Finally, further empirical research is necessary to compare our calculations for the Dutch case with other techniques employed in practice at other locations. For instance, most landfills do not recover energy, and leaking is often a serious problem, especially for older landfill sites. These types of landfills are even less costly to exploit than our modern site, but their environmental costs can be prohibitive. Therefore, proper treatment of and energy recovery from landfills are the more important targets for waste policy.

[29] We do not impose any assumption on optimal policy - that is, on the government setting the optimal shadow price by weighing costs and benefits of final waste disposal methods, or disposal versus recycling, for instance.

Appendix A Calculation environmental costs

In this appendix we explain the method used by CE (1996) to calculate external cost estimates used in our comparative study. If not otherwise indicated, figures are taken from CE (1996).

1. First, the report defines externality sources: emissions to air, emissions to water, production of chemical waste, land use and recycling of energy and materials. All these externalities are jointly produced on the landfill and incineration plant (refer to figure 5.1).

2. Second, substances are defined within each externality source. The physical environmental impacts include emissions to air for 47 different substances, such as climate change emissions, like carbon dioxide (CO_2) and methane (CH_4), and acid rain emissions, like sulfur dioxide (SO_2) and nitrogen oxides (NO_x). Furthermore, 29 water polluting substances are included, as well as the amount of chemical waste produced (like fly-ash), and land use.)

3. Third, emissions are taken from national Environmental Impact Studies, produced by the Dutch Central Waste Agency. For incineration plants, landfills, energy production plants and material producing plants, average emissions for existing plants are measured. Physical data for the most important externalities (in terms of environmental costs) are given in Tables 5.6 to 5.17.

4. Fourth, shadow prices for the different externalities are based on local abatement cost calculations for Dutch environmental policy goals for 2010. These policy goals are usually operationalized as marginal abatement cost estimates for physical aspiration levels of the government for emissions to air and water, land use, waste, etc. (measured in emission volumes). For example, the set of measures chosen by the government to reach the Kyoto-goal determines the shadow price for CO_2. When the chosen measures are ordered according to their cost level, the most expensive measure equals the marginal costs and thus the shadow price of a unit change in CO2-emisssions. Shadow prices are available for a rather large set of environmental impacts, as these aspiration levels include most environmental policy issues. As noted in the main text, the CE (1996) study includes a quite implausible shadow price for land use, as their evaluation is based on the current price of farm land (2 euro per m2). In contrast, we include an estimate based

on the average price of house building land (227 euro per m2), as this price reveals the (average) maximum people are willing to pay for land in general. Table 5.18 summarizes the shadow prices used for the most important (in terms of environmental costs) externalities.

5. Finally, physical emissions are multiplied by the shadow prices to arrive at environmental costs (see Tables 5.6 to 5.17).

Finally, we give a short description of the way we calculated the saved environmental costs in the regular energy system as a different mix of electricity and heat production compared with incineration and landfilling complicates the calculations. As the procedure is exactly equal for landfilling and incineration we describe here only the incineration case. Our incineration plant produces 580 kWh electricity and 299 kWh heat per tonne waste. Thus each tonne waste incinerated saves the environmental costs of an electricity plant producing 580 kWh electricity and 299 kWh heat. Given the shadow prices for externalities, we thus need to know the environmental impact in physical quantities of an electricity plant producing 580 kWh electricity and 299 kWh heat. Table 5.13 presents these figures for emissions to air. A regular electricity plant emits for example 503.03 gram CO_2 per kWh electricity. Thus, 292 kilogram CO_2 (580 * 0.503) is saved when an incineration plant produces 580 kWh electricity. However, the true savings are higher. This is the results of (i) an extra saving of 102.12 gram per kWh electricity as the incineration plants produces more heat per unit electricity than the regular electricity plant, (ii) an extra saving of 64.86 gram per kWh electricity due to less winning, generation and transport of fossil fuels used by the electricity plant and (iii) an extra saving of 3.39 gram per kWh electricity due to less use of CaO in the electricity plant. In total 390.57 kg CO_2 ((503.03+102.12+64.86+3.39)*580) is saved when the incineration plant produces 580 kWh electricity and 299 kWh heat.

Figure 5.1: Waste disposal systems and external effects

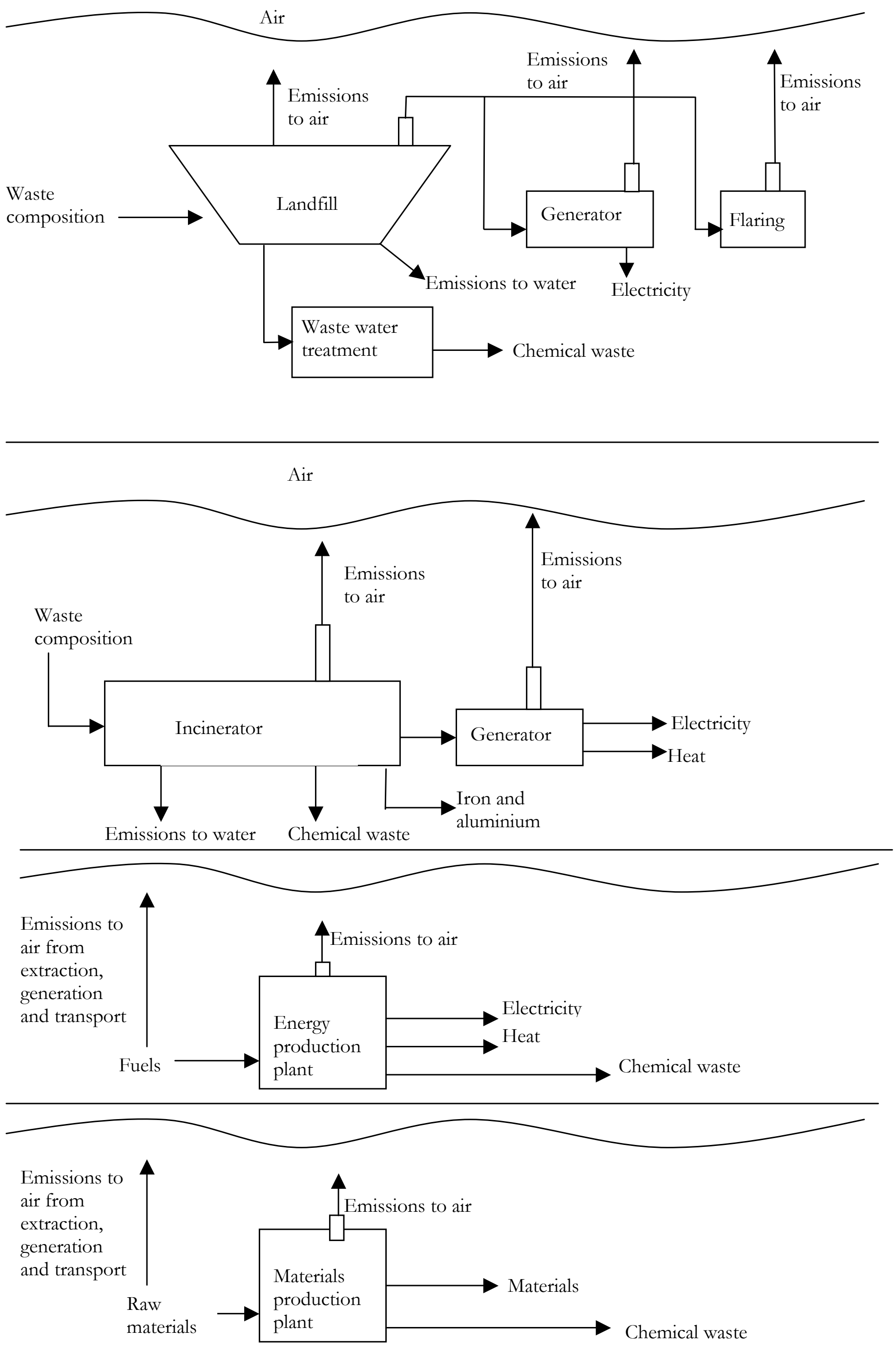


Table 5.6: Landfill gas production

Source	m^3 per year
Total landfill gas production from one ton landfilled waste[a]	148
- Exploitation[b]	78
- Flaring	36
- Emissions	34

[a] Total production is estimated with the following formula: $Gas = 1.86 * a * CO * k1 * e^{-k1*t}$. With a = 0.58, CO = 143, k1 = 0.094.

[b] Assumed first 2 years 50% and 13 years 75% exploitation and from the 16th year 100% flaring.

Table 5.7: Energy production from landfill gas

1	Exploited landfill gas	78	m^3/ton	see Table 5.6
2	Methane	24	mol/m^3	given
3	Heating value CH_4	0.8	MJ/mol	given
4	Heating value landfill gas	19.2	MJ/m^3	row 2 times 3
5	CH_4 percentage gas	57	%	given
6	CO_2 percentage gas	43	%	given
7	Electrical conversion yield	32	%	given
8	Heating value	1488	MJ/ton	row 1 times 4
9	Generation net electricity	476	MJ/ton	row 7 times 8
10	Delivered	438	MJ/ton	92% of 9
11	Delivered	122	kWh/ton	1 kWh = 3.59 MJ

Table 5.8: Landfilling land use; physical values, prices and costs

	Quantity	Price (euro per m^2)	Costs (euro per ton waste)
Gross surface (m^2)	787500		
Landfill capacity (m^3)	10000000		
Density (ton/m^3)	1		
Surface per ton (m^2/ton)	0.08	227	18

Table 5.9: Landfilling: externality physical values, prices and costs

	Emissions in kilogram per ton waste	Price in euro per kilogram	Costs in euro per ton of waste
Air			
CO_2	0.00	0.034	0.00
CH_4	13.30	0.379	5.04
SO_2	0.00	4.701	0.00
NO_x	0.24[a]	3.291	0.80
Water			
As	$4.8 * 10^{-6}$	0.258	0.00
Cd	$1.44 * 10^{-7}$	0.002	0.00
Cr	$9 * 10^{-6}$	0.010	0.00
Cu	$2.4 * 10^{-6}$	0.006	0.00
Ni	$6 * 10^{-6}$	0.003	0.00
Pb	$2.7 * 10^{-6}$	0.180	0.00
Hg	$9 * 10^{-7}$	0.588	0.00
Chemical waste	2	1.316	2.63

[a] Including 0.24 indirect emissions (0.01 for flaring and 0.23 for use of gasmotor).

Table 5.10: Heating value waste

	Part in total waste (%)	Heating value in GJ per ton: Component	Total
Paper and cardboard	22.7	10	2.3
Wood	6.9	14	1.0
Synthetics	13.6	33	4.5
Iron	5.4		
Non-iron	0.7		
Glass	2.8		
Organic	31.0	3	0.9
Rocky	0.8		
Other combustable	16.1	10	1.6
Total	100		10.3

Table 5.11: Energy and materials recovery incineration

1	Heating value	10364	MJ/ton	see Table 5.10
2	Energy contents	2851	kWh/ton	1 kWh = 3.59 MJ
3	Electrical conversion yield	20.35	%	given
4	Delivered electricity	580	kWh/ton	row 2 times 3
5	Thermic conversion yield	10.5	%	given
6	Delivered heat	299	kWh/ton	row 2 times 5
7	Recycled iron	34	kg/ton	given
8	Recycled aluminum	1.6	kg/ton	given

Table 5.12: Incineration; externality physical values, prices and costs

	Direct emissions in kg per ton waste	Price in euro per kilo	Environmental costs in euro per ton waste
Air			
CO_2	447.89	0.034	15.45
CH_4	0.03	0.379	0.01
SO_2	0.20	4.701	0.96
NO_x	0.25	3.291	0.84
Water			
As	$5 * 10^{-8}$	0.258	0.00
Cd	$5.2 * 10^{-6}$	0.002	0.00
Cr	$4.5 * 10^{-6}$	0.010	0.00
Cu	$5.5 * 10^{-6}$	0.006	0.00
Ni	$2.1 * 10^{-5}$	0.003	0.00
Pb	$1.2 * 10^{-5}$	0.180	0.00
Hg	$7.2 * 10^{-8}$	0.588	0.00
Chemical waste	21.80	1.316	28.69

Table 5.13: Environmental opportunity cost emissions to air energy function; Incineration

	Electr. g/kWh	Heat[a] g/kWh	Winning, generation, tranport g/kWh	CaO g/kWh	Total[b] kg/ton	Price euro/kg	Costs euro/ton
CO_2	503.03	102.12	64.86	3.39	390.57	0.034	13.47
CH_4	0.00	0.00	3.28	0.01	1.87	0.379	0.72
SO_2	0.19	0.00	0.23	0.00	0.24	4.701	1.16
NO_x	0.62	0.00	0.08	0.00	0.40	3.291	1.34

[a] As the average incinerator produces relatively more heat than the average energy plant, a heat production boiler is added to make the two energy producers comparable.

[b] Assumed 1 ton waste produces 580 kWh electricity and 299 kWh heat (see Table 5.11).

Table 5.14: Environmental opportunity cost chemical waste energy function; Incineration

	Production gram/kWh	Re-use %	Total[a] kilo/ton	Price euro/kilo	Costs euro/ton waste
Bottom ash	15.3	50	4.34	1.316	5.84
Fly ash	14.4	100	0.00	1.316	0.00
Plaster	6.75	100	0.00	1.316	0.00
Mine waste	87.8	0	49.78	0.00	0.00
Nuclear waste	0.021	0	0.01	1.316	0.02

[a] Assumed 1 ton waste produces 580 kWh electricity and 299 kWh heat (see Table 5.11).

Table 5.15: Environmental opportunity cost emissions to air energy function: Landfilling

	Electricity gram/kWh	Winning, generation, tranport gram/kWh	Extra CaO gram/kWh	Total[b] kilo/ton	Price euro/kg	Costs euro/ton
CO_2	478.50[a]	64.86	3.39	66.70	0.034	2.30
CH_4	0.00	3.28	0.01	0.40	0.379	0.15
SO_2	0.19	0.23	0.00	0.05	4.701	0.24
NO_x	0.62	0.08	0.00	0.09	3.291	0.28

[a] As the average landfill site produces no heat, the emissions related to the heat production of the energy plant are excluded.
[b] Assumed 1 ton waste produces 122 kWh electricity and 0 kWh heat (see Table 5.7).

Table 5.16: Environmental opportunity cost chemical waste energy function: Landfilling

	Production gram/kWh	Re-use %	Total[a] kilo/ton	Price euro per kilo	Costs euro per ton
Bottom ash	15.3	50	0.93	1.136	1.23
Fly ash	14.4	100	0.00	1.136	0.00
Plaster	6.75	100	0.00	1.136	0.00
Mine waste	87.8	0	10.71	0.00	0.00
Nuclear waste	0.021	0	0.00	1.136	0.00

[a] Assumed 1 ton waste produces 122 kWh electricity and 0 kWh heat (see Table 5.7).

Table 5.17: Environmental opportunity cost emissions to air materials function: Incineration

	Total[a] kilo per ton	Price euro per kilo	Costs euro per ton waste
CO_2	51.01	0.034	1.76
CH_4	0.11	0.379	0.04
SO_2	0.79	4.701	3.71
NO_x	0.08	3.291	0.25

[a] Assumed 1 ton waste produces 34 kg iron and 1.6 kg aluminium.

Table 5.18: Shadow prices externalities

	Unit	Price in euro per unit
Emissions to air		
CO_2	kilogram	0.034
CH_4	kilogram	0.379
SO_2	kilogram	4.701
NO_x	kilogram	3.291
Emissions to water		
As	kilogram	0.258
Cd	kilogram	0.002
Cr	kilogram	0.010
Cu	kilogram	0.006
Ni	kilogram	0.003
Pb	kilogram	0.180
Hg	kilogram	0.588
Waste		
Chemical	kilogram	1.316[a]
Bottom ash	kilogram	1.316[a]
Mine	kilogram	0.00
Nuclear	kilogram	1.316[a]
Land-use	m^2	227

[a] The equal prices for chemical waste (a residue of incineration) and bottom ash (a residue of the energy plant) are based on the fact that in the Netherlands these wastes are treated in the same way for the part that is not re-used (note that the part that is re-used has actually a shadow price of zero). The price for nuclear waste is a guess. In the main text we show that these assumptions do not influence our conclusions.

Appendix B Model

In this appendix a simple static model is given which makes clear what the essential elements are in estimating the social costs of landfilling (L) and incineration (I). Assume the government minimizes overall social cost of the treatment of total waste (W_0).

The social costs of landfilling depend on the specific combination of both the private costs of landfills $L_P(W_L)$ and the environmental costs $L_M(W_L)$ for the jointly produced outputs together. The private costs of landfilling include collection and hauling costs, labor and capital costs for operation and maintenance, as well as some savings, which reflect precautionary care for leakage in the future. Note also that these private costs include the cost of investments in energy production, such as an installation for landfill gas, while there is no opportunity for ex post recycling of materials. Energy production $E_L(W_L)$ is assumed to be a linear function of the amount of landfilled waste ($E_L^{'} > 0$ and $E_L^{''} = 0$). We furthermore assume rising marginal private costs ($L_P^{'} > 0$ and $L_P^{''} > 0$), as transport costs usually rise if withdrawal areas become larger. Finally, environmental costs related to landfilling depend on the landfill technology applied. As we do not explicitly model technology choice for landfilling, the environmental costs (savings) are simply assumed to be additive with private costs, following the well-known pattern of $L_M^{'} > 0$ and $L_M^{''} > 0$.

The assumptions on the social-cost components of waste incineration, the private costs $I_P(W_I)$ and the environmental costs $I_M(W_I)$, show a similar pattern as those applying to landfilling with the exception of the recovery of materials. Like landfilling, incineration contributes to energy production, $E_I(W_I)$. Unlike landfilling, however, materials, $G_I(W_I)$, such as aluminum, can be recovered from the ash after incineration and therefore contribute to additional social savings as well. Again, additional private costs for both energy and materials recovery are included in the private costs of the waste incineration plant. For recovery of materials and energy production from incineration, the same curvature characteristics apply as for energy production from landfills ($G_I^{'} > 0$ and $G_I^{''} = 0$ and $E_I^{'} > 0$ and $E_I^{''} = 0$). Thus we have a private incineration cost function with $I_P^{'} > 0$ and $I_P^{''} > 0$, 0 as well as an environmental cost function with $I_M^{'} > 0$ and $I_M^{''} > 0$.

The government must decide how much waste should be incinerated or landfilled. In so doing, it minimizes the social cost for jointly produced energy and recovered materials (including the environmental costs related to all three subsystems, viz. waste disposal as well as savings on energy and materials production elsewhere). If the government chooses not to produce energy or to recycle materials through its waste disposal system, if has to meet these demands elsewhere in the economy. Therefore we include the following constraints taking appropriate notice of the potential jointness characteristic of a given technology: (i) total processed waste W_0 equals total produced waste in the economy ($W_0 = W_L + W_I$); (ii) energy demand E_0 is produced either through landfilling, incineration and/or other energy production facilities ($E_0 = E + E_L(W_L) + E_I(W_I)$); (iii) material demand G_0 is produced as materials recovered either from incineration ash or from existing plants for materials production ($G_0 = G + G_I(W_I)$). Finally, the choice variables (W_L, W_I, E *and* G) in the model have to meet the usual nonnegativity constraint.

Thus, we have the following minimand expression:

$$\begin{aligned}\min C \; &= L_P(W_L) + L_M(W_L) + I_P(W_I) + I_M(W_I) + E_P(E) + E_M(E) \\ &\quad + G_P(G) + G_M(G) + \lambda_1 \left(E_0 - E - E_L(W_L) - E_I(W_I)\right) \\ &\quad + \lambda_2 \left(G_0 - G - G_I(W_I)\right) + \lambda_3 \left(W_0 - W_L - W_I\right).\end{aligned} \tag{5.7.1}$$

Using the Kuhn-Tucker conditions:

$$\frac{\partial C}{\partial W_L} = L_P^{'}(W_L) + L_M^{'}(W_L) - \lambda_1 E_L^{'}(W_L) - \lambda_3 \geq 0 \tag{5.7.2}$$

$$\frac{\partial C}{\partial W_I} = I_P^{'}(W_I) + I_M^{'}(W_I) - \lambda_1 E_I^{'}(W_I) - \lambda_2 G_I^{'}(W_I) - \lambda_3 \geq 0 \tag{5.7.3}$$

$$\frac{\partial C}{\partial E} = E_P^{'}(E) + E_M^{'}(E) - \lambda_1 \geq 0 \tag{5.7.4}$$

$$\frac{\partial C}{\partial G} = G_P^{'}(G) + G_M^{'}(G) - \lambda_2 \geq 0 \tag{5.7.5}$$

$$\frac{\partial C}{\partial \lambda_1} = E_0 - E - E_L(W_L) - E_I(W_I) \geq 0 \tag{5.7.6}$$

$$\frac{\partial C}{\partial \lambda_2} = G_0 - G - G_I(W_I) \geq 0 \tag{5.7.7}$$

$$\frac{\partial C}{\partial \lambda_3} = W_0 - W_L - W_I \geq 0 \tag{5.7.8}$$

and the nonnegativity and complementary-slackness conditions gives (if $W_L > 0$ *and* $W_I >$

0) an optimal interior solution:

$$L'_P(W_L) + L'_M(W_L) - E'_{SC,L} = I'_P(W_I) + I'_M(W_I) - E'_{SC,I} - G'_{SC,I}, \tag{5.7.9}$$

with:

$$E'_{SC,L} = \left(E'_P(E) + E'_M(E)\right) E'_L(W_L) \tag{5.7.10}$$

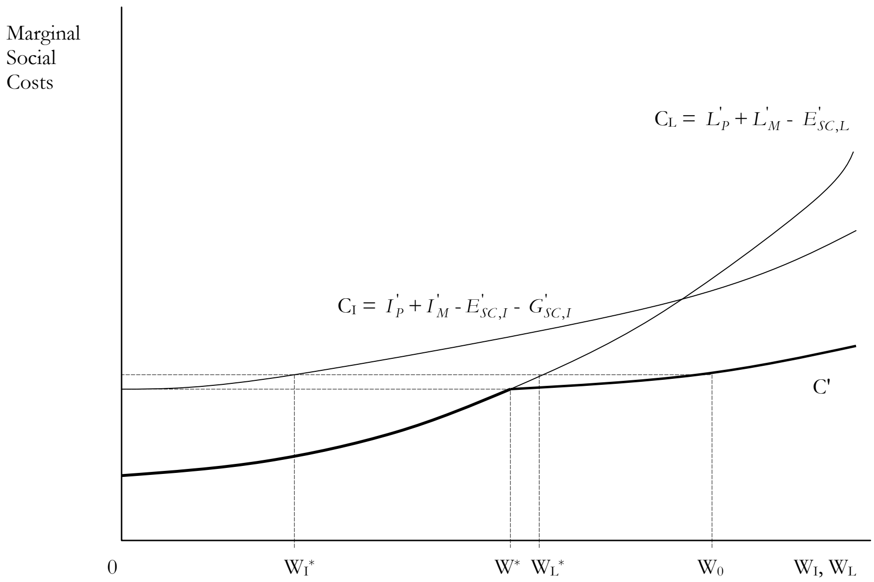
$$E'_{SC,I} = \left(E'_P(E) + E'_M(E)\right) E'_I(W_I) \tag{5.7.11}$$

$$G'_{SC,I} = \left(G'_P(G) + G'_M(G)\right) G'_I(W_I) \tag{5.7.12}$$

This expression has a clear economic meaning. Equation 5.7.9 states that it is only optimal for the government to dispose of its waste through both landfilling and incineration ($W_L > 0$ *and* $W_I > 0$) if the marginal social cost of both opportunities will be equal for a given amount of waste W_0 to be treated. These social costs, however, not only include the private and direct environmental costs of both disposal technologies ($L'_P + L'_M$ vis-à-vis $I'_P + I'_M$), but also the indirect social costs of each method through both the energy - and the materials - recovery system ($E'_{SC,L}$ vis-à-vis $E'_{SC,I} + G'_{SC,I}$). Indeed, due to their jointness characteristics, landfilling and incineration affect not only the processing of final waste, but also - indirectly - the private and environmental costs of energy and materials production. Obviously, if the marginal social costs of landfilling exceed the marginal social costs of incineration for a given range of waste to be processed, then the government should incinerate all waste from a social-cost perspective, and vice versa.

Obviously, how much waste is incinerated and/or landfilled in the optimal solution depends on the overall social-cost function (viz. on the marginal cost of landfilling and incineration together). Figure 5.2 illustrates the social-cost minimizing solution for an imaginary set of marginal-cost functions for incineration and landfilling. Given the marginal condition 5.7.9, one can also derive the total marginal social-cost curve C'. Up to W^* (with $W_0 \leq W^*$), our model generates a corner solution, with landfilling as the only optimal waste disposal method. Only if W_0 exceeds W^* will both options be optimal. The quantities incinerated (W_I^*) or landfilled (W_L^*) depend on the shape of the cost curves for both options. As, in the example of Figure 5.2, the fixed costs of incineration are considerable compared to the (marginal) variable costs, the size of the waste stream has to be large enough in order for it to be part of the optimal disposal mix.

Figure 5.2: Solution model landfilling versus incineration



The choice of how much waste should be treated by what option also depends on the social opportunity costs per unit recovered energy or material As is immediately clear from the marginal conditions in 5.7.9, energy and/or materials productivity play a crucial role in the total opportunity costs of either waste disposal option.

Chapter 6

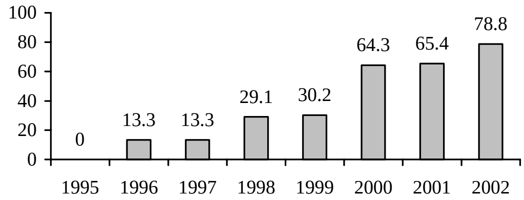
Regulation of the waste disposal market

6.1 Introduction

Historically the market for waste disposal is characterized by a lack of competition. Each local community had its own landfill site, most times operated by the local government. With the growth in other disposal opportunities like waste incineration and reuse, the production scale of waste disposal increased beyond the local level. This was intensified by environmental regulation making small local disposal sites unprofitable. As waste disposal was seen as an unwanted byproduct of consumption and production waste management plans were used in the Netherlands, till the end of the nineties, to plan the capacity of landfills and waste incineration plants (WIP's) on a provincial scale.[1] The goal was to have enough waste disposal capacity to treat the waste generated in each province within the borders of that province. Provincial borders were closed, resulting in monopolistic behavior of landfills and waste incineration plants. As the rising costs of waste disposal became a growing policy issue, the provincial scale was substituted for a national scale at the end of the nineties. Export of waste to disposal plants in other countries is still prohibited. In the Netherlands, as in most other EU-countries, national self-sufficiency for landfilling and waste incineration is a key issue of waste policy. The prohibition of imports and exports of waste destined for landfills and WIP's

[1] The Netherlands are divided into 12 provinces.

Figure 6.1: Landfill tax Netherlands in euro per ton



makes it possible to implement regulation instruments on a national scale. International harmonization of instruments is in this case not necessary. In the Netherlands a landfill ban and a high tax on landfilling waste are examples of instruments that are not applied in most other EU-countries. The reason for the implementation of these instruments is the promotion of WIP's. Since the seventies waste incineration is preferred above landfilling in the Netherlands. The intuition behind this preference was that incineration resulted in less space needed for waste disposal (incineration reduces the volume of the waste) and made more use of the energy content of the waste. Since the Kyoto treaty this last reason is more important as waste incineration can be used to reduce national carbon dioxide emissions.

As landfilling is cheaper than incineration, supplementary regulation instruments were needed to create enough investments in incineration capacity. In 1996 a landfill ban was announced. Landfilling was no longer allowed as long as enough waste incineration capacity was available. However, as the supply of burnable waste was much larger than the available capacity of WIP's a lot of exemptions were given. Given the far lower tariffs for landfilling, waste suppliers had an incentive to wait till the available capacity was filled and then asked for and got an exemption. Thus, the incentives of the landfill ban were not large enough to provoke substantial investments in waste incineration capacity. Therefore, a tax on landfilling was introduced. Starting with a low tax of 13 euro per ton of landfilled waste, the tax rose till the level of 79 euro in 2002 (see figure 6.1). As this level bridged the gap between the tariffs of landfilling and incineration excluding taxes, the waste incineration capacity more than doubled and is now at a level comparable with the supply of burnable waste.

However, it is questionable whether the independency of the Dutch market is continued in the near future. If, for instance, the EU decides to define incineration with energy recovery as a form of reuse a national ban to import and export waste is no longer allowed. EU-regulation implies a free international market for reuse of waste. Currently, waste suppliers try to open borders by legal procedures in Brussels. Furthermore, the differences in waste disposal tariffs between countries put pressure on the maintenance of import and export bans. In the Netherlands waste suppliers do have large incentives to search for maximal possibilities to export waste to foreign waste disposal firms as tariffs in the Netherlands are relatively high. As the definition of waste disposal is not always clear and definitions differ between countries, export of waste is in some cases possible under the flag of reuse.

If the prohibition of imports or exports is no longer maintainable or enforceable, the question arises what the impact is on national waste disposal policies. Is it still possible to achieve specific national waste goals or will the waste market develop like other markets, making only supranational goals possible to achieve? While some expert believe the answer is no (see for example Bakker and Doppenberg (2003)), no quantitative analysis is available. To get hold on the relation between national waste policies and international competition the first part of this chapter focus on two main questions. First, what are the possibilities to achieve the specific goals the Dutch government has set in the current waste market (self sufficiency for waste disposal, a preference for incineration above landfilling and diminishing cost)? Second, do these possibilities change when international borders are opened or closed for waste export? For both questions we analyze whether it is possible to implement national regulation instruments or whether harmonization between countries is necessary.

From an economic perspective it is interesting to analyze what happens when minimization of social costs is the leading principle for choosing governmental goals. Chapter 5 shows that the current preference of incineration above landfilling does not corresponds with the outcome of a social cost benefit analysis. In the second part of this chapter we show what happens to the waste market when regulation instruments are set according to this analysis. This makes clear what the costs are of the specific goals the government has chosen, both in terms of financial costs as in terms of possibilities to implement national regulation.

Section 6.2 presents the methodology used in this chapter to analyze the research questions. In section 6.3 we introduce the simulation model we use. Section 6.4 presents the simulation of the model for the current situation. Section 6.5 analyzes the use of regulation instruments in the current waste market. Section 6.6 handles the possibilities to use national instruments when international borders are open. Section refregecon presents the regulation possibilities and the costs of choosing non-economic goals when regulation instruments are set in accordance with economic principles. In section 6.8 the conclusions are presented.

6.2 Methodology

In the literature the effects of user fees on the generation of waste (see e.g. Jenkins, 1993 and Choe and Fraser, 1999) and the environmental effects of different treatment options (see e.g. Brisson, 1997) or a combination of both (see e.g. Beede and Bloom, 1995 and Kinnaman and Fullerton, 1999) get attention. With respect to the role of competition in waste markets nearly all papers focus on the waste collection market (see e.g. Bivand and Szymanski, 2000 and Szymanski, 1996). Exemptions are Tawil (1999) who analyze the effects of relaxing waste steering instruments in the US on the financial position of waste companies and Ley et al. (2002) who study the effects of restricting interstate trade in the US. However, on the effects of open borders between countries on the national regulation space not much is known. Futhermore, a comprehensive model for European countries is missing.

Economic theory gives an idea what happens when international markets integrate. For fair competition a level playing field is necessary. Thus, moving from national to international competition has influence on the regulation space of national policy makers. If for instance waste suppliers have to pay a high tax on landfill only in their own country, landfill sites have a competition disadvantage compared with foreign landfill sites due to differences in regulation. When national policy makers want to give their landfill sites an equal competition starting-point, harmonization of international landfill taxes is an option.

However, economic theory does not provide insight into the exact consequences of international integration. For example, the influence of differences in regulation are dependent on the transport costs. Differences in regulation between countries that are far away from each other will have less influence on the competition position than countries that are direct neighbors. Only when sufficient knowledge of both transport costs and differences in regulation is present, policy makers can decide on the optimal regulation package. This knowledge is relatively easy to gather as long as the number of countries is limited. However, when many countries are potentially influencing each other, the required information is more complex. In practice even countries that are far apart can influence each other as country A may influence the market of country B while country B influences country C. Thus, not only direct, but also indirect linkages between markets may be important. Furthermore, differences in one regulation instrument may be compensated by differences in other regulation instruments. Assume for instance that country A tries to promote incineration by a high landfill tax and country B stimulates WIP's by imposing a landfill ban. From a theoretical point of view both instruments can result in equal competition positions of landfill sites in both countries. However, in practice the influence of the landfill ban depends on the market circumstances.

From a policy perspective the determination of the national regulation space is a tricky business. Theoretical insights help to formulate optimal regulation packages, but specific empirical knowledge of the involved international markets is necessary. In this chapter we use an economic simulation model of the EU-countries to combine theoretical insights and empirical knowledge. The model is simulated with different regulation regimes to generate information on the national regulation space:

- We start with simulating the model for the current regulation regimes. The goal of this exercise is threefold. First, it gives more information on the way the model works. Second, it tests whether the model outcomes are credible. Third, the social costs of the current regulation regimes can be used as a benchmark for the other simulations.
- Next, the influence of different instruments is simulated. This makes it possible to search for optimal regulation packages that minimize social costs and to determine the national regulation space from a Dutch perspective.

6.3 A model for the European waste market

In this section we present the model used to analyze the relation between national regulation and international competition. As this model is rather complex, first a small model is presented. This small model has all essential behavioral characteristics to enlighten the essence of the large simulation model. This makes it easier to understand the way the simulation model operates. The second subsection describes the main features of the simulation model (see appendix A for a more formal description).

6.3.1 Small model

In the small model k regions exist. Each region produces waste that can be incinerated or landfilled. Per region a waste supplier exists that minimizes total costs by choosing the cheapest available incineration plant or landfill site. The waste supplier can choose between the incineration plants and landfill sites in both regions (export and import is alllowed). The waste supplier minimizes:

$$\min_{Q_s}(p^{ki}_{landf} + t^{ki})q^{ki}_{landf} + (p^{kj}_{incin} + t^{kj})q^{kj}_{WIP}. \tag{6.3.1}$$

with $\mathcal{I}$ the set of all landfill sites in the two regions (with index i), $\mathcal{J}$ the set of all WIP's in the two regions (with index j) and $\mathcal{K}$ the set of the two regions (with index k). Furthermore, the following variables are used:

p^{ki}_{landf}	:	the price of landfilling waste from region k in landfill i,
p^{kj}_{incin}	:	the price of incinerating waste from region k in WIP j,
q^{ki}_{landf}	:	the quantity of waste landfilled in region k and landfill i,
q^{kj}_{WIP}	:	the quantity of waste incinerated in region k in WIP j,
t^{ki}	:	the transport costs per ton of waste from region k to landfill i,
t^{kj}	:	the transport costs per ton of waste from region k to WIP j,
Q_s	:	$\{q^{ki}_{landf}, q^{kj}_{incin}\}$

As the total quantity of waste has to be treated somewhere (assuming no accumulation of waste outside the mentioned treatment options) waste has to be treated. Mathematically

$$\sum_{i\in\mathcal{I}} q^{ki}_{landf} + \sum_{j\in\mathcal{J}} q^{kj}_{WIP} = \overline{Q}^k, \tag{6.3.2}$$

with $\overline{Q}^k$ the supply of waste in region k.

Using the Kuhn-Tucker conditions equations 6.3.1 and 6.3.2 result in either $q^{ki}_{landf} = 0$ or $q^{kj}_{incin} = 0$ or $p^{ki}_{landf} + t^{ki} = p^{kj}_{incin} + t^{kj}$. This means that only both disposal options are used when the market prices including transport costs are equal.

It is assumed that WIP's aim at maximizing profits

$$\max_{q^{kj}_{WIP}}(p^{kj}_{incin} - c^{j}_{WIP})q^{kj}_{WIP}, \tag{6.3.3}$$

with c^j_{WIP} the variable costs of incinerating waste in WIP j.

WIP's have a mechanical constraint:

$$\sum_{k \in \mathcal{K}} q^{kj}_{WIP} \leq CAP^j_{WIP} \tag{6.3.4}$$

with CAP^{kj}_{WIP} the capacity in tons of WIP j.

It is assumed that landfills are price takers (they are indifferent with respect to the time of use of their capacity and operate under full competition) that have no capacity constraint.[2] Mathematically:

$$p^{ki}_{landf} = c^i_{landf}, \tag{6.3.5}$$

with c^i_{landf} the variable costs of landfilling waste in landfill i.

The essence of the model can now be illustrated by the following example. Assume that there are only two regions (A and B) that have both three WIP's and one landfill with variable costs and capacity as reported in Table 6.1. Assume further that transport costs are zero and that the waste supply in region A is 350 kton and in region B 650 kton. Equation 6.3.1 combined with 6.3.2 in fact orders the available disposal options according to their costs and contracts the least expensive options till the level that all waste is treated. Thus, first WIP 1 in region A is contracted. As the capacity of this plant is 150 kton and plants can not contract more than their available capacity (see equation 6.3.4), 850 kton of waste is still untreated. Next, WIP 1 in region B is contracted. This process is going on till the landfill in region A contracts the last 350 kton of waste. The landfill in region B and WIP's 3 in region A and B are not used because enough cheaper capacity is available. According to equation 6.3.3 the market price for incineration will now be equal to the costs of the most expensive disposal option that is used (for the landfill in region A). The outcome of the model is therefore that waste is treated for 80 euro per ton. In region A 250 kton is incinerated, in region B 400 kton, while the remaining 350 kton is landfilled in region A. This means that 250 kton of waste is exported from region B to region A.

The way in which the model calculates the effects of regulation can be illustrated by

[2] An essential difference between WIP's and landfill sites is that WIP capacity that is not used is foregone for ever because the capacity is time related.

Table 6.1: Solution small model

Disposal options	Costs euro per ton	Capacity kton	Used?	Remaining waste kton
WIP region A 1	20	150	Yes	850
WIP region B 1	25	200	Yes	650
WIP region A 2	40	100	Yes	550
WIP region B 2	45	200	Yes	350
Landfill region A	80	Inf.	Partly	0
Landfill region B	120	Inf.	No	0
WIP region B 3	150	200	No	0
WIP region A 3	160	500	No	0

assuming a landfill tax in region A of 90 euro. In this case the landfill in region A has now gross costs of 170 euro per ton. This means that this landfill is no longer contracted as it is the most expensive option. Instead the landfill in region B is contracted for 350 ton. As this is the most expensive option used, the market price will rise from 80 euro per ton till 120 euro per ton. As a result 100 kton waste is exported from region A to B. If both regions introduce a landfill tax of 90 euro landfilling is zero in both regions. WIP 3 in region A is the most expensive option used (for 150 kton), which makes the market price equal to 160 euro. Now 50 kton is exported from region B to A.

The solution of the small model is rather simple. However, this is not the case when all relevant variables are included in the model. In the next section the small model is extended to a full simulation model that makes simulation of the real effects possible.

6.3.2 Simulation model

In the simulation model a number of different regions are distinguished. Each region represents a part of a country. Within each region there is one waste collection firm which collects waste from the inhabitants and firms in the region. Inhabitants and firms differ with respect to the characteristics of the waste they produce. For our model the following characteristics are important:[3]

- Inhabitants produce household waste. This waste is called low caloric waste as the average caloric value is lower than the waste firms produce. A part of this waste stream can be separated into a very low and high caloric part.
- Some firms produce mid caloric waste. This waste is a combination of different

[3] Recall that this chapter only handles burnable non-chemical waste.

materials that can be separated from a technical point of view into two parts: a low and a high caloric part. The separation is possible because the different parts of the waste stream are homogeneous.

- Some firms produce dirty mid caloric waste. This waste is a combination of different materials that can not be separated from a technical point of view. The separation is impossible because the waste is a blend of very heterogeneous parts.

- Some firms produce high caloric waste. This waste is homogeneous and consists mainly of paper, plastics or wood.

Regions differ with respect to the quantity and composition of waste they produce due to differences in the number of inhabitants, the amount per inhabitant of generated waste and the size of different types of firms. The waste collection firm sells the collected waste to waste treatment firms. Depending on the characteristics of the waste the collection firm can choose between:[4]

- Landfills, which can buy all types of waste.

- General incineration plants, which can buy all types of waste within a certain technical limits.

- Dedicated incineration plants, which can buy only the type of waste they are designed for.

- Cement kilns and coal fired electricity plants, which can buy only high caloric waste due to technical constraints of their burning process.

The waste collection firm negotiates with all waste treatment firms in order to find the firm with the lowest treatment tariff (including all relevant costs). He negotiates not only with treatment firms in his own region, but also with firms in other regions. Of course, this is only possible when export to the other regions is allowed.[5] The tariff he finally has to pay includes not only the waste treatment price, but also the costs necessary to transport the waste to the location of the treatment firm and the separation costs he has to make when separation is necessary to make a treatment

[4] We assume that the waste has to be treated directly after collection. Keeping in stock is not possible in the model.

[5] The model assumes that both collection firms and waste treatment firms obey the law.

option possible. In the negotiation process the waste treatment firms try to maximize their profits (the difference between waste treatment price and given costs).

The model has two versions with respect to time. On short term the capacities of the waste treatment firms are fixed. Entry is not possible in the short term version. In the long term version entry is possible for general and dedicated incineration plants.[6] Entry will occur when the long term market price is equal or above the (given) costs of a new plant.[7] For landfills. cement kilns and coal fired electricity plants treatment capacity is still fixed in the long term version. The reasons for these assumptions are that the capacity of current landfills is sufficient for many years while it is not plausible to assume that the capacity of cement kilns or coal fired electricity plants depends on circumstances in the waste market.

A number of regulation instruments are included in the model. Simulation of the effects of instruments is possible by changing:

1. the tax or subsidy per treatment option (included as a positive or negative mark-up on costs);

2. the possibilities of entry per treatment option (if a moratorium on investments of a specific treatment option is present the entry of this treatment option is set at zero);

3. wether export and/or imports of specific waste streams (low, mid, dirty mid and/or high caloric) is allowed per treatment option.

These instruments can be specified per region. Thus, the effects of differences in regulation between regions can be simulated.

The previous section learned that the price of treating a specific waste stream will be equal to the cheapest option available for the last ton treated. However, without simulation of the model with actual data it is impossible to determine what the cheapest option available for the last ton treated is. This depends on supply and demand of the different waste streams, the height of transport costs, openness of borders, taxes,

[6] For three Dutch WIP's extension of their capacity is possible with relatively low cost as the current plants are already partly designed at a higher capacity.

[7] That costs are given means that the only decision investors can make is about the size and the number of the plants they build. Assumed is that all investors choose the same incineration technology with a fixed relation between capital, labor and other inputs. See paragraph 6.8 in appendix B for details on the assumed technology and the costs of new incineration plants.

subsidies, etc. The observation that the price of treating a specific waste stream will be equal to the cheapest option available for the last ton treated does also not make clear what the exact influence is of regulation instruments. Of course, when regulation influences the availability or costs of the cheapest option, market prices will be influenced. However, the magnitude of this effect is unclear a priori. For example, if landfilling is the option that determines the price and subsequently a tax on landfilling is introduced, the effect of this tax depends on the question whether landfilling remains the cheapest option. If this is not the case the change in market price will not be equal to the change in landfill tax as the market price is now determined by the costs of another option. Furthermore, not only regulation that influences the availability or costs of the cheapest option for the last ton treated may have influence on market prices, also other regulation instruments can have effects. If for instance regulation influences the availability of other options, the option used for the last ton treated may change and thus market prices. Concluding, it is not possible to determine a priori which factors determine market prices and the effects of regulation instruments. This depends on the actual situation of the waste market, which makes simulation of the model with actual data necessary.

The model is calibrated on the basis of an empirical investigation of the current and expected waste market characteristics in the EU. All data which are necessary to simulate the model are extensively described in appendix B. Here we summarize the main features:

- In the simulations all EU-countries are represented, split into certain regions per country (dependent on the size of the country). For the Netherlands we used four regions, three regions are used for Germany, France and Italy, two regions are used for Belgium, Denmark, Sweden, the UK, while one region is used for Luxembourg, Spain, Austria, Finland, Greece, Ireland and Portugal.

- Transport costs are calculated between the regions using a route planner and assuming transport per truck as the standard transportation mode. Transport costs are based on a door-to-door concept and include the costs of transshipment between collection truck and main transport and possibly shipping (United Kingdom). The cheaper option of inland shipping, available for transport between the Netherlands and some neighboring countries, is analyzed in a sensitivity analysis.[8] Transportation costs are relative high compared with other markets as value added per ton of waste is small. While the average costs of treatment in a new WIP is 77 euro per ton, transporting a ton of waste 800 kilometers costs about 40 euro.

- Costs of landfilling are estimated using national and international reports. Landfilling in the Netherlands is more expensive than in other countries due to higher landfill costs (primarily as a result of higher land prices and the unstable soil which demands more investments in prevention measures for leakage) and far higher landfill taxes. In the model it is assumed that landfill capacity is unlimited available. Per region one landfill site is modelled.

- Per region all existing waste incineration plants are included. The costs of Dutch waste incineration plants is well-known. This is not the case for WIP's in other countries. The costs of these plants is estimated on the basis of collected annual accounts and a enquiry sent to all WIP's. As existing WIP's have in general long-term contracts with waste suppliers in their own region, only the available capacity (total minus contracted) is included in the model. Based on Dutch figures available capacity equals about 30% of total capacity.

- In the long term the model allows entry for new WIP's. The costs of a new WIP are estimated using detailed figures about a recently build WIP in the Netherlands. This WIP's uses least-cost incineration technology and complies to the recently accepted emission limits set by European law. Costs per ton are 77 euro for a Dutch WIP and are comparable for other countries (costs for other countries are estimated using national labor and capital costs).

- Per region one representing cement kiln and coal fired electricity plant is included. The capacity of these plants is estimated using international reports on the possibilities to substitute fossil fuels for waste. Net-costs are far lower for these plants compared with WIP's. On average treatment of waste in a cement kiln or coal fired electricity plant costs only about 20 euro, while the average costs for a new WIP are 77 euro. In general these plants can only burn high caloric waste. While some firms produce this waste as a by-product, separation plants

[8] Maritime shipping is for longer distance a much cheaper option, but biological problems take place when the transport time of waste that is not pretreated (dehydration) is longer than 48 hours. The transshipment time for maritime shipping is already longer than this time limit.

can generate high caloric waste by separating mid or low caloric waste. Per region one representing separation plant is included in the model. Separation costs are on average nearly 20 euro per ton.

- For each region the waste supply is estimated using national waste statistics. In the EU waste supply is about 404 Mton in the short term (2001), while the capacity of WIP's, cement kiln and coal fired electricity plants is only about 112 Mton. For all countries this undercapacity is present, though the relation between incineration capacity and waste supply differs significant. While in Denmark (98%), Germany (88%) and the Netherlands (76%) a major part of total waste can be burned, this is not the case for Italy (7%), the UK (14%) and France (23%).[9]

- Regulation instruments that are used by EU-members are landfill taxes, landfill bans and incineration taxes (see Table 6.2).[10] A relative high landfill tax is used in the Netherlands (75 euro per ton), Flanders (45 euro) and Denmark (60 euro). These last two countries have also an incineration tax (Flanders 25 euro and Denmark 28 euro). All three countries have a landfill ban, while Germany and France are discussing the introduction of such a ban. For each region the landfill and incineration taxes are included in the model. The landfill ban is ignored as Dutch experience reveals that in a situation of undercapacity of incineration (which is the case in all EU-countries) this ban is not maintainable.

Except for the regulation instruments, uncertainty is present on the estimated values of the exogenous variables necessary to calibrate the model. To get hold on the influence of this uncertainty we simulate the model for different values of the exogenous variables. For each type of variables a basic value represents the best estimate of the true value. We call the model calibrated with all basis values the basic scenario. Furthermore, for each type of variables an estimated minimum and maximum value represent the borders of the uncertainty (see appendix B for a description of the exact values). A number of first-order sensitivity scenarios are defined using the minimum or maximum value of one type of variables and the basic value for the other types for calibration. For example, a first-order sensitivity scenario differs only from the basic scenario with respect to

[9] For the long term waste supply is estimated using the Dutch expected relation between waste per inhabitant for household waste and waste per unit value added for commercial waste (see page 161). Sensitivity analysis shows that the model outcomes are not sensitive for the assumptions made to calculate the waste supply of regions.

[10] Emission limits for WIP's and landfill sites are harmonized between countries by a recently accepted European directive.

Table 6.2: Waste regulation in the EU (tariffs in euro per ton)

	Landfill tax	Incineration tax	Landfill ban
Belgium (Flanders)	45[a]	25	Yes
Belgium (Wallonia)	Yes	No	No
Denmark	60	28	Yes
Germany	No	No	Yes (2005?)
Finland	15	No	Yes (2005)
France	6	No	Yes (2005?)
Greece	No	No	No
Ireland	No	No	No
Italy	5-26[b]	No	No
Luxembourg	>23	n.a.	n.a.
Netherlands	75	0	Yes
Austria	15	No	Yes (2004)
Portugal	No	No	No
Spain	No	No	No
UK	20[c]	No	No
Sweden	30	No[d]	Yes (2002)

Sources: 54 different sources

[a] Tariff for municipal waste.

[b] Tariffs differ per region. Commercial waste: min. 5 and max. 10 euro per ton. Municipal waste: min. 10 and max. 26 euro per ton.

[c] Tariff in 2000. Per year an one pound increase is agreed.

[d] A discussion is going on about an incineration tax.

transport costs (based on costs using a lorry in the basic scenario and using inland shipping and sea shipping in the sensitivity scenario) or with respect to the costs of existing WIP's. We start simulating the model using the basic scenario. This gives the mean expected outcome. Second, we simulate the model using the sensitivity scenarios. This gives the maximum expected deviation from the basic scenario and thus indicates the level of uncertainty.

In total we have 78 different first-order sensitivity scenarios. These scenarios indicate what happens to the results of the basic scenario when only one type of exogenous variables differs from the basic value. However, it is possible that in practice different types of variables differ from their basic value at the same time. The expected effects of this possibility can be simulated using combinations of the first-order sensitivity scenarios. For example, a second-order sensitivity scenario can be simulated when both transport costs and costs of existing WIP's differ from their basic value. Unfortunately, the total number of higher order sensitivity scenarios is very large. The number of second-order sensitivity scenarios is equal to 2966, while the number or third-order scenarios is already 76076. Given an average simulation time of half a minute for all different first to sixth-order scenarios would take 265 years of computer time.[11] Fortunately, most models have the characteristic that third and higher order effects are nearly zero (Montgomery, 2000). Therefore, we simulate first- and second-order scenarios and tested whether second-order effects differ significant from the first-order effects. It showed that this was not the case on a 99% confidence interval. Thus, we can rely on the first-order effects.

6.4 The current waste market

In this section we present the simulations results for the model using the current (2002) regulation regime. The goal of this exercise is threefold. First, it gives more information on the way the model works. Second, it tests whether the model outcomes are credible. Third, the social costs of the current regulation regimes can be used as a benchmark for the other simulations.

The current situation with respect to the regulation regime is characterized by:

- a landfill tax of 75 euro per ton;
- allowing export of high caloric waste because it is not disposal but reuse (as defined by EU-regulation);
- allowing export of mid caloric waste to separation plants because it is not disposal but reuse as defined by EU-regulation;
- a ban for all other forms of exports;

Table 6.3 presents the simulation results. In the first three rows figures are given for the main objectives this chapter focus on. On **short term** the level of **selfsufficiency** (the percentage of total waste that is treated in the Netherlands) is 79%. In other countries 1.3 Mton and 0.8 Mton of Dutch waste is respectively reused and landfilled. The reason for this export is that for Dutch mid and high caloric waste reuse in cement kilns and coal fired electricity plants is the cheapest option. The price of mid and high

[11] The average simulation time is based on a smaller version of the model. As simulations with all EU-countries showed no major influence of member states farther away than approximately 800 kilometers (due to the high transportation costs), the model is limited for the sensitivity analysis to the Netherlands (four regions) and the neighboring countries Belgium (Wallonia and Flanders), France (North, Mid and South), Germany (North, West and South), Denmark (South), Italy (North) and United Kingdom (North and South).

Table 6.3: Simulation current situation

	Short term		Long term	
	Level	Prob.	Level	Prob.
Selfsufficiency (% of total waste)	79		82	
Landfilling (% of total waste)	17		7	
Total costs (million euro per year)				
- excluding quantity effect	738	0.97	720	0.92
- including quantity effect	738	0.97	785	0.92
Tax income (million euro per year)	73	0.67	0	1.00
Total Dutch waste (Mton)	9.9		10.7	
- landfilled in Netherlands	1.0	0.67	0.0	1.00
- incinerated in Netherlands	4.8	1.00	6.6	1.00
- reused in Netherlands	2.1	0.95	2.1	0.95
- landfilled in other countries	0.8	0.95	0.8	0.95
- incinerated in other countries	0.0	0.97	0.0	0.97
- reused in other countries	1.3	0.92	1.2	0.93
Imported from other countries	0.0	1.00	0.0	1.00
New incineration capacity (Mton)				
- low caloric			0.8	0.78
- mid caloric			0.8	0.85
- high caloric			0.0	1.00
Price incineration (euro per ton)				
- Low caloric waste	113	0.95	77	0.95
- Dirty mid caloric waste	116	0.95	77	0.95
- Mid caloric waste	71	0.93	53	0.95
- High caloric waste	29	0.90	29	0.92
Price landfill (euro per ton)	116	1.00	n.a.	n.a.

caloric waste is determined in an international market (export is allowed). In this market the lack of landfill taxes in Germany, Wallonia and France causes a price level that is low compared with other options as waste suppliers of mid and high caloric waste in these countries choose landfilling as the cheapest option. Dutch waste suppliers export part of their waste stream (2.1 Mton) as the national reuse capacity (2.1 Mton) is lower than the supply of Dutch mid and high caloric waste (4.2 Mton). As the main part of the exported waste stream (1.6 Mton) consists of mid caloric waste separation is necessary. Separation plants split this waste in a high caloric part (which is reused) and a low caloric part (which is landfilled).

On short term 17% of total waste is **landfilled**. Besides the landfilling in other countries, 1.0 Mton is landfilled in the Netherlands. Although the landfill tax gives waste suppliers an incentive to choose a cheaper option, this option is not available. For low caloric waste and dirty mid caloric waste, export is not allowed, resulting in a national market. In this market there is shortage of incineration capacity as the waste supply is 5.8 Mton and the available capacity is only 4.8 Mton.[12]

Total costs are 810 million euro on short term (including a tax income of 73 million euro).[13] As the price for treating low caloric waste (113 euro in a WIP and 116 euro in a landfill) is far higher than for mid and high caloric waste (71 and 29 euro) and citizens are the only suppliers of low caloric waste, the costs per ton are for citizens far higher (96.07) than for firms (49.49).

On **long term** the level of **selfsufficiency** rises a little bit as newly build WIP's enlarge the national incineration capacity. The high Dutch landfill tax stimulates entry as Dutch waste suppliers still search for a cheaper option than landfilling. As the costs of a new WIP are significant below the level of landfilling costs, new incineration capacity is build. This results in a decrease of total **landfilling** till a level of 7%. Only the residue resulting from separation in other countries is now landfilled. The entry of new WIP's also results in a decrease of tariffs for (especially) low caloric waste till the level of the costs of a new entrant (77 euro). While citizens still pay more per ton of waste compared with firms as they can not profit from lower prices on the international market, the gap is significant smaller. **Total costs** (including taxes) decrease with 91 million euro due to the lower tariffs, a decrease of 11.2% per ton of waste.[14]

Summarizing, the objectives of selfsufficiency and landfilling are not reached on short and long term. On long term 7% of total waste is landfilled in other countries, while 18% is not treated in the Netherlands. The next section will analyze whether other regulation packages are available that perform better on these objectives and whether cost decreases are attainable. Before this analysis is presented, we first discuss the reliability of the model simulations.

One method to get hold on the reliability of the model outcomes is the confrontation with the current waste market. Are the model outcomes comparable with what happens

[12] In principle part of the low caloric waste can be reused after separation. The model simulations show however that the relative low percentage of high caloric waste that is generated by separation plants of low caloric waste, make this separation never profitable.

[13] All costs presented in this chapter are inclusive the costs of low caloric waste that is already contracted by WIP's in long term contracts. In total about 3,5 Mton low caloric waste is contracted, which represents a total estimated costs of about 370 million euro.

[14] To make short term and long term costs comparable, total cost for the long term are also presented excluding the quantity effect. In this case total cost for the long term is equal to the long term prices times the short term quantities.

in reality? Although information about the prices and treated quantities in the current market is rather scattered, at least there is circumstantial evidence that the simulation results are not contrary to the real market. For instance, the incinerated quantity of waste in the Netherlands was 4.9 Mton in 2002 (VVAV, 2003) while our simulation results suggest a comparable 4.8 Mton. The level of landfilling was higher than simulated by the model in the years before 2003, but as the export of waste rises since 2002 the expected outcome for 2003 (0.7 Mton according to Afval! (2003)) remarkably matches with the 0.8 Mton the model predicts. The rising export quantities (0.5 Mton in 2001, 1.3 Mton in 2002 and at least 1.6 Mton in 2003 according to VVAV (2003)) indicates that the model outcome of 2.1 Mton is not far beside the truth. The biggest gap between reality and model outcomes is related to the level of reuse in the Netherlands. While the model predicts a level of 2.1 Mton of waste used in cement kiln and coal fired electricity plants, the current level is nearly zero. However, according to the AOO investments plans exist to make reuse of waste possible in the Netherlands of at least 1.8 Mton (Van den Brand, 2002). The problem here (which the model neglects) is that currently the emissions limits for incinerating waste in reuse-plants is such that incineration is not attractable. However, at short term these limits change, making reuse possible (Van den Brand, 2001). Not only waste quantities, also predicted prices are not in conflict with market outcomes. The current prices of two Dutch WIP's for free contractable waste are with 128 and 130 euro only slightly above the predicted level of 116 euro.[15] Furthermore, these prices are the tariffs mentioned at the website of the two WIP's. Negotiating lower tariffs should be possible. The RIVM mentions for incineration an average tariff of 106 euro per ton for 2002 and for landfilling 128 euro (RIVM, 2003). For mid caloric waste an export tariff of 70 euro is mentioned (Noordhoek, 2003), only 1 euro less than the model prediction, while for high caloric waste a spot market price less than 45 euro is possible (AOO, 2003). Summarizing, it seems that the model predictions are not in conflict with the outcomes of the real market. However, some frictions exist which makes it possible that the market needs some more time to reach the equilibrium level.

A second method to get hold on the reliability of the model is to analyze whether other assumptions influence model outcomes. In Table 6.3 the uncertainty of the model outcomes is presented in de 'Prob.'-columns. These figures are calculated as the share of second-order sensitivity scenarios that result in an outcome that differs more than

[15] See www.arnbv.nl and www.huisvuilcentrale.nl.

plus or min 10% from the basic scenario. For example, the quantity of waste reused in the Netherlands lies with a probability of 95% within a range of 1,9 to 2,3 Mton.[16] It shows that nearly all variables have a low uncertainty level. Furthermore, only 30 of the 78 sensitivity scenarios do have an effect on the outcomes. Thus, the model is rather reliable. Appendix C presents the effects of all sensitivity scenarios on the basic outcomes. To illustrate the working of the model we here only present the results of the sensitivity scenarios that change total costs in the long term model. It shows that only 7 sensitivity scenarios matter:

- When the costs of a new WIP are higher (17 euro in the Netherlands), total cost rise with 53 million euro. This happens because in the long term model the entrants determine prices for low and dirty mid caloric waste. Higher costs for new entrants does result in higher prices. For example, the price for incinerating low and dirty mid caloric waste rises from 77 euro to 94 euro. This increase in price is exactly equal to the risen costs as waste suppliers do not have a cheaper alternative. Landfilling is only allowed in the Netherlands itself and costs 116 euro, while incineration in a cement kiln or coal fired electricity plant is not possible for this type of waste. The opposite effects occur when the costs of a new WIP are lower. Total costs fall with 44 million euro in this case. Again the change in price is equal to the change in costs for a new entrant.

- When the costs of incinerating waste in a cement kiln or coal fired electricity plant are higher (20 euro) total costs rise with 66 million euro. This is due to the higher price (which increases from 29 to 49 euro) of incinerating high caloric waste. Although some waste now chooses an other option (incineration in a foreign WIP), the total price increase is equal to the cost increase as no cheaper options are available at the margin. Total costs fall with 32 million euro when the costs of cement kilns or coal fired electricity plants are 10 euro lower.

- When the costs of separation plants is 20 euro higher, total costs rise with 31 million euro. This is entirely due to the higher prices separation plants ask. In the basic scenario separation of mid caloric waste is maximal and separation of low caloric waste is minimal. This is due to the fact that the separation of mid caloric waste is more profitable as after separation 50% of the waste, which now

[16] Note that this measure is only sensible when the probability is equal that all different sensitivity scenarios reflect the real situation.

is high caloric, can be incinerated in the much cheaper cement kilns or coal fired electricity plants, while after separation of low caloric waste only 15% of the waste is high caloric. Thus, for low caloric waste the costs of separation is higher per ton of high caloric waste than the price difference between incineration of low and high caloric waste. The higher costs of separation plants in the sensitivity scenario does not change the profitability of separation at the margin. However, when separation costs are 20 euro lower, it is profitable to separate part of the low caloric waste. Therefore, total costs falls with 17 million euro.

- Finally, total costs rise with 25 million euro when the capacity of cement kilns and coal fired electricity plants is 50% lower. This is due to the fact that now suppliers of high caloric waste have to make higher transport costs as only enough capacity is available in other countries.

As this analysis showed that the outcomes are rather reliable, we do not present the sensitivity analysis statistics for other simulations.

6.5 Regulation in the current waste market

The previous section showed that the current regulation regime does not result in the achievement of policy objectives. Waste is landfilled, while the selfsufficiency is not 100%. Furthermore, it is not clear whether the current regulation package minimizes costs. This section analyses whether changes in regulation can improve the waste market outcomes.[17]

The first instrument that can be used to improve the **selfsufficiency** is rather simple. When national borders are closed, selfsufficiency is 100% by definition. Table 6.4 shows that this is indeed the case, but that as a consequence **landfilling** rises on short term from 17% to 32%. This is caused by the lack of incineration and reuse capacity in the Netherlands. The only option available for waste suppliers that formerly exported waste is landfilling. As this option is very expensive compared with reuse **total cost** increases. A second reason for cost increase is that the price of treating high caloric waste in the Netherlands rises. As this price is now determined in the national market and the available capacity is smaller than the supply of high caloric waste, it rises from 29 euro

[17] For all regulation scenario's only results for the three main objectives are given. For detailed results see appendix D

Table 6.4: Regulation instruments in market with closed borders

	Selfsufficiency % Total waste		Landfilling % Total waste		Total cost[a] Mln. euro	
	ST	LT	ST	LT	ST	LT
Current (landfill tax = 75 euro)	79	82	17	7	738[b]	720[c]
Closed borders						
- landfill tax = 75 euro	100	100	32	0	903[d]	888[c]
- landfill tax lower (36 euro)	100	100	32	0	774[e]	888[c]

a. Excluding tax income and including subsidies. For the long term excluding quantity effect.
Tax income is respectively 73 (b), zero (c), 237 (d) and 114 (e) million euro

till a level of 116 euro (the costs of the cheapest alternative: landfilling). In total cost increases with 265 million euro. Table 6.4 shows that part of this cost increase is due to the high landfill tax. When the tax is set at the minimal level necessary to provoke entry on long term (36 euro, which is the difference between the costs of a new WIP and landfilling), total costs diminishes with 129 million euro as a result of lower market prices. As there are no effects on the other policy goals this shows that the current tax level is higher than necessary. However, total costs are still 36 million euro higher that with open borders for reuse. Therefore, with closed borders a selfsufficiency of 100% is only reachable at considerable costs and with an increase in landfilling. In the long term **selfsufficieny** remains 100%, but now **landfilling** is zero. This is caused by the entry of new WIP's. **Total costs** decrease till a level of 889 million euro, which is 169 million higher than with open borders for reuse. Concluding, closed border can be used as an instrument for reaching a selfsufficiency level of 100% and zero landfilling on long term, but total costs rise considerably while landfilling also rises in the short term.

When borders for reuse are left open, as in the current regulation regime, a second instrument to improve the outcomes of the waste market could be a lower level of the landfill tax. The current high tax (compared with neighboring countries) results in an incentive to export waste. Table 6.5 shows that this incentive is less for the short term when the tax is decreased to a level of 36 euro per ton (which is the difference between the costs of a new WIP and landfilling). **Selfsufficiency** rises with 1%-point. However, as the capacity of WIP's, cement kilns and electricity plants is already used, **landfilling** rises also. The lower landfill tax results in lower treatment prices. As a consequence **total costs** diminish till 678 million euro. On long term the decrease in landfill tax has no effect at all. This is because not the landfill tariff sets the price on long term, but

Table 6.5: Regulation instruments in current market

	Selfsufficiency % Total waste		Landfilling % Total waste		Total cost[a] Mln. euro	
	ST	LT	ST	LT	ST	LT
Current (landfill tax = 75 euro)	79	82	17	7	738[b]	720[e]
National instruments						
- landfill tax lower (36 euro)	80	82	20	7	679[n]	720[e]
- landfill tax zero	96	96	29	34[f]	630[e]	628[e]
- and subsidy WIP's (18 euro)	96	96	28	30[f]	711[l]	717[m]
- and subsidy WIP's (36 euro)	96	96	27	0	796[h]	922[i]
- and subsidy WIP's (52 euro)	96	100	27	0	871[k]	1006[j]
International instruments						
- landfill tax (EU=WIP-landfill)	96	89	27	0	751[o]	774[e]

Excluding tax income and including subsidies and, for the long term, excluding quantity effect (a). Prices are beneath costs new WIP: landfilling will be higher (76% of total waste) when existing WIP's are closed (f). Tax income is respectively 73 (b), 72 (c), 88 (d), zero (e), 116 (g), 42 (n) and 98 (o) million euro. Subsidy budget is respectively 168 (h), 294 (i), 447 (j), 242 (k), 82 (l) and 89 (m) million euro

the costs of new entrants. However, a further decrease of the landfill tax does have effects on long term as the landfill tariff then moves beneath these costs. Table 6.5 shows that a zero landfill tax decreases **total cost** significantly due to the substitution in the direction of landfilling. Now **landfilling** is increased to a level of 34% at least. Consequently, the export of Dutch waste is nearly zero leading to a **selfsufficiency** of 96%. Only 0.5 Mton of waste is exported for reuse in other countries. The abolishment of the landfill tax thus leads to far better results for selfsufficiency and total costs, but not for landfilling.

An instrument to promote incineration in the absence of a landfill tax is a subsidy. Table 6.5 present results for a subsidy on WIP costs per ton of 18, 36 and 52 euro per ton. Logically, a subsidy on incineration does not help to decrease **landfilling** in the short term as initially all capacity of WIP's is used. In the long term the subsidy leads to new entrants. If the subsidy is high enough (36 euro), landfilling is zero. For this subsidy level **selfsufficiency** is also higher, but still 0.5 Mton is exported. The subsidy necessary to prevent this export is equal to 52 euro per ton.[18] **Total costs** rise when a subsidy of 36 euro is introduced to a level of 959 million euro on long term.

[18] A higher subsidy leads to import of waste. When this is not desirable, a fixed subsidy budget could prevent this import. Although a fixed budget thus not prohibit imports, in the long term arbitrage will take place so that no imports will take on balance.

Interestingly, this level is only 4 million higher than that of a regulation package with closed borders and a landfill tax of 36 euro. Compared with the costs of the current regulation package total cost rise with 174 million euro. When a selfsufficiency level of 100% is the ultimate goal total cost rise with 242 million euro, compared with the current regulation package.

Up to here we discussed instruments that can be implemented by the Dutch government. As waste destined for reuse can be exported, international instruments might perform better. Table 6.5 shows that this is the case for **total costs**. When all European countries introduce a landfill tax equal to the difference between a new WIP and the costs of landfilling (in the Netherlands this is 36 euro), total cost is 840 million euro.[19] This is 119 million euro less than for the national regulation with a 36 euro subsidy on WIP's. With this subsidy **landfilling** is zero in the Netherlands. On the other hand **selfsufficiency** is 'only' 89% as it is still profitable to reuse part of the Dutch waste stream in other countries.

6.6 Regulation with open international borders

When regulation is changed in order to permit export of all waste, the percentage of **selfsufficiency** drops to 63%, while **landfilling** rises to 37% in the short term (see Table 6.6). In the long term the results are comparable. As waste for landfilling is now allowed to go to other countries and landfilling in some neighboring countries is very cheap, this option is used for 3.7 Mton in the short term and 4.5 Mton in the long term. Only some Dutch WIP's attract waste (above the already contracted 3,5 Mton this is 0.6 Mton) as their variable costs are beneath the landfill price in other countries (including transport costs). Furthermore, Dutch reuse plants operate on full capacity. The rest of the waste is however exported to foreign landfill sites. **Total costs** are only 569 million euro on short term and remain on this level on long term (when the quantity effect is excluded). As no waste in landfilled in the Netherlands tax income is also zero. The large decrease in total costs when borders are open is also found by Ley et al. (2002). For the US they find significant increase in social welfare when interstate trade is allowed. Tawil finds no significant effect of restricting interstate trade on the financial position of landfill, collection and waste incineration firms. However, she suggests that

[19] This scenario corresponds to the situation that all EU-countries prefer incineration or reuse above landfilling. The Dutch experience shows that a landfill ban is not effective to reach this goal.

this result is driven by the fierce competition that is present in the waste markets she looks at.

Table 6.6: Dutch waste market with open borders

	Short term		Long term	
	Level	Change[a]	Level	Change[a]
Selfsufficiency (% of total waste)	63	-16	58	-24
Landfilling (% of total waste)	37	+20	42	35
Total costs (million euro per year)				
- excluding quantity effect	569	-168	569	-151
- including quantity effect	569	-168	596	-189
Tax income (million euro per year)	0	-73	0	0
Total Dutch waste (Mton)	9.9	0	10.7	0
- landfilled in Netherlands	0	-1.0	0	0
- incinerated in Netherlands	4.1	-0.7	4.1	-2.5
- reused in Netherlands	2.1	0	2.1	0
- landfilled in other countries	3.7	+2.9	4.5	+3.7
- incinerated in other countries	0	0	0	0
- reused in other countries	0	-1.3	0	-1.2
Imported from other countries	0	0	0	0
New incineration capacity (Mton)				
- low caloric			0	-0.8
- mid caloric			0	-0.8
- high caloric			0	0
Price incineration (euro per ton)				
- Low caloric waste	n.a.	n.a.	n.a.	n.a.
- Dirty mid caloric waste	31	-85	31	-46
- Mid caloric waste	31	-40	31	-46
- High caloric waste	24	-5	24	-29
Price landfill (euro per ton)	17	-99	17	n.a.

[a] Difference with current market outcomes (compare Table 6.3).

Table 6.7 present the results for different regulation packages. Compared with the situation that export is only allowed for reuse, most instruments are not effective anymore. Only a combination of a zero landfill tax and a high subsidy for WIP's (52 euro) or a EU landfill tax perform comparably.

The reason for the decreased effectiveness of regulation is that landfilling in other countries is now accessible for most waste streams. As these landfill sites have very low costs, prices are not set by Dutch treatment firms. Therefore, changing Dutch taxes or subsidies do have fewer effects in most cases.

Table 6.7: Regulation instruments in market with open borders

	Selfsufficiency % Total waste		Landfilling % Total waste		Total cost[a] Mln. euro	
	ST	LT	ST	LT	ST	LT
Current (landfill tax = 75 euro)	79	82	17	7	738[c]	720[f]
Open borders						
- landfill tax is 75 euro	63	58	37	42[c]	569[b]	569[b]
- landfill tax lower (36 euro)	63	58	37	42[c]	569[b]	569[b]
- landfill tax zero	63	58	37	42[c]	569[b]	569[b]
- and subsidy WIP's (18 euro)	66	62	33	37[c]	653[d]	655[e]
- and subsidy WIP's (36 euro)	68	67	31	32[c]	739[f]	754[g]
- and subsidy WIP's (52 euro)	68	96	30	4	814[h]	979[i]
International instruments						
- landfill tax (EU=WIP-landfill)	96	89	27	0	751[j]	775[b]

Excluding tax income and including subsidies and, for the long term, excluding quantity effect (a). Prices are beneath costs new WIP: landfilling will be higher (76% of total waste) when existing WIP's are closed (c). Tax income is respectively zero (b) and 98 (j) million euro. Subsidy budget is respectively 79 (d), 82 (e), 167 (f), 180 (g), 242 (h) and 423 (i) million euro

Even with a zero landfill tax foreign landfill sites can contract waste cheaper than Dutch landfill sites. Only when costs of incineration are below the tariffs (including transport costs) of foreign landfill sites (this is the case with a subsidy of 52 euro) or when the tariffs of foreign landfill sites are increased by an EU-landfill taxes, all waste is treated in the Netherlands.

6.7 Regulation according to external costs

In this section we simulate the model with external costs internalized by means of taxes. From an economic point of view the outcome of the market is optimal when all external costs are internalized. Waste suppliers then decide on the true costs of treatment and the option they choose including the location of treatment is optimal per definition. Compared with the analysis in the preceding sections this means that the goals chosen by the Dutch government are not decisive whether the outcome of the market is acceptable. The interesting point of this analysis is that it gives an indication what the costs are of the chosen goals of the Dutch government. Leads regulation from an economic point of view to higher or lower costs than regulation based on the specific goals the Dutch government has chosen?

The standard solution to internalize these costs is to tax treatment options according to the external costs. Chapter 5 analyzed the external costs for landfilling and for incineration. As the model uses more detail on waste streams (chapter 5 analyses external costs only for low caloric waste) and includes also other treatment options, additional information is necessary to calculate all external costs necessary to set the required taxes. The following assumptions are made:

- The energy production of WIP's, cement kilns and coal fired electricity plants are linearly dependent on the heating value of the waste streams. Thus, incinerating high caloric waste (18 GJ per ton) generated 1.8 times as much energy as incinerating low caloric waste (10 GJ per ton);
- The energy production of landfilling does not depend on the waste landfilled.
- The emissions of cement kilns and coal fired electricity plants are equal to WIP's except for CH_4, SO_2 and NOx as EU emission limits differ only for these three emissions. This result in higher external cost due to higher emissions of CH_4 (0.02 euro), SO_2 (4.42 euro) and NOx (3.10) euro.
- There are no savings due to recycling of iron and aluminum for cement kilns and coal fired electricity plants (external costs rise with 5.76 euro).
- The environmental costs of chemical waste for cement kilns are zero as there is no final waste that needs to be landfilled (cost decrease with 29 euro).

Table 6.8: Taxes according to environmental costs in euro per ton

Waste	Landfill	Incineration	Electricity plants	Cement kilns
Very low caloric	22	21	n.a.	n.a.
Low caloric	22	18	n.a.	n.a.
Mid caloric	22	9	n.a.	n.a.
High caloric	22	1	14	-15

Calculating external costs for all treatment options and waste streams leads to taxes as presented in Table 6.8. The tax on landfill is equal for all waste streams (22 euro per ton of waste). For incineration of very low caloric waste the tax is nearly equal compared with landfilling. When the heating value is higher, electricity production rises and environmental costs decrease. This results in a decrease of environmental taxes for incineration. For high caloric waste this tax is nearly zero. Coal fired electricity plants and cement kilns can only burn high caloric waste. For coal fired electricity plants the tax is 14 euro per ton, while for cement kilns a negative tax (subsidy) is required of 15 euro per ton (primarily due to the savings on external costs of chemical waste). Compared with the current regulation regime, the tax on landfilling is rather low, while a tax is introduced for incineration and coal fired electricity plants and a subsidy is given to cement kilns.[20]

Table 6.9: Open borders and taxes according to external costs

	Short term		Long term	
	Level	Change[a]	Level	Change[a]
Selfsufficiency (% of total waste)	63	-16	61	-21
Landfilling (% of total waste)	36	+19	39	+32
Total costs (million euro per year)				
- excluding quantity effect	556	-182	591	-129
- including quantity effect	556	-182	636	-149
Tax income (million euro per year)	106	+33	106	+106
Total Dutch waste (Mton)	9.9	0	10.7	0
- landfilled in Netherlands	0.0	-1.0	0.0	0
- incinerated in Netherlands	4.1	-0.7	4.4	-2.2
- reused in Netherlands	2.1	0	2.1	0
- landfilled in other countries	3.6	+2.8	4.1	+3.3
- incinerated in other countries	0.0	0	0.0	0
- reused in other countries	0.1	-1.2	0.1	-1.1
Imported from other countries	0.0	0	0.0	0
New incineration capacity (Mton)				
- low caloric	n.a.		0	-0.8
- mid caloric	n.a.		0	-0.8
- high caloric	n.a.		0	0
Price incineration (euro per ton)				
- Low caloric waste	n.a.	n.a.	48	n.a.
- Dirty mid caloric waste	50	-66	54	-23
- Mid caloric waste	50	-21	45	-8
- High caloric waste	38	+9	42	+13
Price landfill (euro per ton)	37	-79	39	n.a.

[a] Difference with current market outcomes (compare Table 6.3).

Table 6.9 presents the results for the model simulations including taxes in accordance with Table 6.8. The **selfsufficiency** percentage is now only 63% on short term and

[20] Note that these outcomes indicate that for high caloric waste the best option from a social cost perspective is not landfilling nor incineration but using it as a fuel in a cement kiln.

61% on long term. This is the result of a high percentage **landfilled** Dutch waste in other countries. Thus from an economic perspective it is optimal to export nearly 40% of Dutch waste to foreign landfills as the social costs of these landfills are lower than all other options. Still existing national WIP's treat 40% of Dutch waste and national reuse is good for the other 20%. Both options are cheaper than landfilling in other countries. However, prices lie beneath the level of new entrants costs for WIP's. When the existing WIP's have to close, the percentage landfilling will increase further as in that case landfilling will be the cheapest option for the 4.4 Mton of waste currently incinerated. The far lower prices compared with the current regulation regime results in a decrease of **total costs** (net of taxes) with 149 million euro (21% of total current costs) on long term. When these costs are compared with the regulation package of a 52 euro subsidy for WIP's (the best package given the chosen Dutch policy goals) cost savings rise to a level of 394 million euro (40% of total cost).

6.8 Conclusions

This chapter analyses the relation between national waste policies and international competition. First, we show what the possibilities are to achieve the specific goals the Dutch government has set in the current waste market (selfsufficiency for waste disposal, a preference for incineration above landfilling and diminishing cost). Second, we analyze whether these possibilities change when international borders are opened or closed for waste export. Finally, we show what happens when regulation instruments are set according to external costs. This makes clear what the costs are of the specific goals the government has chosen, both in terms of financial costs as in terms of possibilities to implement national regulation.

The analysis shows that the objectives chosen by the Dutch government can be reached independent of market sizes. With a prohibition of all exports, an export allowance for waste destined for reuse and export allowance for all waste a selfsufficiency percentage of 100% and landfilling of 0% is possible. However, the current regulation regime (only allowed export for reuse and a landfill tax of 75 euro) only results in the fulfillment of the chosen objectives if all borders are closed. With open borders for reuse the selfsufficiency percentage is 82%, while landfilling is 7%. The market outcomes are further away from the chosen objectives when all borders are opened. In this case selfsufficiency is only 58% and landfilling is 42% as long as the current WIP's exist and are even worse as prices are beneath the costs of a new WIP so that in the long term closed WIP's are not replaced. As the discussion in the EU both on a political level as in courtrooms may result in open borders in the long term, this is worrying. Changes of the current regulation regime might be necessary.

Fortunately, the chosen objectives are reachable when the landfill tax is abolished while at the same time a subsidy is given to WIP's. In this case even with open borders a selfsufficiency percentage of 100% and a landfilling percentage of 0% is possible. As this regulation package guarantees the fulfillment of national waste policy objectives, harmonization of the landfill tax on a EU-scale might be a better option. When all EU-members introduce a landfill tax equal to the difference in costs between landfilling and a new WIP, landfilling will be zero in the long term, while a selfsufficiency percentage is possible of nearly 90%. As subsidies are rather expensive, this option saves on costs. However, international coordination of a landfill tax is not easy to reach as most EU-members are not keen to introduce such a tax. The best policy option might be to choose for the national instrument of a WIP-subsidy (and a landfill tax of zero), while in the meantime international coordination of a landfill tax is searched for.

However, the chosen policy goals are not in accordance with economic principles. From an economic perspective internalization of external cost is the basis of regulation. When taxes for all waste treatment option are set according to the external costs, a selfsufficiency percentage of only 61% and a landfilling percentage of only 38% is optimal, while these percentages decrease and increase respectively when existing WIP's are closed. This is caused by the lower social costs of landfilling in other countries for most waste streams compared with the social costs of other options or with the social cost of landfilling in the Netherlands. Compared with the best regulation package when the current Dutch waste objectives are fulfilled, a cost saving is possible of 394 million euro per year. Savings are less when the current policy objectives are less stringent interpreted. It could be argued that for the current market (allowed export for reuse) a selfsufficiency percentage beneath 100% and a landfilling percentage above 0% is acceptable as long as the foreign treatment is limited to reuse and landfilling is necessary to facilitate this reuse (in other words as only the residue of separation is landfilled). However, even in this case the savings on total costs are 149 million euro when regulation is based on economic principles.

Appendix A Model

Waste supply

In MEAM waste suppliers (municipalities and companies) minimize treatment costs of their (not yet contracted) waste. These costs are the sum of the direct cost of treatment (including taxes) and transport costs.

In the model countries are split into different regions (the larger the country, the more regions). Within a region transport costs are constant, between regions these costs depend on the transport distance and transport mode.

The model handles only not yet contracted waste. As part of the waste is contracted for a long term (often more than 15 years), changes in market circumstances have no influence on this amount of waste. However, the size of the contracted waste stream determines the free capacity of waste treatment plants.

The model discriminates between five different waste streams: very low caloric waste with an average caloric value of 8.9 GJ per ton, low caloric waste with an average caloric value of 10.3 GJ per ton, mid caloric waste and dirty mid caloric waste with an average caloric value of 14.1 GJ per ton and high caloric waste with an average caloric value of 17,9 GJ per ton.

Very low caloric waste results when low caloric waste (generated by households) is separated. In this separation process a (relatively) small quantity of high caloric waste is produced. Mid caloric waste (generated by firms) can also be separated in a low and a high caloric part.

This is not the case for high and dirty mid caloric waste (also both generated by firms). The composition of dirty mid caloric waste is such that separation is not possible or too expensive, while high caloric waste is homogenous by definition so that separation is not necessary.

Suppliers of waste can choose between several treatment options: landfilling, incineration in a Waste Incineration Plant (WIP), incineration in a cement kiln or coal fired electricity plant or a waste separation plant. Cement kilns and coal fired electricity plants can only handle high caloric waste as mixed waste causes unwanted maintenance problems due to the technical requirements of the furnaces.

Thus, the cost function of a waste supplier from region k is

$$\begin{aligned}\min_{Q_s}\ & (p^{ki}_{landf} + t^{ki})(q^{ki}_{landf,l} + q^{ki}_{landf,m} + q^{ki}_{landf,md} + q^{ki}_{landf,h}) \\ & + \sum_{j\in\mathcal{J}}(p^{kj}_{incin,l} + t^{kj})q^{kj}_{WIP,l} + \sum_{j\in\mathcal{J}}(p^{kj}_{incin,m} + t^{kj})q^{kj}_{WIP,m} \\ & + \sum_{j\in\mathcal{J}}(p^{kj}_{incin,md} + t^{kj})q^{kj}_{WIP,md} + \sum_{j\in\mathcal{J}}(p^{kj}_{incin,h} + t^{kj})q^{kj}_{WIP,h} \\ & + \sum_{r\in\mathcal{R}}(p^{kr}_{incin,h} + t^{kr})q^{kr}_{cemcoa,h} + p^{k}_{sep,l}q^{k}_{sep,l} + p^{k}_{sep,m}q^{k}_{sep,m},\end{aligned} \tag{6.8.6}$$

with $\mathcal{I}$ the set of all landfills in the European Union (with index i), $\mathcal{J}$ the set of all WIP's in the European Union (with index j) and $\mathcal{R}$ the set of all cement kilns and coal fired electricity plants in the European Union (with index r). Furthermore, the following variables are used:

p^{ki}_{landf}	:	the price of landfilling waste from region k in landfill i,
$p^{kj}_{incin,l}$	:	the price of incinerating low caloric waste from region k in WIP j,
$p^{kj}_{incin,m}$	:	the price of incinerating mid caloric waste from region k in WIP j,
$p^{kj}_{incin,md}$	:	the price of incinerating dirty mid caloric waste from region k in
	:	WIP j,
$p^{kj}_{incin,h}$	:	the price of incinerating high caloric waste from region k in WIP j,
$p^{kr}_{incin,h}$	:	the price of incinerating high caloric waste from region k in cement kilns or coal fired electricity plant r,
$p^{k}_{sep,l}$	:	the price of separating low caloric waste from region k,
$p^{k}_{sep,m}$	:	the price of separating mid caloric waste from region k,
t^{ki}	:	the transport costs per ton of waste from region k to landfill i,
t^{kj}	:	the transport costs per ton of waste from region k to WIP j,
t^{kr}	:	the transport costs per ton of waste from region k to a cement kiln or coal fired electricity plant r,
$q^{ki}_{landf,l}$	:	the quantity low caloric waste landfilled in region k and landfill i
$q^{ki}_{landf,m}$	:	the quantity mid caloric waste landfilled in region k and landfill i
$q^{ki}_{landf,md}$	:	the quantity dirty mid caloric waste landfilled in region k and landfill i
$q^{ki}_{landf,h}$	:	the quantity high caloric waste landfilled in region k and landfill i
$q^{kj}_{WIP,l}$	:	the quantity low caloric waste incinerated in region k and WIP j,
$q^{kj}_{WIP,m}$	:	the quantity mid caloric waste incinerated in region k and WIP j,

$q^{kj}_{WIP,md}$	:	the quantity dirty mid caloric waste incinerated in region k and WIP j,
$q^{kj}_{WIP,h}$	:	the quantity high caloric waste incinerated in region k and WIP j,
$q^{kr}_{cemcoa,h}$	:	the quantity high caloric waste incinerated in region k in a cement kiln or coal fired electricity plant r,
$q^{k}_{sep,l}$	:	the quantity low caloric waste separated in region k
$q^{k}_{sep,m}$	:	the quantity mid caloric waste separated in region k
Q_s	:	$\{q^{ki}_{landf,l}, q^{ki}_{landf,m}, q^{ki}_{landf,md}, q^{ki}_{landf,h}, q^{kj}_{WIP,l}, q^{kj}_{WIP,m}, q^{kj}_{WIP,md}, q^{kj}_{WIP,h}, q^{kr}_{cemcoa,h}, q^{k}_{sep,l}, q^{k}_{sep,m}\}$.

As the total quantity of waste has to be treated somewhere (assuming no accumulation of waste outside the mentioned treatment options) a number of conditions is required. Low caloric waste has to be treated in a landfill or separation plant or a WIP. Incineration of low caloric waste in a cement kiln or coal fired electricity plant is not possible because the caloric value is not high enough. Mathematically

$$\sum_{i\in\mathcal{I}} q^{ki}_{landf,l} + \sum_{j\in\mathcal{J}} q^{kj}_{WIP,l} + q^{k}_{sep,l} = \overline{Q}^{k}_{l}, \tag{6.8.7}$$

with $\overline{Q}^{k}_{l}$ the supply of low caloric waste in region k.

Comparably, the formulae for mid caloric waste is

$$\sum_{i\in\mathcal{I}} q^{ki}_{landf,m} + \sum_{j\in\mathcal{J}} q^{kj}_{WIP,m} + q^{k}_{sep,m} = \overline{Q}^{k}_{m}, \tag{6.8.8}$$

with $\overline{Q}^{k}_{m}$ the supply of mid caloric waste in region k.

As separation of dirty mid caloric waste is not possible

$$\sum_{i\in\mathcal{I}} q^{ki}_{landf,md} + \sum_{j\in\mathcal{J}} q^{kj}_{WIP,md} = \overline{Q}^{k}_{md}, \tag{6.8.9}$$

with $\overline{Q}^{k}_{md}$ the supply of dirty mid caloric waste in region k.

As for high caloric waste all option are open

$$\sum_{i\in\mathcal{I}} q^{ki}_{landf,h} + \sum_{j\in\mathcal{J}} q^{kj}_{WIP,h} + \sum_{r\in\mathcal{R}} q^{kr}_{cemcoa,h} = \overline{Q}^{k}_{h}, \tag{6.8.10}$$

with $\overline{Q}_h^k$ the supply of high caloric waste in region k.

For different scenarios supplementary restraints can be imposed. For example, the scenario with open borders for mid and high caloric waste and closed borders for low caloric waste (which is a possibility currently discussed in the European Union) would suggest that incineration in a foreign WIP is not possible for this waste stream. Define

$$v(s,k) = \begin{cases} 1 & \text{, if region k is in country s,} \\ 0 & \text{, else,} \end{cases} \tag{6.8.11}$$

and

$$z(k,j) = \begin{cases} 1 & \text{, if WIP j is in region k,} \\ 0 & \text{, else.} \end{cases} \tag{6.8.12}$$

Thus

$$\forall\, k, j, s \; met \; z(k,j) = 1 \; en \; v(s,k) = 0 \; : \; q_{WIP,l}^{kj} = 0. \tag{6.8.13}$$

Other restriction such as landfill bans can be implemented in a comparable way.

Waste demand

In MEAM all WIP's which are currently operational in the European Union are included. It is assumed that these WIP's aim at maximizing profits on free contractable waste. However, four technical restraints apply. First, the WIP's are constrained mechanically. Per hour a maximum quantity of waste can be handled. Second, the WIP's are constrained thermically. The caloric value of the incinerated waste may not exceed a certain limit. Third, the average caloric value of the incinerated waste has to be at least equal to the minimal caloric value the WIP is able to incinerate. Fourth, the average caloric value of the waste can not exceed the maximal caloric value the WIP is able to incinerate. In the model, the boundaries on the quantity and type of waste a WIP is able to incinerate is given on short term. However, as some investments are already done, some WIP's are able to expand their capacity at low costs in the mid term. The model assumes that on the long term all WIP's can expand (at 'normal' costs) or new

WIP's can enter the market. The profit equation of the WIP's is

$$\begin{aligned}\max_{Q_d} \quad & \sum_{k\in\mathcal{K}}(p^{kj}_{incin,ll} - c^{j}_{WIP,ll})q^{kj}_{WIP,ll} + \sum_{k\in\mathcal{K}}(p^{kj}_{incin,l} - c^{j}_{WIP,l})(q^{kj}_{WIP,l} + q^{kj}_{WIP,ml}) \\ & + \sum_{k\in\mathcal{K}}(p^{kj}_{incin,m} - c^{j}_{WIP,m})q^{kj}_{WIP,m} + \sum_{k\in\mathcal{K}}(p^{kj}_{incin,md} - c^{j}_{WIP,md})q^{kj}_{WIP,md} \\ & + \sum_{k\in\mathcal{K}}(p^{kj}_{incin,h} - c^{j}_{WIP,h})(q^{kj}_{WIP,h} + q^{kj}_{WIP,lh} + q^{kj}_{WIP,mh}),\end{aligned} \tag{6.8.14}$$

with $\mathcal{K}$ the set of all regions in the European Union and

$c^{j}_{WIP,ll}$	:	the variable costs of incinerating very low caloric waste in WIP j,
$c^{j}_{WIP,l}$	:	the variable costs of incinerating low caloric waste in WIP j,
$c^{j}_{WIP,m}$	:	the variable costs of incinerating mid caloric waste in WIP j,
$c^{j}_{WIP,md}$	:	the variable costs of incinerating dirty mid caloric waste in WIP j,
$c^{j}_{WIP,h}$	:	the variable costs of incinerating high caloric waste in WIP j,
$q^{kj}_{WIP,ll}$	:	the quantity of very low caloric waste from region k (out of the separation of low caloric waste) incinerated in WIP j
$q^{kj}_{WIP,ml}$	:	the quantity of low caloric waste from region k (out of the separation of mid caloric waste) incinerated in WIP j
$q^{kj}_{WIP,lh}$	:	the quantity of high caloric waste from region k (out of the separation of low caloric waste) incinerated in WIP j
$q^{kj}_{WIP,mh}$	:	the quantity of high caloric waste from region k (out of the separation of mid caloric waste) incinerated in WIP j
Q_d	:	$\{q^{kj}_{WIP,ll},\ q^{kj}_{WIP,l},\ q^{kj}_{WIP,ml},\ q^{kj}_{WIP,m},\ q^{kj}_{WIP,md},\ q^{kj}_{WIP,lh},\ q^{kj}_{WIP,mh},\ q^{kj}_{WIP,h}\}$.

This formulae implies that the variable costs of the treatment options determine the prices. For existing plants these costs are equal to the operational costs as fixed costs have no influence on the market price. However, in the long term scenario variable costs for new plants are equal to the total costs of treatment. As costs are included per ton we assume that all costs in the long term depend on the quantity of waste treated and that scale effects play no role. Although scale effects do play a role in practice, it is assumed that new plants enter at their optimal scale.

The actual mechanical quantity treated in WIP j is equal to

$$M^j = \sum_{k \in \mathcal{K}} (q^{kj}_{WIP,l} + q^{kj}_{WIP,m} + q^{kj}_{WIP,md} + q^{kj}_{WIP,h} + q^{kj}_{WIP,ll} + q^{kj}_{WIP,ml} + q^{kj}_{WIP,lh} + q^{kj}_{WIP,mh}), \tag{6.8.15}$$

while the actual thermic quantity treated is equal to

$$T^j = \sum_{k \in \mathcal{K}} (\epsilon_{ll} q^{kj}_{WIP,ll} + \epsilon_l (q^{kj}_{WIP,l} + q^{kj}_{WIP,ml}) + \epsilon_m q^{kj}_{WIP,m} + \epsilon_{md} q^{kj}_{WIP,md} + \epsilon_h (q^{kj}_{WIP,h} + q^{kj}_{WIP,lh} + q^{kj}_{WIP,mh})), \tag{6.8.16}$$

Thus, the mechanic constraint of WIP j is given by

$$M^j \leq K^{ton}_j, \tag{6.8.17}$$

with K^{ton}_j the capacity in kiloton (kton) of WIP j. The thermic constraint is given by

$$T^j \leq K^{therm}_j, \tag{6.8.18}$$

with K^{therm}_j the capacity in GigaJoule (GJ) per hour of WIP j. The minimal caloric value constraint is given by

$$\frac{T^j}{M^j} \geq \xi^j_{low}, \tag{6.8.19}$$

with ξ^j_{low} the minimal caloric value that WIP j is able to incinerate. The maximal caloric value constraint is given by

$$\frac{T^j}{M^j} \leq \xi^j_{high}, \tag{6.8.20}$$

with ξ^j_{high} the maximal caloric value that WIP j is able to incinerate.

In MEAM not only the WIP's are included, but also cement kilns and coal fired electricity plants with respect to their capacity to incinerate (high caloric) waste. The assumption is that these plants can substitute 'normal' fuel for waste with a maximum of a certain percentage of the total fuel use. It is assumed that cement kilns and coal fired electricity plants also maximize their profits. That is

$$\max \sum_{k\in\mathcal{K}} p^{kr}_{incin,h}(q^{kr}_{cemcoa,h} + q^{kr}_{cemcoa,lh} + q^{kr}_{cemcoa,mh}) \\ - (c^{r}_{cemcoa,h})(q^{kr}_{cemcoa,h} + q^{kr}_{cemcoa,lh} + q^{kr}_{cemcoa,mh}), \quad (6.8.21)$$

with

$c^r_{cemcoa,h}$: the variable costs of incinerating high caloric waste in a cement kiln or coal fired electricity plant r.

The capacity constraint of cement kilns and coal fired electricity plants is

$$\sum_{k\in\mathcal{K}}(q^{kr}_{cemcoa,h} + q^{kr}_{cemcoa,lh} + q^{kr}_{cemcoa,mh}) \leq \beta_{cemcoa}K^{r}_{cemcoa}, \quad (6.8.22)$$

with β_{cemcoa} the percentage of 'normal' fuels which can be substituted for high caloric waste and K^r_{cemcoa} the total fuel use of cement kiln or coal fired electricity plant r (converted to tons of high caloric waste using the average caloric values).

The separation plants in MEAM separate waste streams in a low and high caloric fraction. After separation the low caloric waste can be incinerated in a WIP, while the high caloric fraction can be incinerated in a WIP, a cement kiln or a coal fired electricity plant. The profit maximizing equation gives for the low caloric waste separation plant

$$\max_{q^k_{sep,l}} (p^k_{sep,l} - c_{sep,l})q^k_{sep,l} - \sum_{j\in\mathcal{J}}(p^{kj}_{incin,ll} + t^{kj})q^{kj}_{WIP,ll} - \\ \sum_{j\in\mathcal{J}}(p^{kj}_{incin,h} + t^{kj})q^{kj}_{WIP,lh} - \sum_{r\in\mathcal{R}}(p^{kr}_{incin,h} + t^{kr})q^{kr}_{cemcoa,h} \quad (6.8.23)$$

$$\text{z.d.} \sum_{j\in\mathcal{J}} q^{kj}_{WIP,ll} = (1-\theta_{lh})q^k_{sep,l} \\ \sum_{j\in\mathcal{J}} q^{kj}_{WIP,lh} + \sum_{r\in\mathcal{R}} q^{kr}_{cemcoa,lh} = \theta_{lh}q^k_{sep,l}. \quad (6.8.24)$$

Separation plants for mid caloric waste can be modelled in the same way. Used variables are

θ_{lh} : the percentage of high caloric waste that can be separated from low caloric waste,

θ_{mh} : the percentage of high caloric waste that can be separated from mid caloric waste,

$c_{sep,l}$: costs of the separation of low caloric waste,

$c_{sep,h}$: costs of the separation of high caloric waste.

Both for separation plants and landfills no capacity constraints are assumed.

Regulation

Countries differ with respect to the regulation with which they try to influence the way waste streams are treated. The main instruments are landfill taxes, incineration taxes and landfill bans. MEAM can handle these instruments. Landfill bans are included using the procedure described in equation 6.8.12 and 6.8.13. Taxes are included in the variable costs of landfill (landfill tax) and incineration (incineration tax).

Simulation

The most important variables calculated by the model are the market prices for the different waste streams and the allocation of waste to the different treatment plants. Givens these prices and quantities a number of variables can be calculated: the utilization rate per plant, the margin on a treated ton of waste, the quantity of exports and imports, the quantity of waste separated, the way waste is treated and the newly build capacity.

The model makes it possible to vary the institutional environment. Not only is it possible to include different regimes of waste regulation, but also the type of waste for which national borders are opened can vary. Furthermore, the model is simulated for the short, mid and long term.

Appendix B Data

In this appendix we explain the data used for the model simulations. Table 6.10 presents the variables which are exogenous for the model.In column two the page is indicated in which the explanation of the exogenous variables starts.[21]

Table 6.10: Overview exogenous variables

Variables	Page
Transport costs	150
Regions per country	150
Costs and capacity landfilling	151
Variable costs incineration WIP's: existing WIP's	153
Total costs incineration WIP's: new WIP's	157
Total costs cement kilns and coal fired electricity plants	159
Total costs separation	159
Capacity existing WIP's	159
Contracted capacity existing WIP's	159
Capacity cement kilns and coal fired electricity plants	159
Waste supply	161
Waste regulation (landfill tax, incineration tax and landfill ban)	164

Collection of the necessary data makes clear that availability of data is a serious problem in Europe. For almost every variable unambiguous, orderly and accessible sources are not available. Information from a lot of sources has to be combined to shed light on the actual situation. Thus, a number of assumptions have to be made to quantify the variables. As this increases the uncertainty of the model outcomes, we simulate the model for different values of the variables. Thus, not only a basic value is given for most variables but also a minimum and maximum value. In this appendix we call the set of basic values the 'standard scenario', while the sets of minimum and maximum values are called the 'sensitivity analyses'.

Transport costs and regions

Transport costs included in the model are minimal costs given the available transport modes. We reckon with the way waste is collected (with a traditional system or a container system), different ways of main transport (bulk and container with lorries, train, inland and maritime shipping), costs of door-to-door delivery and of transshipment. We

[21] More details can be found in Dijkgraaf et al. (2001).

assume that:

1. Transportation by train is no option given its high costs.
2. Maritime shipping is no alternative for waste that is not dried because the biological consequences of longer transports than 48 hours are not acceptable.
3. Inland shipping is an alternative for transport between the Netherlands, parts of Germany, Belgium, Austria and France given the availability of infrastructure.
4. Lorry transport is the main way of transport.

The main scenario assumes that all waste is transported by lorry. In the sensitivity analyses other transport modes are also analyzed. Uncertainty in door-to-door delivery costs (especially regarding the distance between the place of collection and the start of the main transport as well as the distance between the end of the main transport and the place of treatment) are captured by simulating a cheap and an expansive case.

As it is not workable to calculate the transport costs between each possible place of collection and treatment in Europe we use regions. A specific municipality is chosen per region to represent the region. We distinguish more regions for large countries (four at most) than for small countries (one at least).

To calculate the transport costs between regions we used a route planner to calculate the distance between regions as well as the transport costs equation

$$C_T = 0.045D_L + 4.54 + 0.004D_F + 8.85N_F \tag{6.8.25}$$

The transport costs (C_T) from the place of collection to a treatment plant are equal to the sum of 4.5 eurocents times the distance travelled by lorry (D_L) plus 4.54 euro (door-to-door delivery and transshipment), 0.4 eurocents times the distance travelled by ferry (D_F) and 8.85 times the number of ferries used (N_F). Thus, within a region the costs of transport are 4.54 euro.

Costs landfilling

Table 6.11 gives an overview of landfill tariffs in the European Union excluding VAT and landfill taxes. Big differences exist between countries. One reason could be that

Table 6.11: Landfill tariffs in the EU in euro per ton (excl. taxes)

	Minimum tariff	Maximum tariff	Average tariff
Belgium	17	69	41
Denmark	3	161	27
Germany	25	50	53
Finland	6	24	13
France	40	99	53
Greece	6	13	9
Ireland	36	na	35
Italy	23	na	na
Netherlands	41	114	54
Austria	36	215	93
Portugal	6	14	na
Spain	14	14	na
UK	20	35	27
Sweden	4	200	30

Sources: Kossina (2000), EEA (2000) and own calculations.

countries differ with respect to environmental measures taken. However, as countries differ also with respect to population density and stability of the soil (especially regarding the leakage effect to groundwater) differences in facilities, and thus costs, need not lead to differences in environmental performance. Furthermore, environmental regulation is harmonized between EU-countries as a consequence of the Landfill Directive (COM(99)31). For the model simulations we assume that the presented minimal tariffs mirror variable costs. Sensitivity analyses are carried out with variable costs 11 euro higher and lower per country.

The assumed variable costs are also used for the long term version of the model. In general available landfill capacity is very large in comparison with the waste supply (in the Netherlands 78 Mton in 1999 compared with a yearly landfill of 7 Mton). Thus, investments in new capacity are not necessary in the near future. Furthermore, it is plausible that minimum tariffs mirror also full costs for new landfill capacity. The reason for this is that for landfills the relation between time and capacity is very loose. If capacity is not used in one period, the same capacity is still available at an other period. Losses for the landfill are limited to the funding of investments, which are relatively low due to the fact that most investments only have to be done just before actual use. Thus, landfill owners have not much incentives to ask tariffs that are beneath total costs.[22] For the Netherlands this assumption can be tested on the basis of a

[22] This implies that landfills that can ask the maximum tariffs mentioned in Table 6.11 make huge profits. This is indeed the case in the Netherlands.

comparison between the minimum tariff presented in Table 6.11 and total costs of a new landfill. The Ministry of Environmental Affairs calculated that the total costs of a new landfill were 34 euro per ton in 1992 (VROM, 1992). With an average inflation of 2.5% per year this compares to 43 euro per ton in 2002, which is nearly equal to the minimum tariff of Table 6.11. As environmental regulation did not change between 1992 and 2002, it is plausible to assume that both for existing and new landfills the variable costs are about 41 euro per ton.

Variable costs WIP's: existing WIP's

For the Netherlands detailed information on the costs of existing WIP's is available (AOO, 1997). Table 6.12 gives an overview of total and variable costs of Dutch existing WIP's. The variable costs are used in the model simulations.[23] As the presented variable costs are dependent on the quantity of energy sold, variable costs for incineration of very low, mid or high caloric waste are corrected using a linear relationship between energy sold and caloric value of the incinerated waste.

Table 6.12: Costs existing Dutch WIP's in euro per ton

Name of waste incineration plant	Total costs	Variable costs
GAVI-Wijster	110	54
AVI-Twente	98	35
AVIRA-Arnhem	79	32
ARN-Nijmegen	120	44
HVC-Alkmaar	90	11
AVI-Amsterdam	73	22
AVR-Rijnmond	64	27
AVI-Rotterdam	86	34
GEVUDO-Dordrecht	90	44
AZN-Moerdijk	91	28

Source: AOO (1997)

Given the quality of the AOO (1997) study discussion is possible about the representativeness of the used data. Using annual reports we tried to verify the quality of the data. Only for AVI-Amsterdam and HVC-Alkmaar a careful check was possible. This indicated that current figures are not very much different (respectively 2 and 3 euro per ton) from the data presented in Table 6.12. Sensitivity analysis with 10 euro lower

[23] As the variable costs of WATCO-Roosendaal are not given in AOO (1997), they are estimated on the same way as the costs for incineration plants in other countries.

and higher variable costs will be used to test whether conclusions are sensitive for these assumptions.

For other countries not much public information is available. Although various sources present information about the tariffs in EU-countries (see Table 6.13), this information is rather unreliable (mostly based on expert opinions). Furthermore, results differ per source and the range in tariffs per country gives no basis to assume a sound relation between these tariffs and the variable costs of individual installations.

Table 6.13: Tariffs incineration in EU-WIP's in euro per ton (excl. taxes)

Country	National	TIP	Senat	EEA
Belgium	53-110	74	94	44-97
Denmark	14-39	44	60	27
Germany	36-340	89-349	184	88
France	60-98	74-79	62	69-129
Italy		64-79		
Netherlands	69-125	109-140	96	64-119
Austria		99-180		103
Spain	16-52			34
UK		24-39		49
Sweden		29	38	35

Sources: TIP (1999), Senat (2000), EEA (2000) and national sources.

To estimate the variable costs of incineration in EU-WIP's a two step procedure is followed. First, cost information is collected for individual WIP's. We have send all EU-WIP's (with the exception of the French and Italian WIP's because individual figures or addresses where not available) a request for annual reports as well as an inquiry. Furthermore, for the Italian WIP's an useful study exists (Ecoistituto Veneto, 2000). Second, we estimated a statistical model with the data we obtained to extrapolate cost information for WIP's for which no cost data were available.

Table 6.14 presents the estimated cost model. Variable costs[24] are influence by the capacity of the plant. As the quadratic therm is also significant, a U-shaped cost function applies (see figure 6.2). First, variable costs decrease when capacity is increased. However, after a certain optimal scale variable costs increase. It shows that for our sample an installation of about 600 kton has the lowest costs.

The estimation shows clear cost differences between countries. Especially Danish, Italian, Spanish and Swedish installations have far lower costs than Dutch installations (the reference used in the estimation). Underlying data make clear that these cost

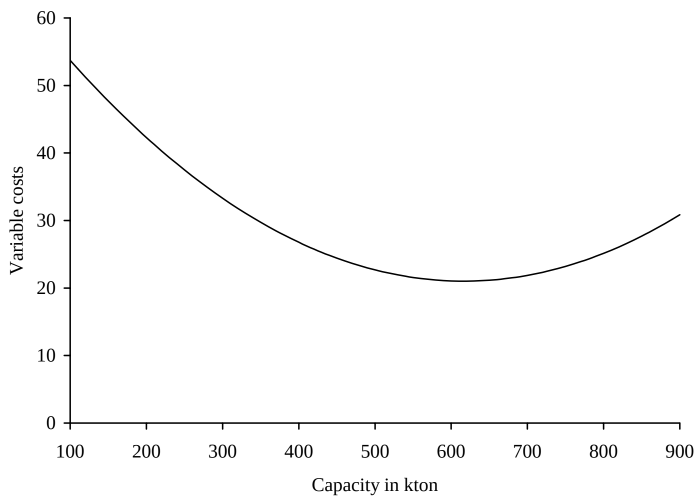
[24] Defined as total costs minus depreciation and costs of capital

Table 6.14: Estimated variable cost function EU-WIP's

Variable	Description	Coefficient	St. error	t-value
Q	Capacity	-0.151	0.050	-3.03
Q^2	Capacity squared	0.00012	0.00005	2.45
D_{Bel}	Dummy Belgium	-9	15	-0.56
D_{Dnk}	Dummy Denmark	-34	13	-2.50
D_{Ger}	Dummy Germany	29	10	2.67
D_{UK}	Dummy UK	-7	13	-0.63
D_{Ita}	Dummy Italy	-47	13	-3.55
D_{Spa}	Dummy Spain	-26	11	-2.28
D_{Swe}	Dummy Sweden	-58	17	-3.26
D_{Swi}	Dummy Switserland	0	11	0.00
C	Constant	68	13	5.14
R^2		0.53		

differences have a clear reason. First, in Denmark, Italy and Sweden the revenues from electricity production are far higher than in the Netherlands. In Denmark and Sweden this is due to the utilization of heat of the WIP's. Italy has lower costs because the price per unit electricity produced is more than twice the price in the Netherlands. The low costs of Spain are mainly caused by the lack of environmental facilities. Plants in Belgium, Germany, the UK and Switzerland have more comparable costs with the Netherlands.

Figure 6.2: Variable costs WIP's in euro per ton waste (all dummies zero)



As the capacity is available for all WIP's, the estimated cost function makes it possible to calculate the variable costs of all individual WIP's. For France and Austria it is assumed that the fixed part of the estimated equation is 10 and 6 euro higher than in the Netherlands, based on the average tariffs presented in Senat (2000) and EEA(2000).

Table 6.15 presents an overview of the maximum, minimum and average variable costs used in the model simulations.

Table 6.15: Estimated variable costs of WIP's in euro per ton

Country	Maximum	Minimum	Average (unweighted)
Belgium	54	23	44
Denmark	31	-9	21
Germany	93	50	66
France	49	12	38
Italy	19	-16	11
Netherlands	44	21	31
Austria	63	40	50
Spain	36	-3	18
UK	55	12	34
Sweden	9	-30	-2

The estimated and calculated variable costs apply to the incineration of low caloric waste. However, as the model distinguish waste streams of several caloric value, we have to estimate also the variable costs for other waste streams. We assume that the relation between energy revenue and caloric value is linear. Note that this does not mean that incineration of waste with a higher caloric value is always cheaper. The reason for this is that the total capacity of the WIP decreases when the average value of the caloric value of incinerated waste increases (see the model description).

For the Spanish installations we mentioned that less equipment for emissions resulted in lower costs. As it is not clear whether the existing installations comply with the EU-directive, this could also apply to other countries. Therefore, we simulate also a sensitivity analysis for the case that installations in some countries have to improve their plant. Based on EC (1996) and information from the inquiry we simulate a scenario with higher variable costs for installations that possibly do not comply with the EU-directive.[25] As we only for Germany and the Netherlands know for sure that all installations comply, this scenario reckons with higher costs for installations in all other countries.

Furthermore, uncertainty in the estimated variable costs is handled by simulating scenarios with higher or lower variable costs (11 euro) per country.

[25] EC(1996) gives information on the renovation costs of installations that do not comply. As these costs are dependent on the scale of the plant, we estimated a renovation cost equation that counts for this property and calculate the costs for each individual installation using data on their capacity.

Total costs incineration: new WIP's

In the long term scenario new plants can enter the market. The crucial variable for this is the total costs of a new entrant. When these costs are below market price, new plants will enter the market. Unfortunately no data are available on the costs of new plants. Therefore, we build on the experience we have with Dutch installations and count for the differences between countries.

Table 6.16: Assumptions costs new Dutch WIP

Nr.	Assumption	Standard scenario
1.	Investment (in mln euro)	436
2.	Mechanic capacity (kiloton at 100% availability)	648
3.	Availability (as % of total capacity)	87.5
4.	Company capital (as % of total capital)	40
5.	Net return on company capital (%)	12
6.	Interest outside capital (%)	7
7.	Tax on profits (%)	32
8.	Inflation (%)	2
9.	Efficiency improvement (as % of total costs)	15
10.	Economic life span (in year)	25
11.	Depreciation term (in years)	15
12.	Reinvestment costs (in euro per ton)	4
13.	Variable costs (in euro per ton)	39
14.	Energy revenue (in euro per ton)	26

Table 6.16 summarizes the assumptions we made for the costs of a new Dutch WIP. As a reference we used the figures of a recent build Dutch WIP (HVC-Alkmaar). In the Netherlands this WIP is seen as an example of newest technology, complying with the new EU-directive. Furthermore, this WIP has the optimal scale we find in our estimation of foreign WIP's. According to the Dutch Waste Processing Association the used incineration technology (a grid) is the cheapest option available. This is confirmed by KEMA (1995) and CADDET (1998).

As the reference WIP is build in a market environment that can be classified as a protected monopoly and as the plant is rather luxury, we assume a possible efficiency improvement of 15%.[26] Based on a yearly constant annuity the costs of a new entrant per ton are 77 euro.[27] As uncertainty exist about the used assumptions we also calculated the total costs with other assumptions. This results in minimum entrants costs

[26] Assumptions and outcome are verified by the Society of Dutch Waste Treatment Firms, VVAV.

[27] Although the reference plant is a low caloric waste incineration plant, calculations show that costs for mid of high caloric incineration in a WIP are comparable.

of 63 euro[28] and maximum costs of 94 euro.[29]

Table 6.17: Total costs new WIP-plants in the European Union in euro per ton

	Basis[a]	A[b]	B[c]	C[d]	D[e]	E[f]
Belgium	74	74	75	75	91	60
Denmark	73	43	45	76	91	59
Germany	81	81	81	81	98	66
Finland	71	71	74	74	88	57
France	77	77	78	78	95	63
Greece	61	61	65	65	78	47
Ireland	67	67	68	68	83	53
Italy	73	33	35	76	90	58
Luxembourg	88	88	79	79	107	74
Netherlands	76	76	76	76	93	62
Austria	76	76	73	73	93	62
Portugal	60	60	63	63	76	46
Spain	66	66	70	70	83	52
UK	72	72	74	74	88	58
Sweden	73	43	44	75	90	59

[a] Standard: different landfill costs incineration residue, equal energy revenues.
[b] Analysis: different landfill costs incineration residue, equal energy revenues.
[c] Analysis: equal landfill costs incineration residue, different energy revenues.
[d] Analysis: equal landfill costs incineration residue, energy revenues.
[e] Analysis: high costs for new plants.
[f] Analysis: low costs for new plants.

For the entrants costs of other countries we use as a reference the Dutch costs, but correct for country specific circumstances. The following corrections are made:

1. Capital costs are calculated with the national effective tax rate on profits.

2. Variable costs are calculated with the national average wage rate.

3. Total costs are calculated taking into account the differences with respect to environmental facilities. In practice this means that total costs for most countries are about 4 euro less than in the Netherlands because the Netherlands has stricter environmental requirements for NO_x.

4. Energy revenues are calculated assuming no difference between countries. Although the energy revenue in Italy, Denmark and Sweden currently differ from other countries, it is not reasonable to assume the same conditions for new plants as

[28] Availability is 90%, company capital 30%, interest outside capital 6%, depreciation term 25 years, energy revenue 20% higher

[29] Availability is 85%, net return on company capital 13%, interest outside capital 8%, efficiency improvement 10%, energy revenue 20% lower

the infrastructure to take advantage of the produced heat (Denmark and Sweden) is often not available for new plants, while the high energy price (Italy) is not sustainable in the future because this can lead to very large imports when waste borders are open.

Table 6.17 present the total costs of new WIP-plants in the European countries. The second column give the figures for our standard scenario. The columns three, four and five represent figures for sensitivity analysis scenarios. In these scenarios we vary with the degree of harmonization of the costs of landfilling the incineration residue and the energy revenue. Furthermore, the low and high cost scenario are given in the last two columns.

Total costs incineration: other options

For incineration of waste in a cement kiln or coal fired electricity plant waste has to be pretreated which leads to additional costs. Separation, drying, shredding and compressing amounts to a minimal cost of 18 euro per ton. Furthermore, dependent on the available facilities, extra investments can be necessary to comply with environmental regulation. On the other side the 'normal' fuel costs are avoided. We assume that minimal costs for incineration in a cement kiln or coal fired electricity plants amounts to 18 euro per ton. In a sensitivity analysis we reckon with 36 euro per ton.

Capacity incineration

MEAM reckons with the individual WIP's.[30]. Thus we need the individual WIP capacity data. Partly based on BM (2000) we made a complete list of the mechanical capacity of all EU plants (including plants to be build).

Data on the thermic capacity and the other caloric constraints are not available for individual installations outside the Netherlands. Therefore, we use the average numbers of the Dutch plants for all EU plants. The assumption is that the average thermic capacity is 9.35 GJ per ton. The minimal caloric value is 6.4 GJ per ton and the

[30] To make the model workable individual WIP's are sometimes combined when their costs are comparable.

Table 6.18: Capacity WIP's, coal fired electricity plants and cement kilns in Mton

Country	Coal-fired plants	Cement kilns	Incineration plants	Total
Belgium	0.9	0.5	1.7	3.1
Denmark	1.5	0.6	3.0	5.1
Germany	18.0	9.8	20.7	48.5
Finland	1.3	0.6	0.1	2.0
France	1.5	1.8	11.8	15.1
Greece	1.8	0.8	0.0	2.6
Ireland	0.5	0.1	0.0	0.6
Italy	1.2	1.1	3.9	6.2
Luxembourg	0.0	0.0	0.2	0.2
Netherlands	1.5	0.6	5.4	7.5
Austria	0.3	0.3	0.5	1.1
Portugal	0.7	0.3	0.0	1.0
Spain	3.5	1.9	1.8	7.2
UK	6.5	4.5	5.6	16.6
Sweden	0.2	0.1	2.0	2.3
EU	39.4	23.0	56.8	119.2

maximum value is 12.9 GJ per ton. In sensitivity analyses we reckon with 10% higher and lower values.

Comparing the utilization rates of different WIP's in countries with not much incentives to incinerate, assuming an average contract rate of 70% seems reasonable. In sensitivity analyses we reckon with 10%-point lower and higher values.

The potential capacity of coal fired electricity plants is calculated based on the amount of actual used coal in 1997 (based on OECD, 2000). The assumption is that 10% of the total energy use can be substituted for waste. This is in line with a study in the US (EPA, 2000). In the same way the potential capacity of cement kilns is estimated. These figures are based on Global (2000). Following Vanderborght (2000) and AOO (1996) fossil fuels can be substituted for waste at a maximum percentage of 30% in the cement process.

Table 6.18 presents the total capacity in Mton per year of WIP's (assuming a 87.5% availability rate), cement kilns and coal fired electricity plants. Especially in the UK and Germany the capacity of coal fired electricity plants and cement kilns is very big in absolute terms and even more than the capacity of WIP's. Only Luxembourg and Sweden have a cement kiln and coal fired electricity plant capacity that is relatively low. Only Germany, France and the UK have more WIP capacity than the Netherlands, while four countries have no WIP-capacity at all.

Table 6.19: Waste supply 2001 in the European countries (Mton per year)

	MW	IW	DW	SW	Total	Cap-WIP	Cap-tot
Belgium	3.1	1.9	0.3	1.1	6.4	1.7	3.1
Denmark	3.0	1.3	0.2	0.7	5.2	3.0	5.1
Germany	25.0	15.7	2.3	9.0	52.0	18.1	45.9
Finland	2.1	3.2	0.1	0.6	6.0	0.1	2.0
France	23.8	34.2	1.6	6.2	65.8	11.8	15.1
Greece	5.3	4.9	0.1	0.5	10.8	0.0	2.6
Ireland	1.9	4.0	0.1	0.4	6.4	0.0	0.6
Italy	28.8	44.5	1.2	4.8	79.3	3.6	5.9
Luxembourg	0.1	0.1	0.0	0.1	0.3	0.2	0.2
Netherlands	4.8	3.0	0.4	1.7	9.9	5.4	7.5
Austria	2.5	1.6	0.2	0.9	5.2	0.5	1.1
Portugal	5.0	4.4	0.1	0.5	10.0	0.0	1.0
Spain	19.8	23.6	0.7	2.6	46.7	1.8	6.4
UK	29.7	54.0	1.5	5.8	91.0	1.6	12.6
Sweden	3.6	5.6	0.3	1.0	10.5	2.0	2.4
Total	157.1	202.0	9.1	35.9	404.1	49.8	111.5

Waste supply

In MEAM the quantity of waste available for incineration is an important variable. However, as public sources do not present enough data to determine this quantity directly, we follow an indirect approach. First, we collect the necessary data for the Netherlands. Second, we extrapolate these data to other countries.

The quantity of Dutch waste is based on 'standard' figures for household and commercial waste. As the model is forward looking a projection for the future is necessary. The short term simulations are based on actual figures (2001), while we choose 2011 as reference for the long term. The Dutch Waste Management Council made projections for us for these years (see AOO, 2000), using a standard waste model.[31]

MEAM distinguishes between low, (dirty) mid and high caloric waste. Based on the average caloric value of the different waste streams and the potential to separate the waste we calculated the quantities for these waste streams. The following assumptions are made:

- Municipal waste (MW) is low caloric waste (10.3 MJ per kilo). Separation in a

[31] In these projections total waste supply is estimated. Thus, not only the waste supply for incineration and landfilling is estimated, but also the waste supply for composting and recycling. As recycling is no part of this analysis, we only present the figures for supply available for incineration and landfilling. Thus, for 2011 1.44 Mton compostable household waste, 1.37 Mton recyclable household waste and 39,35 Mton recyclable industrial waste are excluded from the analysis.

high (plastics, wood) and very low caloric part (8.9 MJ per kilo) is possible for 15% of the total waste stream.

- Commercial waste (CW) can be split in a mid caloric (14.1 MJ per kilo), dirty mid caloric (also 14.1 MJ per kilo) and high caloric part (17.9 MJ per kilo). Sources of commercial waste are industrial waste (IW), demolition waste (DW) and sludge waste (SW).
- High caloric waste, 50% of total commercial waste, can be incinerated in cement kilns and coal fired electricity plants without major pre-handling.
- Mid caloric waste, 30% of total commercial waste, can be separated in a low (50%) and high caloric part. The high caloric part can be incinerated in cement kilns and coal fired electricity plants.
- Dirty mid caloric waste, 20% of total commercial waste, can not be separated due to major contamination making incineration in cement kilns and coal fired electricity plants not possible.
- Waste available is total waste generated minus recycling.

For the other countries the quantity of household waste is extrapolated from the Dutch figures using the quantity of waste per inhabitant, where commercial waste is extrapolated using the quantity of waste per unit of national income.[32]

Table 6.19 presents the waste supply figures for 2001. Compared with the total EU-capacity of incineration, a very large amount of waste is generated. Especially France, Italy, Spain and the United Kingdom have an enormous shortage of incineration capacity. In eleven countries the quantity of household waste exceeds the total incineration capacity. Only Denmark shows a balance tween capacity and supply.

Tables 6.20 and 6.21 present the waste supply figures for 2011. For 2011 we use a minimum and maximum scenario. In the minimum scenario it is assumed that all countries reach in 2011 the high Dutch recycling figures, resulting in less waste available for landfilling and incineration. The maximum scenario assumes that the recycling figures rise, but not more than 50% of the gap between the current percentage and the Dutch expected percentage in 2011. Therefore, in this scenario more waste is available

[32] Figures for the number of inhabitants and national income come from Eurostat (2000). For example, the estimated number of inhabitants in 2011 for the Netherlands is 16.7 million (in 2001: 15.9 million), while GDP grows on average 3.66% in these years.

for disposal options. The Tables make clear that the main conclusion remains valid on long term: the total available capacity is in nearly all countries far less than the supply of waste.

Table 6.20: Waste supply in 2011 in Mton per year (minimum scenario)

	MW	IW	BW	SW	Total	Cap-WIP	Cap-tot
Belgium	3,5	1,7	0,3	1,0	6,5	1,7	3,1
Denmark	3,0	1,1	0,2	0,7	5,0	3,0	5,1
Germany	28,4	13,2	2,1	8,0	51,7	20,7	48,5
Finland	1,8	1,1	0,2	0,7	3,8	0,1	2,0
France	20,6	10,1	1,6	6,1	38,4	11,8	15,1
Greece	3,7	0,9	0,1	0,5	5,2	0,0	2,6
Ireland	1,3	1,2	0,2	0,7	3,4	0,0	0,6
Italy	19,3	6,6	1,1	4,0	31,0	3,9	6,1
Luxembourg	0,2	0,2	0,0	0,1	0,5	0,2	0,2
Netherlands	5,6	2,9	0,5	1,8	10,8	5,4	7,5
Austria	2,8	1,4	0,2	0,9	5,3	0,5	1,1
Portugal	3,5	0,8	0,1	0,5	4,9	0,0	1,0
Spain	13,5	4,5	0,7	2,7	21,4	1,9	7,2
UK	20,1	8,6	1,4	5,2	35,3	5,6	16,6
Sweden	3,1	1,6	0,3	1,0	6,0	2,0	2,4
Total	129,2	55,9	9,0	33,9	228,0	56,9	119,2

Table 6.21: Waste supply in 2011 in Mton per year (maximum scenario)

	MW	IW	BW	SW	Total	Cap-WIP	Cap-tot
Belgium	3,6	1,8	0,3	1	6,7	1,7	3,1
Denmark	3,0	1,2	0,2	0,7	5,1	3,0	5,1
Germany	29	14,2	2,1	8	53,3	20,7	48,5
Finland	2,1	2,6	0,2	0,7	5,6	0,1	2,0
France	24,5	23,3	1,6	6,1	55,5	11,8	15,1
Greece	5	3,2	0,1	0,5	8,8	0,0	2,6
Ireland	1,7	4,2	0,2	0,7	6,8	0,0	0,6
Italy	26,1	23,1	1,1	4	54,3	3,9	6,1
Luxembourg	0,2	0,2	0	0,1	0,5	0,2	0,2
Netherlands	5,6	2,9	0,5	1,8	10,8	5,4	7,5
Austria	2,8	1,5	0,2	0,9	5,4	0,5	1,1
Portugal	4,7	2,8	0,1	0,5	8,1	0,0	1,0
Spain	18,3	15,8	0,7	2,7	37,5	1,9	7,2
UK	27,3	30,3	1,4	5,2	64,2	5,6	16,6
Sweden	3,7	3,7	0,3	1	8,7	2,0	2,4
Total	156,5	130,8	9,0	33,9	330,2	56,9	119,2

Because we followed a necessary but rough estimation process for the standard scenario, a number of sensitivity analyses are simulated. In the standard simulations we use the figures of the minimumscenario. The figures of the maximumscenario are used in sensitivity analysis. Furthermore, sensitivity analysis are done with other figures for the split in low, (dirty) mid and high caloric waste and the quantity of total waste generation. Therefore, not only the uncertainty about the figures used is captured but also the possible indirect effects of the incineration and landfilling market on recycling and prevention.

Waste regulation

Table 6.22 presents the waste stream steering regulation in the different countries. Eleven countries do have a landfill tax, but the level of this tax shows a high deviation. For Wallonia, Italy and Luxembourg it is assumed that the landfill tax is zero or equal to the presented minimum.

Table 6.22: Waste regulation in the EU (tariffs in euro per ton)

	Landfill tax	Incineration tax	Landfill ban
Belgium (Flanders)	45[a]	25	Yes
Belgium (Wallonia)	Yes	No	No
Denmark	60	28	Yes
Germany	No	No	Yes (2005?)
Finland	15	No	Yes (2005)
France	6	No	Yes (2005?)
Greece	No	No	No
Ireland	No	No	No
Italy	5-26[b]	No	No
Luxembourg	>23	n.a.	n.a.
Netherlands	75	0	Yes
Austria	15	No	Yes (2004)
Portugal	No	No	No
Spain	No	No	No
UK	20[c]	No	No
Sweden	30	No[d]	Yes (2002)

Sources: 54 different sources

[a] Tariff for municipal waste.

[b] Tariffs differ per region. Commercial waste: min. 5 and max. 10 euro per ton. Municipal waste: min. 10 and max. 26 euro per ton.

[c] Tariff in 2000. Per year an one pound increase is agreed.

[d] A discussion is going on about an incineration tax.

An incineration tax is less common. Only Flanders and Denmark have an incineration tax. Flanders, Denmark and the Netherlands have a landfill ban. However, the Dutch experience show that a landfill ban is a very difficult instrument to implement. As long as the total waste supply is more than capacity, exemptions have to be given to waste suppliers not able to incinerate their waste. If landfill taxes are not high enough, each waste supplier has incentives to ask for an exemption. Thus, suppliers can profit from the lower landfill tariff. For this reason we assume in the standard scenario that a landfill ban has no effect on the quantity of waste going to WIP's. In sensitivity analysis it is shown what the effects are if a landfill ban has positive effects. In the mid and long term scenarios a landfill ban is also simulated for Sweden, Austria, Finland, Germany and France. For these countries a landfill ban is expected to come in practice before 2005.

Appendix C Sensitivity analysis

In this appendix detailed results of the sensitivity analysis are presented for the simulation of the current waste market. For each endogenous variable (see Table 6.23 for a description of these variables) the tables indicate whether a specific sensitivity scenario leads to results that differ from the basic scenario outcome (the results of the basic scenario are given in the first row).

Table 6.23: Description endogenous variables

Ctot	Total costs treatment Dutch waste
Chou	Total costs treatment Dutch household waste
Cfir	Total costs treatment Dutch firm waste
GProf	Gross profits (before interest and depreciation) existing Dutch WIP's
PI_{LL}	Dutch price incineration very low caloric waste (result of separation low caloric waste)
PI_{L}	Dutch price incineration low caloric waste
PI_{LM}	Dutch price incineration low caloric waste (result of separation mid caloric waste)
PI_{M}	Dutch price incineration mid caloric waste
PI_{MV}	Dutch price incineration dirty mid caloric waste
PI_{HL}	Dutch price incineration high caloric waste (result of separation low caloric waste)
PI_{HM}	Dutch price incineration high caloric waste (result of separation mid caloric waste)
PI_{H}	Dutch price incineration high caloric waste
PLan	Dutch price landfilling
Lan_{NL}	Amount of waste landfilled in Netherlands
WIP_{NL}	Amount of waste incinerated in existing Dutch WIP's
CoC_{NL}	Amount of waste incinerated in Dutch coal fired electricity plants and cement kilns
$NewL_{NL}$	Amount of waste incinerated in new Dutch low caloric WIP's
$NewM_{NL}$	Amount of waste incinerated in new Dutch mid caloric WIP's
$NewH_{NL}$	Amount of waste incinerated in new Dutch high caloric WIP's
Lan_{For}	Amount of Dutch waste landfilled in other countries
WIP_{For}	Amount of Dutch waste incinerated in WIP's in other countries
CoC_{For}	Dutch waste incinerated in foreign coal fired electricity plants and cement kilns
Imp_{NL}	Amount of foreign waste imported from other countries
TSM	Total amount of separated mid caloric waste
TSL	Total amount of separated low caloric waste

The effects of the sensitivity scenarios are estimated on the basis of data generated by simulating all second-order sensitivity scenarios. OLS-estimations with the endogenous variables as dependent and dummies for the sensitivity scenarios (and a constant for the average effect) as independent variables give the estimated effects of the different sensitivity scenarios. Only effects that are significant on 90% confidence are presented. Thus, when no figure is presented the effect of a sensitivity scenario is expected to be zero. Table 6.24 presents the different sensitivity scenarios. For example, in sensitivity scenario 5-Lower the costs of landfilling in France are 10 euro per ton lower, while in sensitivity scenario 5-Upper the costs of landfilling in France are 10 euro per ton higher.

Table 6.24: Overview sensitivity analysis

Nr.	Region	Variable	Lower	Upper
1	Flanders	Costs landfill (euro per ton)	-10	10
2	Wallonia		-10	10
3	Denmark		-10	10
4	Germany		-10	10
5	France		-10	10
6	Italy		-10	10
7	NL		-10	10
8	UK		-10	10
9	Belgium	Costs existing Waste Incineration Plants (euro per ton)	-10	10
10	Denmark		-10	10
11	Germany		-10	10
12	France		-10	10
13	Italy		-10	10
14	NL		-10	10
15	UK		-10	10
16	Miscellaneous	Levelplaying field environment	na	Model
17	All	High costs new WIP's	na	Model
18	All	Low costs new WIP's	Model	na
19	All	Costs cement and coal (euro per ton)	-10	20
20	All	Costs separation (euro per ton)	-10	20
21	All	Transportcosts	Model	na
22	All	Occupation rate existing Waste Incineration Plants	-2.5%	2.5%
23	All	Contract percentage existing Waste Incineration Plants	-7%	7%
24	All	Thermic capacity	-10%	10%
25	All	Heating value waste	Lower	Higher
26	All	Capacity cement and coal	-50%	50%
27	Belgium	Supply low caloric waste	-20%	20%
28	Denmark		-20%	20%
29	Germany		-20%	20%
30	France		-20%	20%
31	Italy		-20%	20%
32	NL		-5%	5%
33	UK		-20%	20%
34	Belgium	Supply mid and high caloric waste	-25%	25%
35	Denmark		-25%	25%
36	Germany		-25%	25%
37	France		-25%	25%
38	Italy		-25%	25%
39	NL		-25%	25%
40	UK		-25%	25%
41	All	Separation possibility low caloric waste	-5%	5%

Tables 6.25 till 6.28 present the effects of the sensitivity scenarios. We illustrate the tables for the endogenous variable total costs (Ctot). Table 6.28 shows that in the long run total costs for the current situation are 415 million euro in the basic scenario.

The following sensitivity analysis lead to a change in total costs: 17-Upper (+53 million euro), 18-Lower (-44 million euro), 19-Upper (+66 million euro), 19-Lower (-32 million euro), 20-Upper (+31 million euro), 20-Lower (-17 million euro) and 26-Lower (+25 million euro). Other sensitivity analysis do not have any effect on total costs.

Table 6.25: Sensitivity analysis current short term 1

Nr.	Sc.	Ctot	Chou	Cfir	GProf	Pl_{LL}	Pl_{L}	Pl_{LM}	Pl_{M}	Pl_{MV}	Pl_{HL}	Pl_{HM}	Pl_{H}	PLan	Lan_{NL}
0	Bas	440	153	288	101	109	113	n.a.	n.a.	116	28	29	29	116	966
1	Low														
2	Up														
2	Low														
3	Up														
3	Low														
4	Up														
4	Low														
5	Up	20		20		-3					6	6	6		22
5	Low											-1	-1		
6	Up														
6	Low														
7	Up	23	13	10	13	11	8			10				10	
7	Low	-22	-12	-10	-12	-23	-8			-10	-6			-10	178
8	Up														
8	Low														
9	Up														
9	Low														
10	Up														
10	Low														
11	Up														
11	Low														
12	Up														
12	Low														
13	Up														
13	Low														
14	Up				-11										23
14	Low				12										
15	Up														
15	Low														
16	Up														
19	Up	64		64		-12	1				13	19	19		175
19	Low											0	0		
20	Up	31		31		-109					-28				195
20	Low	-16		-15	9	13	-113			3					-61
21	Low	22		21		-3					6	6	6		22
22	Up				3										-36
22	Low				-3										34
23	Up		2		-29		3								409
23	Low	-16	-16		13	-23	-12				-5				-175
24	Up				24	-2	3				-1				-165
24	Low				-29		3			-116					251
25	Up				-32					-116					324
25	Low				26		2								-163
26	Up											0	0		
26	Low	23		22		-3					3	7	7		24
27	Up														
27	Low														
28	Up														
28	Low														
29	Up														
29	Low														
30	Up														
30	Low														
31	Up														
31	Low														
32	Up		2		2		3								236
32	Low	-8	-8		-7	-8	-7								-135
33	Up														
33	Low														
34	Up														
34	Low														
35	Up														
35	Low														
36	Up														
36	Low														
37	Up											0			
37	Low														
38	Up														
38	Low														
39	Up										-2				256
39	Low														-256
40	Up														
40	Low														
41	Up				5	5	-8			3					-103
41	Low					-109					-28				187

Table 6.26: Sensitivity analysis current short term 2

Nr.	Sc.	WIP_{NL}	CoC_{NL}	$NewL_{NL}$	$NewM_{NL}$	$NewH_{NL}$	Lan_{For}	WIP_{For}	CoC_{For}	Imp_{NL}	TSL	TSM
0	Bas	4749	2113	n.a.	n.a.	n.a.	767	0	1329	40	783	1534
1	Low											
2	Up											
2	Low											
3	Up											
3	Low											
4	Up											
4	Low											
5	Up									12	-84	5
5	Low											
6	Up											
6	Low											
7	Up									2	35	
7	Low	-77							-100	-37	-671	
8	Up											
8	Low											
9	Up											
9	Low											
10	Up											
10	Low											
11	Up											
11	Low							10				
12	Up											
12	Low											
13	Up											
13	Low											
14	Up									-9	-82	
14	Low											
15	Up											
15	Low											
16	Up											
19	Up	-76						320	-419	-29	-660	
19	Low									20	20	
20	Up	-84							-111	-39	-733	
20	Low								55	30	369	
21	Low								-16	20	-85	
22	Up	132										
22	Low	-131										
23	Up								-33	-18	-229	8
23	Low	-85							-95	-35	-626	
24	Up	212							-49	-28	-312	9
24	Low	-290							41	20	257	-8
25	Up	-343							21	7	129	
25	Low	220							-58	-33	-371	10
26	Up		1049						-1046	-29	3	
26	Low		-1057						1043	33	-93	
27	Up											
27	Low											
28	Up											
28	Low											
29	Up											
29	Low											
30	Up									6	6	
30	Low											
31	Up											
31	Low											
32	Up											
32	Low								-58	-25	-389	
33	Up											
33	Low											
34	Up									13	13	7
34	Low											
35	Up											
35	Low											
36	Up											
36	Low											
37	Up									15	15	
37	Low											
38	Up									-7	-7	
38	Low											
39	Up						192		831	54	-2	384
39	Low						-192		-823	-30		-384
40	Up									-13	-13	
40	Low											
41	Up								67	42	151	
41	Low	-80							-107	-38	-699	

Table 6.27: Sensitivity analysis current long term 1

Nr.	Sc.	Ctot	Chou	Cfir	GProf	PI_{LL}	PI_{L}	PI_{LM}	PI_{M}	PI_{MV}	PI_{HL}	PI_{HM}	PI_{H}	PLan	Lan_{NL}
0	Bas	415	168	247	79	n.a.	77	n.a.	n.a.	77	n.a.	29	29	n.a.	0
1	Low														
2	Up														
2	Low														
3	Up														
4	Up														
4	Low														
5	Up														
5	Low														
6	Up														
6	Low														
7	Up														
7	Low														
8	Up														
8	Low														
9	Up														
9	Low														
10	Up														
10	Low														
11	Up														
11	Low														
12	Up														
12	Low														
13	Up														
13	Low														
14	Up				-15	1					0				
14	Low				15										
15	Up														
15	Low														
16	Up														
17	Up	53	35	17	26	1	16			17	0				
18	Low	-44	-30	-15	-21	-1	-14		1	-14	0	0			
19	Up	66		66					1			20	20		
19	Low	-32		-32								-10	-10		
20	Up	31		31					3			0			
20	Low	-17	-2	-15		73	-3				28				
21	Low														
22	Up				2										
22	Low				-2										
23	Up				-17										
23	Low		-2		14		-1								
24	Up				15					-2					
24	Low				-18										
25	Up				-20	1					0				
25	Low			-4	14				1	-4					
26	Up											-1			
26	Low	25		25								8	7		
27	Up														
27	Low														
28	Up														
28	Low														
29	Up														
29	Low														
30	Up														
30	Low														
31	Up														
31	Low														
32	Up														
32	Low		-2		-2		-1								
33	Up														
33	Low														
34	Up														
34	Low														
35	Up														
35	Low														
36	Up														
36	Low														
37	Up														
37	Low														
38	Up														
38	Low														
39	Up														
39	Low											0			
40	Up														
40	Low														
41	Up					3	-1				1				
41	Low					-1					0				

Table 6.28: Sensitivity analysis current long term 2

Nr.	Sc.	WIP_{NL}	CoC_{NL}	$NewL_{NL}$	$NewM_{NL}$	$NewH_{NL}$	Lan_{For}	WIP_{For}	CoC_{For}	Imp_{NL}	TSL	TSM
0	Bas	5027	2113	789	807		770		1225			1541
1	Low											
2	Up											
2	Low											
3	Up											
4	Up											
4	Low											
5	Up											
5	Low											
6	Up											
6	Low											
7	Up											
7	Low											
8	Up											
8	Low											
9	Up											
9	Low											
10	Up											
10	Low											
11	Up											
11	Low											
12	Up											
12	Low											
13	Up											
13	Low											
14	Up			32	-23							
14	Low											
15	Up											
15	Low											
16	Up											
17	Up									3	24	
18	Low			64	-22		-10					-20
19	Up				8			315	-320			-12
19	Low											
20	Up				29		-14		-14			-29
20	Low	88		-202					120	76	797	
21	Low							4				
22	Up	130		-28								
22	Low	-130		29								
23	Up			346								
23	Low			-347								
24	Up	250		274	-523							
24	Low	-306		87	218							
25	Up	-369		147	221							
25	Low	264		422	-685							
26	Up		1048						-1046			
26	Low		-1056						1057			
27	Up											
27	Low											
28	Up											
28	Low											
29	Up											
29	Low											
30	Up											
30	Low											
31	Up											
31	Low											
32	Up			280								
32	Low			-279								
33	Up											
33	Low											
34	Up											4
34	Low											
35	Up											
35	Low											
36	Up											
36	Low											
37	Up											
37	Low											
38	Up											
38	Low											
39	Up				256		193		835			385
39	Low				-255		-192		-825			-385
40	Up											8
40	Low											
41	Up			-9						5	39	
41	Low											

Appendix D Detailed results regulation scenarios

In this appendix detailed results are given for the different regulation scenarios. Table 6.29 defines the scenarios.

Table 6.29: Regulation scenarios

Nr.	Region	Variable	Level
42	Netherlands	Landfill tax	36 euro
43	Netherlands	Landfill tax	0 euro
44	Netherlands	Landfill tax	0 euro
44	Netherlands	Subsidy incineration	18 euro
45	Netherlands	Landfill tax	0 euro
45	Netherlands	Subsidy incineration	36 euro
46	Netherlands	Landfill tax	0 euro
46	Netherlands	Subsidy incineration	52 euro
47	All	Landfill tax	Difference costs of a new WIP and landfilling

Table 6.30: Simulation results short term market with closed borders

	Short term		Long term	
	Closed	42	Closed	42
Selfsufficiency (% of total waste)	100	100	100	100
Landfilling (% of total waste)	32	32	0	0
Total costs (million euro per year)				
- excluding quantity effect	903	777	889	889
- including quantity effect	903	777	955	955
Tax income (million euro per year)	237	114	0	0
Total Dutch waste (Mton)	9.9	9.9	9.9	9.9
- landfilled in Netherlands	3.2	3.2	0	0
- incinerated in Netherlands	4.7	4.7	5	5
- reused in Netherlands	2.1	2.1	2.1	2.1
- landfilled in other countries	0	0	0	0
- incinerated in other countries	0	0	0	0
- reused in other countries	0	0	0	0
Imported from other countries	0	0	0	0
New incineration capacity (Mton)				
- low caloric	0	0	0.8	0.8
- mid caloric	0	0	2.4	2.4
- high caloric	0	0	0.4	0.4
Price incineration (euro per ton)				
- low caloric waste	115	76	76	76
- Dirty mid caloric waste	116	77	77	77
- mid caloric waste	115	76	77	77
- high caloric waste	116	77	77	77
Price landfill (euro per ton)	116	77	n.a.	n.a.

Table 6.31: Simulation results short term current market

	Current	42	43	44	45	46	47
Selfsufficiency (% of waste)	79	80	96	96	96	96	96
Landfilling (% of waste)	17	20	29	28	27	27	27
Total costs (mln euro/year)							
- excluding quantity effect	738	679	630	711	796	871	751
- including quantity effect	738	679	630	711	796	871	751
Tax income (mln euro/year)	73	42	0	0	0	0	98
Total Dutch waste (Mton)	9.9	9.9	9.9	9.9	9.9	9.9	9.9
- landfilled in Netherlands	1.0	1.2	2.9	2.8	2.7	2.7	2.7
- incinerated in Netherlands	4.8	4.7	4.5	4.6	4.7	4.7	4.7
- reused in Netherlands	2.1	2.1	2.1	2.1	2.1	2.1	2.1
- landfilled in other countries	0.8	0.8	0	0	0	0	0
- incinerated in other countries	0	0	0	0	0	0	0
- reused in other countries	1.3	1.2	0.4	0.4	0.4	0.4	0.4
Imported from other countries	0	0	0	0	0	0	0
New incin. capacity (Mton)							
- low caloric							
- mid caloric							
- high caloric							
Price incineration (euro/ton)							
- low caloric waste	113	75	41	39	39	39	75
- Dirty mid caloric waste	116	77	41	41	41	41	77
- mid caloric waste	71	52	35	34	34	34	68
- high caloric waste	29	29	29	29	29	29	61
Price landfill (euro/ton)	116	77	41	41	41	41	77

Table 6.32: Simulation results long term current market

	Current	42	43	44	45	46	47
Selfsufficiency (% of total waste)	82	82	96	96	96	100	89
Landfilling (% of total waste)	7	7	34	30	0	0	0
Total costs (million euro per year)							
- excluding quantity effect	720	720	629	717	922	1001	774
- including quantity effect	785	785	665	754	958	1030	840
Tax income (million euro per year)	0	0	0	0	0	0	0
Total Dutch waste (Mton)	10.7	10.7	10.7	10.7	10.7	10.7	10.7
- landfilled in Netherlands	0	0	3.7	3.2	0	0	0
- incinerated in Netherlands	6.6	6.6	4.5	5.0	8.1	7.9	7.4
- reused in Netherlands	2.1	2.1	2.1	2.1	2.1	2.1	2.1
- landfilled in other countries	0.8	0.8	0	0	0	0	0
- incinerated in other countries	0	0	0	0	0	0	0
- reused in other countries	1.2	1.2	0.5	0.5	0.5	0	1.2
Imported from other countries	0	0	0	0	0	0	0
New incineration capacity (Mton)							
- low caloric	0.8	0.8	0	0	0.8	0.8	1.6
- mid caloric	0.8	0.8	0	0	2.3	2.4	0.8
- high caloric	0	0	0	0	0	0.7	0
Price incineration (euro per ton)							
- low caloric waste	77	76	41	40	40	24	77
- Dirty mid caloric waste	77	77	41	41	41	25	77
- mid caloric waste	53	53	35	34	35	25	55
- high caloric waste	29	29	29	29	29	25	34
Price landfill (euro per ton)	n.a.	n.a.	41	41	n.a.	n.a.	n.a.

Table 6.33: Simulation results short term market with open borders

	Open	42	43	44	45	46	47
Selfsufficiency (% of total waste)	63	63	63	66	68	68	96
Landfilling (% of total waste)	37	37	37	33	31	30	27
Total costs (million euro per year)							
- excluding quantity effect	570	570	570	653	739	814	751
- including quantity effect	570	570	570	653	739	814	751
Tax income (million euro per year)	0	0	0	0	0	0	98
Total Dutch waste (Mton)	9.9	9.9	9.9	9.9	9.9	9.9	9.9
- landfilled in Netherlands	0	0	0	0	0	0	2.7
- incinerated in Netherlands	4.1	4.1	4.1	4.4	4.6	4.7	4.7
- reused in Netherlands	2.1	2.1	2.1	2.1	2.1	2.1	2.1
- landfilled in other countries	3.7	3.7	3.7	3.3	3.0	3.0	0
- incinerated in other countries	0	0	0	0	0	0	0
- reused in other countries	0	0	0	0.1	0.1	0.1	0.4
Imported from other countries	0	0	0	0	0	0	0
New incineration capacity (Mton)							
- low caloric							
- mid caloric							
- high caloric							
Price incineration (euro per ton)							
- low caloric waste	n.a.	n.a.	n.a.	27	27	26	76
- Dirty mid caloric waste	31	31	31	32	27	26	77
- mid caloric waste	31	12	12	26	27	26	68
- high caloric waste	24	25	25	26	27	27	61
Price landfill (euro per ton)	17	17	17	17	17	17	77

Table 6.34: Simulation results long term market with open borders

	Open	42	43	44	45	46	47
Selfsufficiency (% of total waste)	58	58	58	62	67	96	89
Landfilling (% of total waste)	42	42	42	37	32	4	0
Total costs (million euro per year)							
- excluding quantity effect	570	573	573	655	750	964	775
- including quantity effect	593	596	596	681	780	1003	840
Tax income (million euro per year)	0	0	0	0	0	0	0
Total Dutch waste (Mton)	10.7	10.7	10.7	10.7	10.7	10.7	10.7
- landfilled in Netherlands	0	0	0	0	0	0	0
- incinerated in Netherlands	4.1	4.1	4.1	4.5	5.0	8.1	7.4
- reused in Netherlands	2.1	2.1	2.1	2.1	2.1	2.1	2.1
- landfilled in other countries	4.5	4.5	4.5	4	3.5	0.5	0
- incinerated in other countries	0	0	0	0	0	0	0
- reused in other countries	0	0	0	0.1	0.1	0	1.2
Imported from other countries	0	0	0	0	0	0	0
New incineration capacity (Mton)							
- low caloric	0	0	0	0	0	0.7	1.6
- mid caloric	0	0	0	0	0	2.1	0.8
- high caloric	0	0	0	0	0	0.3	0
Price incineration (euro per ton)							
- low caloric waste	n.a.	n.a.	n.a.	27	27	24	77
- Dirty mid caloric waste	31	31	31	31	26	25	77
- mid caloric waste	31	12	12	26	27	24	56
- high caloric waste	24	25	25	26	27	25	34
Price landfill (euro per ton)	17	17	17	17	17	17	n.a.

Chapter 7

Conclusions and summary

7.1 Introduction

Governmental regulation plays an important role in the waste collection and treatment market. The potential effects on health and environment when no regulation exists are such that regulation seems to be necessary. Indeed, current waste regulation differs significant with instruments used thirty years ago. First, to prevent health and environmental problems resulting from piling waste in the streets, municipalities have now an obligation to supply a proper collection infrastructure. However, they are free to decide in what way they organize this task. Second, emissions limits apply to landfills and waste incineration plants to fight the environmental and health consequences of waste disposal. Before emission limits were given by the national government, waste incineration resulted, for example, in undesirable high concentrations of cancer causing dioxines and furanes while leakage of chemical substances out of landfills was also a major health risk. Third, regulation is now present to steer waste streams in the direction of specific treatment options to stimulate the use of options with the lowest level of unwanted side-effects. This is based on an explicit ordering of options with respect to the supposed effects on the environment. Finally, the Dutch government focuses on the use of national waste treatment firms as export of waste for disposal was seen as an undesirable option as this would lead to the export of environmental and health problems.

As a result of regulation the costs of waste treatment rose at a high speed during the last thirty years. As long as this rise in costs is necessary to prevent unwanted side-effects of collecting and treating waste, it is just a historical fact. However, from an economic point of view at least circumstantial evidence exist that other types of regulation might decrease total costs. In this thesis the central question is whether changes in waste market regulation can indeed decrease total social costs. Total social costs in this thesis is equal to the sum of private and external costs. Private costs are equal to the necessary production costs. Social costs can be higher than private costs when external costs exist. External costs are costs related to production or consumption, but which are not valuated in the market. For example, the price paid for waste incineration is based on the private costs of capital, labor and other inputs, but not on the amount of emitted carbon dioxide or dioxins. As these emissions impose in fact a cost burden on society a proper economic analysis is not only based on private costs, but also on external costs. Mathematically total social cost is equal to:

$$TSC = W\left(C_C^P + C_C^E + C_T^P + C_T^E\right), \tag{7.1.1}$$

with TSC total social costs (the sum of private and external costs), W the total quantity of waste collected, C_C^P the private cost of waste collection (excluding treatment costs), C_C^E the external cost of waste collection, C_T^P the private costs of waste treatment and C_T^E the external costs of waste treatment.

Each of the chapters of this thesis analyses a part of equation 7.1.1 for the Netherlands. Four specific questions are dealt with. The **first two questions** relate to the waste collection market:

- Is it possible to decrease the quantity of waste collected (W) by making use of unit-based pricing systems for household waste (chapter 2)?
- Is it possible, given the amount of collected waste (W), to decrease total private collection costs (C_C^P) by contracting out the waste collection of municipalities (chapter 3) and, if this is the case, which factors may explain the relative low penetration of contracting out (chapter 4)?

Whether the (obligatory) collection of household waste is done by municipalities themselves or contracted out to (public or private) firms is an issue left to the municipalities. Municipalities have also the right to decide the way households pay for waste collection. Dutch municipalities show important differences in choices made. At first sight, these differences may be in conflict with empirical evidence. International experiences reported in the economic literature suggest some evidence that municipalities that contract out waste collection and use a payment scheme related to the amount of waste collected have lower total collection costs than municipalities that collect the waste themselves and use a fixed fee to cover collection costs. The use of contracting out is meant as an efficiency improving instrument by introducing competition. According to the literature unit-based pricing leads to lower levels of waste and thus collection costs as waste suppliers now have an incentive to reduce waste. However, international experience might be misleading due to differences in local circumstances. Furthermore, the international literature has important caveats which make a proper judgement difficult as long as no specific Dutch evidence is present using the appropriate methodology. Chapter 2 and 3 are meant to improve the empirical basis for such a judgement. As the results are that both methods seems to be effective indeed, the question remains why not all Dutch municipalities use these methods. While for unit-based pricing the reason could be that it is only recently introduced which could imply a difference between avant-gardists and hesitaters (see topics for future research), contracting-out is a proven method that is used for a long time. However, the economic theory presents some reasons that might explain this behaviour. For example, it is interesting to analyse whether political parties on the left might be more hesitant to be dependent on private waste collection firms than right parties or to see whether vested interests of workers at municipal collection services influence the penetration of contracting out. Chapter 4 presents an empirical test of these theories.

The **last two question** relate to the waste treatment market:

- Given the amount of collected waste, is it possible to decrease total treatment costs ($C_T^P + C_T^E$) by changing the preference for incineration above landfilling (chapter 5)?
- Given the amount of collected waste (W) and the preferred treatment options, is it possible to decrease total private treatment costs (C_T^P) by increasing international competition (chapter 6)?

To ensure a proper treatment of waste, the government has chosen for a preference of incineration above landfilling. This preference for incineration was at the time motivated by environmental concerns regarding landfilling. However, no explicit comparison was made between the two options on the basis of both external and private costs. As long as the decrease in external costs is offset by the increase in private costs, incineration is indeed the best option from an economic point of view. However, the far higher private costs of incineration at least questions whether this is the outcome when a proper total social costs analysis is the basis for a choice between landfilling and incineration. We present such an analysis in chapter 5.

The type of regulation chosen for the waste treatment market resulted in monopolistic behaviour. Competition between landfilling and incineration firms disappeared due to the high landfill tax. This tax was introduced to steer waste streams in the direction of incineration, which was seen as a better disposal option than landfilling. Furthermore, competition between waste incineration plants was low due to the small geographic market. Not only export to foreign waste incineration plants was prohibited, during some years the national market was even divided in different parts. An interesting question is whether these market restrictions are necessary to fulfil the goals the government has set. Moreover, from an economic point of view the question is whether the chosen form of regulation minimizes total treatment costs. Chapter 6 presents therefore a simulation model that makes it possible to analyse these issues.

Finally, two remarks are necessary to put this thesis in perspective. First, the four central questions cover the most important determinants of total social waste collection and treatment costs that can be influenced by changing regulation. This means that not all aspects of the waste market and social costs are dealt with. In general, we focus on the determinants of total social costs for which it can be expected that changes in national regulation will change these costs significantly. The prior is that this may be the case when regulation exists that hinders the minimization of social costs. In fact, the chosen most important determinants of social costs that can be changed by regulation are based (except for the preference order between landfilling and incineration) on the observation that the markets for prevention, recycling and collection of firm waste are not significantly hindered by national regulation. For example, the role of regulating prevention and recycling is not dealt with as the market for prevention and recycling (according to EU-regulation) is already international oriented. Second, the statement that the four questions cover the most important determinants of total social waste collection and treatment costs does not mean that the answers given in this thesis are comprehensive. The list of open research questions presented in the last paragraph indicates rather that it is a building stone in the edifice of empirical economic waste research.

7.2 Main findings

In chapter 2 the central question is whether a unit-based pricing scheme reduces total waste collected in the Netherlands (W). The idea is that the introduction of a price per unit waste collected gives citizens an incentive to reduce the total amount of waste and to substitute between priced and unpriced waste collection options. This last substitution is possible as Dutch municipalities are obliged to supply a free collection infrastructure for glass, paper and textile. The effects of weight-based, bag-based, frequency-based and volume-based pricing of household waste collection are estimated using a panel data-set for (nearly all) Dutch municipalities. The quantity of waste collected in municipalities with these unit-based pricing systems is compared with the quantity of waste collected in municipalities with a fixed fee. In general, unit-based pricing is shown to be effective in reducing unsorted and compostable waste and increasing recyclable waste. Part of this effect results from a higher level of environmental activism in municipalities with a unit-based pricing system. This indicates that the level of waste collected is already lower in municipalities that introduce a unit-based pricing system as their citizens are more environmentally friendly oriented. Thus, overestimation is present for municipalities with a high level of environmental activism, but not for municipalities with a low level as unit-based pricing might internalize the lack of environmental activism. The estimation results indicate indeed that the quantity of waste is lower in municipalities that introduce unit-based pricing in later years. Thus, an environmental activism effect is present.

The results show that important differences exist between the different unit-based pricing systems. The bag-based and weight-based systems perform equally and far better than the frequency-based and volume-based systems. As administrative costs are significantly lower for the bag-based system than for the weight-based system, the bag-based system is preferred from a cost minimizing point of view.

Although illegal dumping is said to be responsible for at least a part of this effect, leading to external costs of waste collection (C_C^E), we found no empirical evidence that the decrease in waste results (partly) from substitution to other municipalities. Pricing has no effect on the amounts of waste collected in surrounding municipalities without a unit-based pricing system (waste tourism). However, other forms of illegal dumping could not be tested due to lacking data.

When all municipalities introduce a bag-based pricing system a total social cost saving of about 400 million euro, 32% of current costs, is possible (see Appendix A). Thus, unit-based pricing has a major influence on total costs.

In chapter 3 we analyse the possible cost savings of contracting out refuse collection (C_C^P) in the Netherlands. Our findings indicate that similar to results in other countries cost savings of approximately 15% apply to the Netherlands. Moreover, we show that internal municipal waste collection units and external refuse collection firms use different production technologies. Different cost functions have to be estimated for the subsamples. Furthermore, we found that, though significant cost savings exist when waste collection is contracted out, households have not experienced these cost savings on a one to one basis. Private refuse collection firms had to pay VAT while public entities were exempted. Thus, the fiscal system hindered a more pronounced role for private refuse collection firms. This pleas for the VAT compensation fund that is in use for public collectors since January 2003.

When all Dutch municipalities that can gain from contracting out (private outside firms instead of inside collection) introduce this instrument total social cost will diminish with an estimated value of about 30 million euro (see Appendix A). That these savings are considerable lower than for unit-based pricing results not only from the higher incidence of contracting out, but also from the fact that contracting out reduces only collection costs while unit-based pricing saves also waste treatment and external costs.

The cost effectiveness of contracting out suggests that municipalities should contract out waste collection from an economic point of view. However, a majority of Dutch municipalities provides for waste collection services themselves. In chapter 4 we seek an explanation for the reservations of local authorities towards contracting out. Based on theoretical insights we model the choice between private, public, in-house, and out-house refuse collection. The models are estimated using a database comprising nearly all Dutch municipalities. We find evidence that the number of inhabitants, the transfer by central government, and interest group arguments are important explanations. Interestingly, ideology seems to play a minor role. Compared to earlier studies we estimate more general models. Although the same qualitative results are found for parametric and semiparametric models, we find strong statistical evidence that a parametric specification is far too inflexible. Differences between the parametric and the semiparametric marginal effects are substantial. Thus, more attention is needed for the implications of model specification.

Chapter 5 uses private and external cost data for the Netherlands to evaluate the social cost of two final waste disposal methods, landfilling versus incineration. The data only provide some support for the widespread policy preference for incineration over landfilling if the analysis is restricted to environmental costs. Private costs, however, are so much higher for incineration, that landfilling is the social cost minimizing option at the margin even in a densely populated country such as the Netherlands. Implications for waste policy are not only that current regulation instruments give the wrong incentives in the direction of incineration, but also that proper treatment of and energy recovery from landfills seems to be an important target for waste policy. If the preference order is indeed changed, total social costs will diminish with about 115 million euro according to our analysis. This figure is based on the assumption that only household waste that is not yet contracted with long term contracts by waste incineration plants goes to the cheaper landfills. This means that the decrease in total social costs is larger (about 220 million euro) when these contracts expire. (see Appendix A).

Chapter 6 shows that the objectives chosen by the Dutch government (selfsufficiency for waste disposal and a preference for incineration above landfilling) can be reached independent of market sizes. The chosen objectives are reachable when the landfill tax is abolished while at the same time a subsidy is given to waste incineration plants (WIP's). In this case even with open borders a selfsufficiency percentage of 100% and a landfilling percentage of 0% is possible. While this regulation package guarantees the fulfillment of national waste policy objectives, harmonization of the landfill tax on a EU-scale might be a better option. When all EU-members introduce a landfill tax equal to the difference in costs between landfilling and a new WIP, landfilling will be zero in the long term, while a selfsufficiency percentage is possible of nearly 90%. As subsidies are rather expensive, this option saves on costs. However, international coordination of a landfill tax is not easy to reach as most EU-members are not keen to introduce such a tax. The best policy option might be to choose for the national instrument of a WIP-subsidy (and a landfill tax of zero), while in the meantime international coordination of a landfill tax is searched for.

However, the chosen policy goals are not in accordance with economic principles. From an economic perspective internalization of external cost is the basis of regulation. When taxes for all waste treatment option are set according to external costs, a selfsufficiency percentage of only 61% and a landfilling percentage of only 38% is optimal, while these percentages decrease and increase respectively when existing WIP's are closed. This is caused by the lower social costs of landfilling in other countries for most waste streams compared with the social costs of other options or with the social cost of landfilling in the Netherlands. Compared with the best regulation package when the current Dutch waste objectives are fulfilled, a private cost saving is possible of nearly 400 million euro per year. Savings are less when the current policy objectives are less stringent interpreted. It could be argued that for the current market (allowed export for reuse) a selfsufficiency percentage beneath 100% and a landfilling percentage above 0% is acceptable as long as foreign treatment is limited to reuse and landfilling is necessary to facilitate this reuse (in other words as only the residue of separation is landfilled). However, even in this case the savings on total social costs are still about 80 million euro when goals are set according to economic principles (see Appendix A). In all cases are open national borders preferred from a social cost perspective.

The potential cost decrease when a specific regulation instrument is changed is not always independent of changes in other instruments. For example, when a bag-based pricing system is introduced in municipalities that have currently a flat rate system, the total quantity of waste will diminish. This means that the cost reduction of opening national borders for waste treatment is smaller. If all instruments are implemented according to the goal of minimizing social cost (a bag-based pricing system in all municipalities, private outside collection in municipalities that have currently inside collection, open national borders and taxes on treatment options in accordance with external costs) social costs will decrease from about 1250 million euro to 780 million euro (see Appendix A). This means that changes in regulation can result in a total social cost reduction of about 470 million euro per year (38% of current costs).

7.3 Future research

This thesis analyses the most important parts of the waste chain for which could be expected that social cost reduction by changing regulation instruments is possible. It also sheds light on future research projects that could increase the insight in the influence of regulation instruments on total social costs of the waste market.

First, the most important disadvantage of unit-based pricing systems is the effect on illegal dumping (see chapter 2). However, not much empirical evidence is present whether this effect is significant. This thesis sheds only light on the effect on neighbouring municipalities. Although this effect seems to be negligible in the Netherlands, a comprehensive analysis of illegal dumping is missing. As illegal dumping is the most important argument used by opponents, future research could invest in the quality of this evidence. If illegal dumping shows to be a major side-effect, more attention should be given to instruments that introduce incentives to minimize illegal dumping. Examples are effective monitoring and fining systems. If illegal dumping is not present, the penetration process of unit-based pricing could be analysed. Given the large effects on waste quantities the fact that many municipalities do not use this instrument asks for a sound explanation. Maybe the fact that unit-based pricing is only recently introduced plays an important role. The uncertainty about the true effects and the expected occurrence of unwanted side-effects might imply that only a small number of avant-gardists are willing to try this instrument while a large number of hesitaters wait with the introduction of unit-based pricing till more evidence is present.

Second, chapter 3 makes clear that outside and inside collection firms use a different production technology. To increase the validity of this conclusion and to shed more light on the consequence of this conclusion for the estimation of efficiency effects when institutional forms are changed, future research based on a more extensive dataset could result in more precise predictions of the influence of contracting out on collection costs. Furthermore, it makes pooled estimations possible which can generate insight in the persistence of the efficiency effect. This can be important as collusion or lock-in effects might hamper efficiency in the long term.

Third, chapter 4 shows that not only cost considerations determine the mode of waste collection. As a large number of municipalities chooses another institutional form than suggested on the basis of a social cost analysis, future research could generate a better understanding of municipal behaviour if other arguments than direct cost decreases are taken into mind. For example, municipalities might not contract out waste collection to ensure grip on the future coarse of waste collection. The fear to loose control if an irreversible instrument is chosen can add an indirect cost to contracting out. If these costs are included contracting out might loose (some) attractiveness. The same principle might influence the decision on the size of the geographical market for waste disposal as a decision to allow exports is probably irreversible from a legal point of view (chapter 6).

Fourth, the analysis in chapter 5 includes future costs only in a coarse way. The difference between static and dynamic consequences can influence the decision between landfilling and incineration as the last option concentrates environmental effects to the moment of disposal, while landfilling generates emissions for a long time period. The assumption of constant marginal cost behind the presented estimates is rather strong, and implies that landfilling is the preferred solution for all disposable waste now and in the future. Therefore, future research should shed more light on the dynamic characteristics of landfilling vis-à-vis incineration.

Fifth, further empirical research is necessary to investigate whether the results presented in chapter 5 hold for all types of waste. The analysis in this chapter is focussed on so-called low caloric waste (waste from households). First calculations indicate that also for waste with a higher caloric value (part of the waste from firms) landfilling is preferred from a social cost perspective above incineration in a waste incineration plant although the environmental costs are far lower (see chapter 6). However, using high caloric waste in a cement kiln might be the best option. This stems primarily from the lower environmental costs as no chemical waste is produced. Future research might compare more treatment options for different waste streams on a social cost basis.

Sixth, the analysis presented in chapter 6 does not include the relations between prevention, reuse and disposal. When market circumstances change in the waste disposal market, prices and quantities might change in the prevention and reuse market. For example, when the tariffs for waste incineration decrease, incentives for prevention and reuse decrease at the same time possibly leading to a substitution in the direction of more disposal. In chapter 6 it is (implicitly and in accordance with the current Dutch policy) assumed that instruments can be found that compensate for these effects. However, as the changes in waste disposal tariffs are significant, more emphasis on the linkages between the waste markets might lead to interesting results.

Seventh, the model used in chapter 6 assumes free competition. Although no explicit information is present to assume a significant role for collusion, the potential impact of this type of market failure makes it interesting to analyse whether collusion exists and what the implications are for the market outcomes.

Eighth, the analysis presented in chapter 6 is dominated by Dutch policy questions. Given the international developments, the model could be used to generalize conclusions when more detailed information is available from other EU-countries.

Appendix A Effects on total costs

Introduction

In this appendix we present an estimation of the effects on total social costs of the instruments analysed in the preceding chapters. The reason that these effects are not calculated in the preceding chapters is that for the calculation of these effects often information is necessary for the whole waste chain.

In chapter 1 we defined total social costs as:

$$TSC = W\left(C_C^P + C_C^E + C_T^P + C_T^E\right), \tag{7.3.2}$$

with TSC total social costs (the sum of private and external costs), W the total quantity of waste collected, C_C^P the private cost of waste collection (excluding treatment costs), C_C^E the external cost of waste collection, C_T^P the private costs of waste treatment and C_T^E the external costs of waste treatment. Thus, the effects on total social costs of an instrument like unit-based pricing requires not only information about the effects of waste quantity, but also of waste collection, treatment and external costs. This necessitates the bundling of figures presented in the different chapters.

In the next paragraph we present the total social costs of current regulation as a benchmark for the effects of the analysed instruments.[1] In the following paragraphs the effects on total social costs of respectively unit-based pricing, contracting out waste collection, a preference for landfilling and open national borders for waste treatment are summarized. Finally, the last paragraph shows the effect on total costs when all instruments are combined.

Current regulation

In this paragraph the total social costs of the current regulation regime is calculated as a benchmark for the effects of changes in regulation. To make this calculations possible the following assumptions are made:

- The quantities of waste are equal to 5.60 Mton unsorted waste, 1.44 Mton com-

[1] The presented figures are only average estimates. See for more information on the probability of the outcomes the discussions presented in the different chapters.

postable waste and 1.37 Mton recyclable waste. This means that the calculated costs refer to the long term case as these quantities apply to 2011 (see page 161).

- Collection costs for unsorted and compostable waste are equal to 58 euro per ton. This figure is calculated using the average costs of the 85 municipalities presented in chapter 3 (see page 45) divided by the average number of pickup-points (see page 45) and by the average waste quantity per pickup-point. The average waste quantity per pickup-point is equal to the total quantity of waste collected divided by the estimated number of pickup-points (see page 32 and 45). The collection costs for recyclables are equal to 35 euro per ton and based on Doppenberg and Cras (1997). Note that these costs are lower than the costs for unsorted and compostable waste as for recyclables the number of pickup-points is significant lower due to the chosen bring system.

- The waste treatment costs per ton for unsorted waste are equal to 96 euro per ton (including transport costs) and follow directly from the model simulations presented in chapter 6 (see page 129). Note that this implicates that we assume that in the long term all household waste is incinerated, which is in accordance with the current regulation and the model simulations. The treatment costs for compostable waste are equal to the average costs of Dutch compost plants (71 euro per ton) and based on Dijkgraaf et al. (1999). The treatment costs for recyclable waste are not easy to observe as they are based on a combination of different technologies for which no detailed cost information is available. The used 30 euro per ton is a guestimate based on information from VROM (2003) and the internetpages of representing organizations. Note that for the effects of instruments these last two assumptions are not important as the instruments do not have effects on these costs.

- External costs for unsorted waste are equal to 18 euro per ton, the environmental costs of incineration (see page 88). Thus we assume that in the long term all household waste is incinerated, which is in accordance with current regulation and the outcome of the model simulations presented in chapter 6. External costs for compostable waste are an estimated 50% of the external costs for incineration. This assumption is based on the calculations presented in Brisson (1997). As no reliable source could be found for the external costs of recyclables, these costs are omitted (as the instruments analysed do not influence these costs this does not influence the conclusions).

Based on these assumptions Table 7.1 presents total social costs of household waste collection and treatment for the current regulation regime. In total these costs are 1252 million euro, while the collection and treatment of unsorted waste is responsible for the major part of these costs (964 million euro).

Table 7.1: Total yearly social costs with current regulation

	Unsorted	Compostable	Recyclable	Total
Quantity in Mton	5.60	1.44	1.37	8.41
Collection costs per ton	58	58	35	
Treatment costs per ton	96	71	30	
External costs per ton	18	9	na	
Total costs in million euro	964	198	89	1252

Unit-based pricing

Chapter 2 shows that the bag-based system is probably the preferred unit-based pricing system from an economic point of view. When bag-based pricing is introduced in all municipalities two effects occur. First, waste quantity decreases. Second, administrative costs increase. The decrease in waste quantity after introduction of the bag-based system is calculated by summing the number of inhabitants living in municipalities without bag-based pricing and multiplying this number by the estimated decrease in unsorted and compostable waste and the increase in recyclable waste per inhabitant.[2] This results in a decrease of total waste with 2.44 Mton, mainly driven by the decrease in unsorted waste of 2.0 Mton. Compostable waste diminishes by more than 60%. It seems that many Dutch households use home composting methods to reduce this type of waste. Also, the effect on unsorted waste is large. One of the important mechanisms generating this result is that the amount of recyclable waste increases when a unit-based pricing system is introduced. This is due to the fact that Dutch citizens do not have to pay a marginal price for the collection of this type of waste. The rise in collection costs due to higher administrative costs is assumed to be equal to 3.18 euro per inhabitant (see page 36). This means that per ton of waste collection costs increases with 8 euro, as the average waste quantity per inhabitant is 420 kg.

[2] The assumption is that the environmental activism effect is internalized by unit-based pricing for municipalities with an a-priori low level of activism. If this internalization is incomplete total costs will be higher.

Table 7.2: Cost effects of the bag-based system

	Unsorted Level Change	Compostable Level Change	Recyclable Level Change	Total Level Change
Quantity in Mton	3.60 (-2.0)	0.61 (-0.83)	1.75 (+0.38)	5.97 (-2.44)
Collection costs per ton	66 (+8)	66 (+8)	35	
Treatment costs per ton	96	71	30	
External costs per ton	18	9	na	
Total costs in million euro	647 (-317)	90 (-109)	114 (+25)	851 (-401)

Change: change with current regulation

Table 7.2 summarizes the estimated influence on total costs if all municipalities with a fixed fee system introduce a bag-based system and if all municipalities with another unit-based pricing system replace this by the bag-based system. The rise in collection costs, due to the administrative costs of the bag-based system, are more than compensated by the net decrease in collected waste. In total a cost saving of 401 million euro is possible when all municipalities introduce a bag-based pricing system.[3] Thus, unit-based pricing has a major influence on total costs.

Contracting out waste collection

Chapter 3 shows that a cost decrease is possible when municipalities that collect the waste themselves contract out to private firms. When all Dutch municipalities that can gain from contracting out introduce this instrument their costs will decrease with 14.8% (see page 52). As 51.5% of the inhabitants live in a municipality with inside collection, the effect on the total average collection costs is 7.6% (compare page 45). This means that total social cost will diminish with an estimated value of 31 million euro (see Table 7.3). That these savings are considerable lower than for unit-based pricing results not only from the higher incidence of contracting out and the smaller effect in relative terms, but also from the fact that contracting out reduces only collection costs while unit-based pricing saves also waste treatment and external costs. Nonetheless, municipalities that introduce contracting out can reduce collection costs considerably.

[3] Note that we assume that the external costs do not rise when unit-based pricing is introduced. This is only the case when there is no increase in illegal dumping. This assumption is based on the observation that in the administrative costs of the bag-based system costs for more surveillance are included and on the fact that clear anecdotal evidence for significant illegal dumping does not exists. See paragraph 2.5.

Table 7.3: Contracting-out inside collection to private outside firms

	Unsorted		Compostable		Recyclable		Total	
	Level	Change	Level	Change	Level	Change	Level	Change
Quantity in Mton	5.60		1.44		1.37		8.41	
Collection costs per ton	54	(-4)	54	(-4)	35			
Treatment costs per ton	96		71		30			
External costs per ton	18		9		na			
Total costs in million euro	939	(-25)	192	(-6)	89		1220	(-31)

Change: change with current regulation

Landfilling preferred

Chapter 5 shows that landfilling is preferred from a social cost point of view. As the treatment costs per ton are significant lower than for incineration (36 euro instead of 79, see page 87) and the net external costs are only marginal higher (see page 88) the use of landfilling instead of incineration will decrease total social costs with 116 million euro (see Table 7.4). Note that, in accordance with the analysis in chapter 6, this figure is based on the assumption that only household waste that is not yet contracted with long term contracts by waste incineration plants (37% of total waste, compare page 124) goes to the cheaper landfills. This means that the decrease in total social costs is larger (221 million euro) when these contracts expire.

Table 7.4: Landfilling preferred (for household waste not yet contracted)

	Unsorted		Compostable		Recyclable		Total	
	Level	Change	Level	Change	Level	Change	Level	Change
Quantity in Mton	5.60		1.44		1.37		8.41	
Collection costs per ton	58		58		35			
Treatment costs per ton	74	(-20)	71		30			
External costs per ton	19	(+1)	9		na			
Total costs in million euro	848	(-116)	198		89		1135	(-116)

Change: change with current regulation

Open borders

Chapter 6 shows that minimization of social costs requires open borders for the treatment of waste with taxes on treatment options set to their external costs. When these changes are implemented the model simulations presented in chapter 6 estimate a treatment cost decrease of 15 euro per ton, while the external costs rise with only

1 euro per ton. Compared with the current regulation regime and including external costs, total costs decrease with 81 million euro (see Table 7.5).[4]

Table 7.5: Open borders, taxes according to external costs

	Unsorted		Compostable		Recyclable		Total	
	Level	Change	Level	Change	Level	Change	Level	Change
Quantity in Mton	5.60		1.44		1.37		8.41	
Collection costs per ton	58		58		35			
Treatment costs per ton	81	(-15)	71		30			
External costs per ton	19	(+1)	9		na			
Total costs in million euro	884	(-81)	198		89		1171	(-81)

Change: change with current regulation

All instruments

The potential cost decrease when a specific regulation instrument is changed is not always independent of changes in other instruments. For example, when a bag-based pricing system is introduced in municipalities that have currently not such a system, the total quantity of waste will diminish. This means that the cost reduction of opening national borders for waste treatment is smaller. Table 7.6 summarizes the interaction between the different instruments.

Table 7.6: All instruments

	Unsorted		Compostable		Recyclable		Total	
	Level	Change	Level	Change	Level	Change	Level	Change
Quantity in Mton	3.60	(-2.0)	0.61	(-0.83)	1.75	(+0.38)	5.97	(-2.44)
Collection costs per ton	61	(+3)	61	(+3)	35			
Treatment costs per ton	81	(-15)	71		30			
External costs per ton	19	(+1)	9		0			
Total costs in million euro	579	(-385)	111	(-111)	114	(+25)	780	(-471)

Change: change with current regulation

If all instruments are implemented according to the goal of minimizing social cost (a bag-based pricing system in all municipalities, private outside collection in municipalities that have currently inside collection, open national borders and taxes on treatment options in accordance with external costs) social costs will decrease from 1252 million euro to

[4] Table 7.5 only reports cost effects for household waste. While most instruments only influences costs of household waste collection and treatment, open national borders may also result in a 27 million euro cost decrease for firm waste.

780 million euro. This means that changes in regulation can result in a total social cost reduction of 471 million euro per year (38% of current costs).[5]

[5] This is exclusive the 27 million euro cost decrease for firms.

Chapter 8

Samenvatting

8.1 Introductie

Regulering speelt een belangrijke rol in the afvalmarkt. De potentiële effecten op gezondheid en milieu als geen regulering aanwezig is zijn dusdanig groot dat maatregelen nodig zijn. In de afgelopen dertig jaar zijn dan ook een aantal instrumenten ingezet. Een van de eerste (nationale) maatregelen die genomen werd, is de verplichting van gemeenten om een inzamelstructuur te garanderen zodat voorkomen wordt dat afval zich ophoopt in de woonomgeving. Dit om te voorkomen dat burgers gezondheidseffecten en hinder ondervinden. Een tweede maatregel die genomen werd bestond uit het implementeren van richtlijnen voor stortplaatsen en afvalverbrandingsinstallaties (AVI's). Het storten en verbranden van afval kan dusdanige effecten op gezondheid en milieu hebben dat dergelijke regelgeving wenselijk werd geacht. De wenselijkheid hiervan wordt onderstreept door affaires waarbij het verbranden van afval leidde tot te hoge concentraties kankerverwekkende dioxines en furanen en door tal van voorbeelden van stortplaatsen waar chemische substanties doordrongen tot het grondwater. In de derde plaats werd regelgeving opgesteld om afvalstromen te sturen naar verwerkingsopties met het laagste niveau van ongewenste effecten op het milieu. De verschillende verwerkingsmogelijkheden werden expliciet geordend zodat een prioriteitsvolgorde ontstond (de ladder van Lansink). Tenslotte verbood nationale regelgeving de export van afval naar buitenlandse stortplaatsen en AVI's omdat export werd gezien als een onwenselijke optie. Het buitenland zou geen hinder moeten ondervinden van afval dat in Nederland werd geproduceerd.

Als gevolg van deze regulering stegen de afgelopen dertig jaar de kosten aanzienlijk. Zolang deze stijging nodig is om onwenselijke externe effecten van inzameling en verwerking te voorkomen is dit niet meer dan een historisch feit. Er bestaat echter vanuit economisch oogpunt op zijn minst indirect bewijs dat andere vormen van regulering mogelijk zijn die tot lagere kosten leiden. De centrale vraag in dit proefschrift is dan ook of veranderingen in regulering ertoe kunnen bijdragen dat de totale sociale kosten van het inzamelen en verwerken van afval afnemen. Hierbij zijn de totale sociale kosten gelijk aan de som van private en externe kosten. Private kosten zijn gelijk aan de noodzakelijke productiekosten. De sociale kosten kunnen echter hoger zijn dan deze productiekosten als er externe kosten bestaan. Externe kosten zijn kosten gerelateerd aan productie of consumptie die geen rol spelen in de beslissingen die op de afvalmarkt genomen worden. De prijs betaalt voor het verbranden van een ton afval is bijvoorbeeld gebaseerd op de kosten van benodigd kapitaal en arbeid, maar niet op de kosten veroorzaakt door het emitteren van CO_2 of dioxines. Daar deze emissies wel kosten veroorzaken voor de maatschappij, bijvoorbeeld door het veroorzaken van het broeikaseffect of het vergroten van de kans op kanker, moet een goede economische afweging zowel de private als externe kosten meenemen. In wiskundige termen zijn de sociale kosten gelijk aan:

$$TSC = W\left(C_C^P + C_C^E + C_T^P + C_T^E\right), \tag{8.1.1}$$

met TSC de totale sociale kosten (de som van private en externe kosten), W de totale hoeveelheid opgehaald afval, C_C^P de private kosten van afvalinzameling (exclusief de externe kosten), C_C^E de externe kosten van afvalinzameling, C_T^P de private kosten van afvalverwerking en C_T^E de externe kosten van afvalverwerking.

De hoofdstukken 2 tot en met 6 van dit proefschrift onderzoeken ieder een deel van vergelijking 8.1.1 voor Nederland aan de hand van vier specifieke vragen. De **eerste twee vragen** zijn gerelateerd aan de inzamelmarkt:

- Is het mogelijk de hoeveelheid ingezameld afval (W) te verlagen door gebruik te maken van systemen met een marginale prijs voor afval (Diftar) (hoofdstuk 2)?
- Is het mogelijk, gegeven de hoeveelheid ingezameld afval (W), om de totale private inzamelingskosten (C_C^P) te verminderen door gebruik te maken van het aanbeste-

den van de inzameling (hoofdstuk 3) en, als dit het geval is, welke factoren kunnen dan verklaren dat relatief veel gemeenten geen gebruik maken van dit instrument (hoofdstuk 4)?

Zoals gezegd hebben gemeenten een verplichting om een inzamelstructuur voor huishoudelijk afval te organiseren. Hoe ze dit doen mogen ze zelf weten. Ze kunnen ervoor kiezen om de inzameling zelf uit te voeren met behulp van een eigen gemeentelijke dienst, maar ze kunnen er ook voor kiezen om de inzameling uit te besteden aan andere gemeenten of private bedrijven. Tevens kunnen gemeenten kiezen hoe ze hun burgers de kosten van de inzameling in rekening brengen. Sommige gemeenten kiezen voor een vast bedrag per jaar. Anderen rekenen burgers af met Diftar. In dat geval is de uiteindelijke rekening afhankelijk van het volume van de container, het aantal geledigde containers of het aantal ingezamelde kilogrammen of zakken.

Nederlandse gemeenten verschillen aanzienlijk in de keuzes die ze maken. Op het eerste gezicht conflicteren deze verschillen met de empirische literatuur. Internationale ervaringen die in deze literatuur gerapporteerd worden suggereren dat gemeenten die de inzameling aanbesteden en met behulp van Diftar afrekenen aanzienlijk goedkoper uit zijn. Aanbesteden is een instrument gericht op het introduceren van marktprikkels en het daardoor bevorderen van de efficiëntie. Volgens de literatuur leidt de invoering van Diftar tot prikkels om minder afval aan te bieden omdat nu voor het ingezameld afval betaald moet gaan worden. De internationale literatuur kan echter misleidend zijn omdat de buitenlandse ervaringen mogelijk niet representatief zijn voor de specifieke Nederlandse situatie. Bovendien geeft de literatuur geen sluitend antwoord op een aantal aanbestedings- en Diftarvragen doordat een aantal aspecten niet onderzocht is. Daardoor ontbreekt informatie om tot een goede economische afweging te komen tussen de beschikbare instrumenten. Hoofdstuk 2 en 3 zijn bedoeld om deze informatie te genereren.

In de hoofdstukken 2 en 3 wordt gevonden dat Diftar en aanbesteden inderdaad te prefereren zijn vanuit een economische invalshoek. De vraag is dan wel waarom niet alle gemeenten van deze methoden gebruik maken. Terwijl voor Diftar een duidelijke reden voor de hand ligt, het betreft immers een recent ontwikkeld systeem zodat vooruitstrevende gemeenten een dergelijke systeem eerder invoeren dan gemeenten die een meer afwachtende houding hebben, is dit voor aanbesteden niet het geval. Daarom wordt in hoofdstuk 4 geanalyseerd welke oorzaken kunnen verklaren dat aanbesteden van de afvalinzameling in een groot aantal gemeenten niet toegepast wordt. Op basis van de economische theorie worden deze oorzaken empirisch getoetst.

De **laatste twee vragen** zijn gerelateerd aan de afvalverwerkingsmarkt:

- Is het mogelijk, gegeven de hoeveelheid ingezameld afval (W), om de totale verwerkingskosten($C_T^P + C_T^E$) te verminderen door de voorkeursvolgorde tussen verbranden en storten te veranderen (hoofdstuk 5)?
- Is het mogelijk, gegeven de hoeveelheid ingezameld afval (W) en de voorkeursvolgorde, om de totale private verwerkingskosten (C_T^P) te verlagen door internationale concurrentie te introduceren (hoofdstuk 6)?

In het kader van vermindering van de milieudruk heeft de overheid gekozen voor een voorkeursvolgorde waar verbranden van afval de voorkeur verdient boven storten. Destijds werd deze volgorde gemotiveerd op basis van de milieubelasting van storten. Er vond destijds echter geen expliciete vergelijking plaats tussen beide opties op basis van externe en private kosten. Zolang de afname in milieukosten opweegt tegen de hogere private kosten is verbranden inderdaad de te prefereren optie vanuit economisch oogpunt. Gezien de grote toename in private kosten als omgeschakeld wordt naar verbranden is het echter op zijn minst de vraag of dit in de praktijk ook opgaat. Daarom presenteert hoofdstuk 5 een afweging tussen beide opties op basis van een inschatting van externe en private kosten.

Door de regulering waarmee de afvalmarkt geconfronteerd werd, ontstond ruimte voor monopolistisch gedrag. Concurrentie tussen stortplaatsen en AVI's verdween door de introductie van een hoge stortbelasting. Deze belasting werd geïntroduceerd om ervoor te zorgen dat afvalstromen daadwerkelijk bij AVI's aangeboden zouden worden. Daar komt nog bij dat de concurrentie op de AVI-markt eveneens beperkt is. Niet alleen is export naar buitenlandse installaties verboden, gedurende een aantal jaren was zelfs sprake van een regionale Nederlandse markt. Een interessante vraag is of deze marktrestricties nodig zijn om de doelen van de overheid te halen. Deze vraag is van belang omdat monopolistisch gedrag tot hogere kosten kan leiden. Hoofdstuk 6 presenteert dan ook een simulatiemodel waarmee deze vragen geanalyseerd kunnen worden.

Ter afsluiting van deze introductie is het noodzakelijk twee opmerkingen te maken die dit proefschrift in het juiste perspectief plaatsen. In de eerste plaats behandelen de vier centrale onderzoeksvragen de belangrijkste determinanten van de totale sociale inzamelen verwerkingkosten die beïnvloed kunnen worden door veranderingen van regulering. Dit betekent echter niet dat alle aspecten van de afvalmarkt en de sociale kosten onderdeel van dit proefschrift uitmaken. In het algemeen is het zo dat dit proefschrift gericht is op die determinanten van de totale sociale kosten waarvoor verwacht kan worden dat veranderingen van nationale regulering een significant effect kunnen hebben. De aanname daarbij is dat dat het geval is wanneer op dit moment regulering bestaat die minimalisering van de sociale kosten belemmert. In feite zijn de gekozen determinanten van de sociale kosten, met uitzondering van de voorkeursvolgorde verbranden - storten, gebaseerd op de observatie dat de markten voor preventie en hergebruik en de markt voor de inzameling van bedrijfsafval niet in belangrijke mate gehinderd worden door nationale regulering. De markten voor preventie en hergebruik zijn bijvoorbeeld geen onderwerp van onderzoek omdat het hier (in overeenstemming met EU-wetgeving) markten betreft die op Europese schaal concurreren. In de tweede plaats is van belang te constateren dat dit proefschrift geen alomvattende antwoorden bevat over de onderzochte deelmarkten. De lijst met vragen voor toekomstig onderzoek maakt duidelijk dat dit proefschrift eerder een steen is in het groeiende bouwwerk van economisch afvalonderzoek.

8.2 Belangrijkste bevindingen

In hoofdstuk 2 staat de vraag centraal of invoering van Diftar de totale hoeveelheid Nederlands afval reduceert. Het achterliggende idee is dat de introductie van een marginale prijs per eenheid ingezameld afval burgers de prikkel geeft om hun afval te reduceren. Bovendien ontstaat zo een prikkel om het afval beter te scheiden omdat een aantal gescheiden ingezamelde afvalstoffen (papier, glas, textiel) ook in Diftar gemeenten gratis ingezameld wordt. Nederlandse gemeenten zijn namelijk verplicht om een gratis inzamelstructuur voor deze afvalstoffen aan te bieden.

Op basis van een dataset met waarnemingen voor bijna alle Nederlandse gemeenten wordt geschat of Diftar tot de voorspelde effecten leidt. Dit wordt gedaan voor systemen gebaseerd op gewicht (betaling per kilo), zakken (er moeten dan zogenaamde dure zakken gekocht worden, andere zakken worden niet ingezameld), frequentie (betaling per lediging) of volume (de burger kan dan kiezen tussen verschillende maten containers). Gecorrigeerd voor bepaalde factoren die verschillen tussen gemeenten wordt de hoeveelheid ingezameld afval vergeleken tussen gemeenten met en zonder Diftar. In het algemeen blijkt dat Diftar inderdaad effectief is in het reduceren van restafval en GFT en het stimuleren van gescheiden ingezamelde afvalstoffen. Een deel van dit effect blijkt voort te komen uit het hogere niveau van milieubewustzijn in gemeenten die Diftar invoeren. Dit duidt erop dat het niveau van ingezameld rest- en GFT-afval al lager is in gemeenten die later Diftar invoeren. Dit effect komt voort uit het hogere milieubewustzijn van de burgers in deze gemeenten. Als niet voor dit effect gecorrigeerd wordt, leiden de schattingen dus tot een overschatting van het Diftareffect. Voor gemeenten met een relatief laag niveau van milieubewustzijn kan invoering van Diftar echter leiden tot internalisering van het gebrek aan milieubewustzijn zodat voor deze gemeenten Diftar een relatief groot effect kan hebben.

De resultaten duiden er vervolgens op dat er belangrijke verschillen bestaan tussen de diverse Diftarsystemen. Het zakken- en gewichtssysteem presteren vergelijkbaar en beter dan het frequentie- en volumesysteem. Daar de administratieve kosten significant lager zijn voor het zakkensysteem is dit systeem te prefereren vanuit economisch oogpunt.

Alhoewel illegale dumping nogal eens verantwoordelijk wordt geacht voor een deel van het Diftareffect, wat dan leidt tot een stijging van de externe inzamelkosten (C_C^E), duiden de schattingsresultaten erop dat er geen empirisch bewijs is voor afvaltoerisme. Invoering van Diftar leidt er niet toe dat de hoeveelheid ingezameld afval in naburige gemeenten die geen Diftar hebben toeneemt. Hierbij moet wel aangetekend worden dat andere vormen van illegale dumping niet geanalyseerd konden worden omdat hiervoor onvoldoende gegevens beschikbaar zijn.

Als alle gemeenten een zakkensysteem invoeren dalen de totale sociale kosten met ongeveer 400 miljoen euro per jaar. Dit is 32% van de huidige kosten. De conclusie is daardoor gerechtvaardigd dat invoering van Diftar leidt tot een significante en grote daling van de totale sociale kosten.

In hoofdstuk 3 worden de mogelijke kostenbesparingen (C_C^P) geanalyseerd als Nederlandse gemeenten hun afvalinzameling aanbesteden. Dit wordt gedaan op basis van een enquête onder 120 a-select getrokken Nederlandse gemeenten, waarvan er 85 gereageerd hebben. Het blijkt dat een kostenbesparing van zo'n 15% mogelijk is bij aanbesteding. Dit is vergelijkbaar met resultaten gevonden voor andere landen. Verder blijkt uit de schattingen dat gemeentelijke inzameldiensten een andere productietechnologie gebruiken vergeleken met externe inzamelbedrijven. Hierdoor moeten verschillende kostenfuncties geschat worden en voldoet de standaard aanpak die in de literatuur gebruikt wordt niet. Overigens blijkt dat als deze systematiek gehanteerd wordt de inschatting van de te behalen kostenbesparing bij uitbesteden niet significant verandert. Gemeenten ondervinden door het fiscale systeem niet de juiste prikkels voor aanbesteding. Dit komt omdat bij aanbesteding BTW betaald moet gaan worden terwijl dit niet hoeft als de gemeente zelf het afval inzamelt. Hoewel dit geen effect heeft op de maatschappelijke kosten, leidt het er wel toe dat de financiële prikkel voor gemeenten afneemt. Het recente opgerichte BTW-compensatiefonds is vanuit dit gezichtspunt dan ook een goede zaak.

Als alle Nederlandse gemeenten die kunnen profiteren van aanbesteding (dit zijn de gemeenten die zelf hun afval inzamelen en vervolgens aanbesteden aan private inzamelbedrijven) dit instrument gebruiken, kunnen de totale sociale kosten afnemen met zo'n 30 miljoen euro. Dat dit fors lager is dan de effecten van Diftar komt niet alleen omdat de penetratiegraad van aanbesteding al fors hoger is en het relatieve effect kleiner, maar ook omdat aanbesteding 'slechts' de inzamelkosten vermindert terwijl Diftar zowel bespaart op de inzamelkosten als op de verwerkingskosten.

Gezien de kostenverlaging die gemeenten kunnen bereiken door de afvalinzameling aan te besteden zou vanuit economisch oogpunt verwacht kunnen worden dat gemeenten dat ook massaal doen. De meerderheid van de Nederlandse gemeenten verzorgt echter de afvalinzameling met behulp van een eigen inzamelingsdienst zonder dat ooit gebruik gemaakt is van aanbesteding. In hoofdstuk 4 wordt een verklaring gezocht voor deze gereserveerdheid. Gebaseerd op theoretische inzichten wordt de keuze gemodelleerd tussen private, publieke, interne en externe inzameling. Met behulp van een dataset met gegevens van nagenoeg alle Nederlandse gemeenten worden deze theorieën getoetst. Het blijkt dat het aantal inwoners, de bijdrage van het Rijk aan het gemeentefonds en belangengroepen een belangrijke verklaring bieden. Interessant is dat ideologische argumenten nauwelijks een rol blijken te spelen. Vergeleken met eerdere studies worden in hoofdstuk 4 meer algemene modellen geschat. Alhoewel de resultaten in kwalitatieve zin vergelijkbaar zijn voor parametrische en semi-parametrische schattingsmethoden, blijkt de parametrische methoden te inflexibel. De marginale effecten die beide methoden schatten laten aanzienlijke verschillen zien. Daarom is meer aandacht nodig voor de implicaties van modelspecificatie.

Hoofdstuk 5 gebruikt gegevens over private en externe kosten om de voorkeursvolgorde voor verbranden boven storten te evalueren op basis van een vergelijking van de sociale kosten. Het blijkt dat slechts op basis van milieukosten een dergelijke voorkeursvolgorde op economische gronden te verdedigen valt. Als de private kosten in de analyse betrokken worden, is storten duidelijk te prefereren boven verbranden omdat de iets hogere milieukosten van de laatste optie niet opwegen tegen de grote stijging van de private kosten. Dit is, gezien het ruimtebeslag van storten, een opmerkelijke conclusie voor een dichtbevolkt land als Nederland. Voor het Nederlandse beleid betekent dit niet alleen dat het huidige instrumentarium gericht op het sturen van afvalstromen niet optimaal is, maar ook dat het stimuleren van energieterugwinning van stortplaatsen van groot belang is.

Als de voorkeursvolgorde inderdaad omgedraaid wordt, dalen de totale sociale kosten met ongeveer 115 miljoen euro per jaar. Dit cijfer is gebaseerd op de aanname dat alleen het afval dat niet middels lange termijn contracten gecontracteerd is bij AVI's daadwerkelijk naar de stortplaats gaat. Dit betekent dat de afname van de sociale kosten nog groter is als deze contracten aflopen. Naar schatting gaat het dan om een bedrag van zo'n 220 miljoen euro.

Hoofdstuk 6 laat zien dat de gekozen Nederlandse doelstellingen voor de afvalmarkt (zelfvoorziening voor verwijdering en een voorkeur voor verbranden boven storten) bereikt kunnen worden onafhankelijk van de vraag of nationale grenzen geopend of gesloten worden. De doelstellingen kunnen bereikt worden als de stortbelasting afgeschaft wordt en er een subsidie gegeven wordt aan AVI's. In dit geval is zelfs met open grenzen een zelfvoorzieningspercentage van 100% en een stortpercentage van 0% mogelijk. Vanuit kostenperspectief is echter een harmonisatie van stortbelastingen op Europees niveau een betere optie. De totale kosten dalen hierdoor terwijl een zelfvoorzieningspercentage van 90% en een stortpercentage van 0% haalbaar is. In de huidige beleidscontext is dit echter een moeizaam begaanbare weg daar momenteel de meeste EU-lidstaten liever geen stortbelasting invoeren. De beste beleidsoptie is dan om op korte termijn te kiezen voor een nationale subsidie (en een stortbelasting van nul euro), terwijl op de langere termijn gestreefd wordt naar harmonisatie van een stortbelasting op een dergelijk niveau dat de kosten inclusief belasting van storten en verbranden gelijk zijn. In alle gevallen is het vanuit het kostenperspectief aantrekkelijk om nationale grenzen te openen.

Als niet uitgegaan wordt van de huidige doelstellingen maar van doelstellingen die in overeenstemming zijn met economische uitgangspunten leidt dit tot andere reguleringsinstrumenten. Als voor alle verwerkingsopties een belasting geïntroduceerd wordt die gelijk is aan de externe kosten is op korte termijn een zelfvoorzieningspercentage van slechts 61% en een stortpercentage van 38% optimaal. Voor de langere termijn, als de bestaande AVI's in de loop van de tijd buiten gebruik gesteld worden, neemt het zelfvoorzieningspercentage af, terwijl het stortpercentage toeneemt. Dit wordt veroorzaakt door de lagere sociale kosten van storten in naburige landen waardoor deze optie voor de meeste afvalstromen te prefereren valt. Vergeleken met het beste reguleringspakket op basis van de huidige beleidsdoelstellingen is een besparing op de private kosten mogelijk van zo'n 400 miljoen euro per jaar. Als deze beleidsdoelstellingen minder strikt geïnterpreteerd worden, daalt de besparing. Een dergelijke interpretatie valt te verdedigen als er slechts export van afval plaatsvindt naar kolencentrales en cementovens, terwijl in het buitenland alleen afval gestort wordt dat noodzakelijkerwijs vrijkomt bij het scheidingsproces nodig om het afval voor deze opties geschikt te maken. Zelfs in dit geval is de totale besparing op de sociale kosten nog 80 miljoen euro als niet van de huidige (minder strikt geïnterpreteerde) beleidsdoelstellingen wordt uitgegaan maar van instrumenten gebaseerd op de externe kosten.

Tenslotte zij opgemerkt dat de effecten van specifieke instrumenten op de totale sociale kosten niet onafhankelijk zijn van de vraag of andere instrumenten al dan niet ingevoerd worden. Als bijvoorbeeld het zakkensysteem ingevoerd wordt in alle Nederlandse gemeenten zal de totale hoeveelheid te verwerken afval verminderen. Dit betekent dat de kostenreductie van het openen van nationale grenzen voor te verwerken afval afneemt. Als alle in dit proefschrift besproken instrumenten die op basis van economische inzichten optimaal zijn tegelijkertijd ingezet worden (een zakkensysteem, aanbesteding van de inzameling, open nationale grenzen en belastingen op verwerkingsopties die gelijk zijn aan de externe kosten) nemen de totale sociale kosten af van zo'n 1250 miljoen euro naar zo'n 780 miljoen euro. Dit betekent dat veranderingen in de regulering van de afvalmarkt een totale besparing op de sociale kosten kunnen opleveren van zo'n 470 miljoen euro per jaar. Dit is gelijk aan ongeveer 38% van de huidige kosten.

8.3 Toekomstig onderzoek

Dit proefschrift analyseert de belangrijkste onderdelen van de afvalketen waarvoor a-priori verwacht mocht worden dat een reductie van de totale sociale kosten mogelijk is door veranderingen door te voeren van reguleringsinstrumenten. Het onderzoek gepresenteerd in dit proefschrift heeft een aantal zaken aan het licht gebracht die in de toekomst nader onderzoek verdient.

In de eerste plaats is het belangrijkste nadeel van Diftar het effect op illegale dumping (zie hoofdstuk 2). Op dit moment is echter onvoldoende bewijs voorhanden of dit effect nu belangrijk is of niet. Hoewel dit proefschrift aantoont dat van afvaltoerisme (stijging van de afvalhoeveelheid in naburige gemeenten zonder Diftar) geen sprake is, is geen omvattende analyse mogelijk gebleken van illegale dumping. Daar illegale dumping het belangrijkste overblijvende argument is voor de opponenten van Diftar zou het goed zijn als toekomstig onderzoek gericht wordt op het vergroten van het bewijs of al dan niet sprake is van significante effecten op illegale dumping. Als dit zo blijkt te zijn is vervolgens van belang welke instrumenten ingezet kunnen worden om dit effect tegen te gaan. Hierbij kan gedacht worden aan een effectief monitorings- en boetesysteem. Als illegale dumping geen belangrijk issue blijkt te zijn is het aan te bevelen om het penetratieproces van Diftar te analyseren. De grote geschatte effecten op de hoeveelheid afval in combinatie met de relatief lage penetratiegraad van Diftar vragen om een zorgvuldige analyse van de vraag welke factoren bepalen of Diftar al dan niet ingevoerd wordt. Het zou bijvoorbeeld zo kunnen zijn dat het feit dat Diftar een relatief nieuw instrument is een belangrijke verklaring biedt voor de lage penetratiegraad. De onzekerheid over de werkelijke effecten en over illegale dumping zouden ervoor kunnen zorgen dat alleen vooroplopende gemeenten bereid zijn om Diftar te proberen terwijl een groot aantal afwachtende gemeenten dit pas doen als meer duidelijkheid over de effecten van Diftar aanwezig is.

In de tweede plaats concludeert hoofdstuk 3 dat interne en externe inzamelingsbedrijven een verschillende productietechnologie gebruiken. Om deze conclusie verder te onderbouwen en te analyseren welke consequenties dit heeft voor de geschatte efficiëntie-effecten zou toekomstig onderzoek gebaseerd moeten zijn op een grotere dataset zodat meer precieze schattingen plaats kunnen vinden over de invloed van aanbesteding op de inzamelingskosten. Bovendien maakt dit panelschattingen mogelijk waardoor inzicht kan ontstaan in de persistentie van het kostenvoordeel. Dit kan belangrijk zijn omdat collusie en lock-in effecten het kostenvoordeel op langere termijn kunnen mitigeren.

In de derde plaats laat hoofdstuk 4 zien dat niet alleen kostenoverwegingen een rol spelen bij de keuze voor de inzamelingsmethode. Aangezien een groot aantal gemeenten niet kiest voor aanbesteding verdient nader onderzoek naar deze overwegingen aanbeveling. Het zou bijvoorbeeld zo kunnen zijn dat gemeenten de inzameling liever niet aanbesteden omdat ze dan bang zijn de grip op de toekomstige koers van de inzameling te verliezen. De vrees om de controle te verliezen als een onomkeerbaar besluit wordt genomen zou impliceren dat de (indirecte) kosten van aanbesteding hoger zijn. Als deze kosten meegenomen worden, zou dit de attractiviteit van aanbesteden kunnen schaden. Ditzelfde mechanisme zou overigens een rol kunnen spelen bij de keuze voor een nationale dan wel een internationale afvalverwijderingsmarkt (hoofdstuk 6).

In de vierde plaats verdient de wijze waarop de toekomstige kosten worden meegenomen meer aandacht bij toekomstig onderzoek naar de vraag welke verwerkingsoptie de voorkeur verdient vanuit economisch perspectief. In hoofdstuk 5 is niet veel aandacht aan deze vraag besteed terwijl het verschil in tijdspatroon voor wat betreft de milieueffecten van respectievelijk storten en verbranden aanzienlijk verschilt. Waar de emissies van verbranden voor het overgrote deel nagenoeg onmiddellijk plaatsvinden, geldt dit veel minder voor de milieueffecten van storten. Daar komt nog bij dat de gehanteerde schaduwprijzen constant in de tijd zijn verondersteld. Het zou daarom goed zijn als toekomstig onderzoek meer aandacht schenkt aan de dynamische karakteristieken van storten versus verbranden.

In de vijfde plaats is aanvullende onderzoek vereist naar de vraag of de conclusies zoals gepresenteerd in hoofdstuk 5 ook opgaan voor alle typen afval. In hoofdstuk 5 is de analyse met name gericht op zogenaamd laagcalorisch afval (afval van huishoudens). Eerste vingeroefeningen (zie hoofdstuk 6) duiden erop dat vanuit een economische oogpunt storten ook de voorkeur verdient boven verbranden voor hoogcalorische afval (afval van bedrijven), maar dat verbranden in een cementoven een nog betere optie is. Dit laatste wordt met name veroorzaakt door het ontbreken van milieukosten verbonden met het storten van chemisch afval daar in het cementovenproces al het afval geabsorbeerd wordt. Toekomstig onderzoek zou daarom gericht kunnen worden op het vergelijken van meer opties voor diverse afvalstromen.

In de zesde plaats houdt de analyse zoals gepresenteerd in hoofdstuk 6 geen rekening met de relatie tussen preventie, hergebruik en verwijdering. Als marktomstandigheden in de verwijderingsmarkt veranderen zou dit tot gevolg kunnen hebben dat prijzen en hoeveelheden op de markt voor preventie en hergebruik beïnvloed worden. Als bijvoorbeeld het tarief voor verbranden afneemt, verlaagt dit de prikkels voor preventie en hergebruik wat kan leiden tot een toename van te verwijderen afvalstoffen. In hoofdstuk

6 is (impliciet en in overeenstemming met het huidige Nederlandse beleid) aangenomen dat inzet van andere instrumenten mogelijk is om deze effecten te compenseren. Als echter de veranderingen in de verwijderingsmarkt leiden tot significante tariefaanpassingen is het van belang om meer aandacht te schenken aan de verbanden tussen de diverse markten.

In de zevende plaats is het model zoals gepresenteerd in hoofdstuk 6 gebaseerd op de aanname dat sprake is van volledige concurrentie. Hoewel geen expliciete informatie voorhanden is om aan te nemen dat sprake is van collusie, zou het goed zijn om te analyseren of dit inderdaad niet het geval is en wat de gevolgen zijn voor de marktuitkomsten als hiervan wel sprake is.

In de achtste plaats is hoofdstuk 6 opgezet vanuit het oogpunt om Nederlandse beleidsvragen te kunnen beantwoorden. Gegeven de internationale ontwikkelingen kan het model gebruikt worden om meer algemene vragen te analyseren als gedetailleerdere informatie beschikbaar is van andere EU-lidstaten.

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Elbert worked for the Dutch Ministries of Economic Affairs, Finance, Housing, Land Use and the Environment, Social Affairs and Employment, Health, Welfare and Sports, for the Dutch Parliament, the provinces of North-Holland and Gelderland, the Dutch Waste Processing Association (VVAV), the Netherlands Waterworks Association (VEWIN), Europe Container Terminals and the Dutch waste company AVR. He published five articles in international scientific journals and more than sixty other professional publications (national journals, policy reports, scientific papers and newspaper articles). Dijkgraaf published further about 60 articles in newspapers and magazines which are not directly related to his scientific career.

Professional publications

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