INCINERATION OR DUMPING?

A Social Cost Comparison of Waste Disposal Options

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Research Memorandum 9808

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We like to thank R.H.J.M. Gradus, G. Kadekodi and S. Smulders

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Abstract

In this paper we develop a model allowing a social cost minimizing government to choose between two well-known solid waste disposal techniques, viz. landfilling and incineration. As social cost of waste disposal techniques typically depend on jointly produced outputs such as energy recovery, material recycling and external costs our model allows for these characteristics explicitly. Furthermore, the model is used to reveal the implicit cost of current lexicographical preferences for waste incineration over landfilling using social cost estimates for both waste disposal options in the Netherlands. These estimates show that even in this densely populated country the current policy to expand waste incineration is not optimal from a social cost minimizing perspective. Finally, we examine the robustness of our result for i) changes in the shadow prices for externalities, ii) recycling precommitments and iii) the role of energy recovery. We find that the well-known result that recycling would improve the relative attractiveness of landfilling compared to waste incineration is not true in general. Moreover, energy recovery from landfills seems to be the most important target for any waste policy.
Thus, we find that existing waste hierarchy is not based on a proper social cost evaluation and that current waste policy could better be targeted at both the reduction of environmental externalities as well as the production of energy from landfills.

1. INTRODUCTION

Most developed countries have adopted a hierarchical approach to solid waste management. First of all waste should be reduced, if not recycled, then incinerated waste should go to a landfill and only if nothing else works. Although judgements tend to differ about the order of the first three possibilities, no such difference of opinion exist with respect to landfilling. Landfilling is generally considered to be the worst option mainly because of environmental reasons. In densely populated countries like Singapore, Japan and the Netherlands, this is hardly surprising, but also other countries with more land available tend to prefer incineration, especially in so-called Waste-to-Energy (WTE) facilities (Miranda and Hale, 1997). These facilities even promise a double dividend as they not only reduce final disposal of waste, but also produce electricity and/or heat saving (energy) resources elsewhere.

Interestingly, evidence on the social optimality of this final waste disposal hierarchy is hardly present in the literature. For instance, Keeler and Renkow's (1994) interesting contribution to the waste literature analyzes the desirability of incineration relative to recycling and landfilling only by including their private costs characteristics but simply neglect the role of environmental externalities. Although several authors have stressed the importance of the role of social cost for choosing between different waste disposal options (e.g. Turner, 1995; Jarvinen, 1995; Turner and Brisson, 1995) only recently this topic has become subject of serious analytical and empirical study (Brisson, 1997).

To analyze the choice between final waste disposal options one should account for all important social cost characteristics. These characteristics include energy recovery from WTE plants as well as from landfills, but also their potential for material recovery and their remaining environmental externalities. In this paper we develop a model for a social cost minimizing government which allows for each of these characteristics explicitly. Using this model we are able to examine the economic determinants in any choice between incinerating waste or landfilling. Moreover, using social cost estimates for both options in the Netherlands it turns out that even for this densely populated country landfilling appears to be the optimal solution from a social cost minimizing perspective. The preferences of governments in practice for incineration over landfilling because of its lower environmental cost is not confirmed either. The Dutch case also shows that these cost savings are rather small in a proper environmental impact analysis. Furthermore, we show that our results remain valid within a reasonable confidence margin for the shadow prices used. Finally, we show why the well-known result in the literature that recycling improves the relative attractiveness of landfilling vis-à-vis incineration is not generally true, while energy recovery from landfills seems to be the most important target for any social cost minimizing waste policy.

The paper is organized as follows. We first of all discuss the existing hierarchy of waste disposal options in several developed, particularly European countries. Next, we develop our model for a social cost-minimizing choice between landfilling and incineration. The model shows under which economic conditions landfilling would be inferior to incineration. We then illustrate the model using a point estimate for the Netherlands showing that landfilling is superior to incineration at the margin. To explore the sensitivity of our results for the shadow prices used as well as the different arguments in the current policy debate on waste disposal options, the fifth section illustrates how the choice between incineration and landfilling is governed by differences in marginal valuations of several social cost components. The last section presents some conclusions and discusses further research.

2. EXISTING WASTE HIERARCHIES

In most developed countries a widespread belief in the previously mentioned hierarchy for waste disposal options exists. This belief is often reflected in governmental documents or even in existing environmental law. For instance, the European Union recently confirmed this hierarchy explicitly with reference to the external cost of landfilling:

> "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 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. (EC COM(97) 105, p.3).

Thus the European Commission seems to rely on a rather asymmetric judgement in comparing the different waste strategies. It does include environmental cost for landfilling but only stresses environmental benefits for the other options. Whether private cost should play a role seems to be unimportant, at all.

In contrast, the USA does not have a clear preference of incineration over landfilling. For instance, the Environmental Protection Agency explicitly mentions indifference with respect to both 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)

Nevertheless, the use of energy and residual materials is unquestioned as well:

> "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)

Thus although incineration is not preferred to landfilling in general, reuse of materials and WTE facilities are undisputed to the EPA as well, again, however, without any explicit reference to private cost.

Notwithstanding these stated preferences, existing final waste disposal policies are still dominated by landfilling. On average only 23% of total waste is incinerated in the early nineties, as table I reveals. Only a few countries like Denmark, Belgium, Luxembourg, Japan and France incinerate a large part of their domestic waste. In contrast, Spain, Italy, the United Kingdom and the US have a very low percentage of incinerated waste, while Portugal, Ireland and Greece incinerate no waste at all.

TABLE 1. WASTE INCINERATION CHARACTERISTICS IN SOME DEVELOPED COUNTRIES, 1993

Country	Incinerated domestic waste as a % of total waste	Incineration with energy recovery, as a % of total incinerated	Population density per 1000 ha	Domesticated land as a % of total land
Europe:				
Belgium	52	33	3311	46
Denmark	59	100	1219	66
Germany	26	100	2308	50
Greece	0	0	792	71
France	44	33	1043	56
Ireland	0	0	505	82
Italy	6	na	1966	57
Luxembourg	47	100	na	na
Netherlands	28	78	4502	59
Portugal	0	0	1073	44
Spain	5	na	784	61
UK	10	30	2393	74
Japan	75[1]	na	3319	14
USA	3[1]	na	281	47

Sources: European Countries: Coopers & Lybrand (1996), World Resources Institute (1995).
USA and Japan: Miranda and Hale (1997).
Data are for early nineties.

1. Percentage of municipal waste incinerated with energy recovery.

One of the major reasons mentioned for the dominance of waste incineration in particular countries is their scarcity of land. Interestingly, table 1 shows no clear correlation between population density or domesticated 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 domesticated land. On the other hand France and Belgium incinerate a lot of domestic waste with much lower land scarcity indicators.

Indeed, besides land scarcity other environmental concerns also influence preferences over waste disposal methods (Miranda and Hale, 1997; Menell, 1990). Especially leakage to ground water is supposed to be important causing considerable environmental costs of landfilling. Moreover, methane (CH_4) emissions, one of the principal Greenhouse Gases, are considered a typical landfilling problem. Usually, incinerating waste is thought to provide the solution to these problems as incinerating waste reduces the volume of waste reclaiming fewer land to be reserved for waste disposal sites. Furthermore, energy recovery is seen as another main advantage of incinerating waste (Miranda and Hale, 1997; Vollebergh, 1997). If waste is burned the energy released can be used for electricity production or heating systems. Modern WTE plants can be efficient energy plants especially if combined heat-power (CHP) plants are applied.

Although preferred in theory, WTE plants are not as wide spread as one might expect. Only four EU countries use WTE on a significant scale. Moreover, several countries who incinerate on a comparatively large scale use WTE only partially. Furthermore, it should also be noted that incinerating waste causes externalities as well, varying from incineration ash - which is rather polluted with chemical substances -, toxic substances like dioxins, and all sorts of air pollutants. Therefore a comparative analysis of both waste disposal options should take all social cost components into account, including potential technologies like energy recovery from landfills.

3. MODELING THE CHOICE BETWEEN WASTE DISPOSAL OPTIONS

To analyze the role of economic determinants in choosing between final waste disposal options we develop a simple static model for a government minimizing social cost. The rationale for this assumption is that governments have easier access to information on cost characteristics of different waste disposal programs compared to information on individual preferences, while (social) cost minimization is a necessary condition for welfare maximization anyway.[1] We denote the total amount of waste left for final disposal as W_0, and the quantities disposed of via landfilling and incineration as W_L and W_I respectively.[2]

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1 It is still perfectly possible to test for myopic policy preferences over private and social cost as we show in a later section. These myopic preferences are quite common in the ongoing policy debate on waste disposal options.

2 As we analyze 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.

As noted we consider the case where the government minimizes the overall social cost taking appropriate notice of the jointness characteristics of the different waste disposal methods. By choosing the amount of waste to be landfilled or incinerated the government not only decides over final waste reduction and their respective environmental cost, but also implicitly over the composition of energy production and the amount of material recycling given the existing technologies applied. In other words, an analysis of social cost minimization should take account of the potential of jointly produced useful outputs of each waste disposal possibility. To include both useful outputs into our analysis we simply assume that the government has to meet a given amount of energy and material demand, denoted as E_0 and G_0 respectively. Furthermore, and rather crucial to our analysis, we also account for environmental cost (or benefits) depending on the specific technology of waste disposal facilities as each different combination of functions produces a different set of environmental externalities.[3]

Social cost of *landfilling* depend on the specific combination of both private cost of landfills $L_P(W_L)$ and the environmental costs $L_M(W_L)$ for the jointly produced outputs together. Private cost of landfilling include collection and hauling cost, labor and capital cost for operation and maintenance as well as some savings reflecting precautionary care for leakage in the future. Note also that these private cost 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 cost ($L_P^{'} > 0$ and $L_P^{''} > 0$) as transport cost usually rise if withdrawal areas become larger. Finally, environmental cost related to landfilling depend on the landfill-technology applied. As we do not explicitly model technology choice for landfilling, the environmental cost (savings) are simply assumed to be additive with private cost, 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 cost $I_P(W_I)$ and the environmental cost $I_M(W_I)$ are similar to those of landfilling with the exception of the recovery of materials.[4] Like landfilling incineration contributes to energy production,

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3 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, G accounting for the useful outputs waste reduction, energy production and materials recovery, 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.

4 In contrast to Keeler and Renkow (1994) we do not impose a fixed cost component on incineration plants as the fixed cost components they mention, such as the cost of

$E_I(W_I)$. 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 cost for both energy and materials recovery are included in the private cost 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 $I_P^{''} > 0$ 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 doing so she minimizes 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 recycle materials through its waste disposal system, she 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 either produced 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 either produced as materials recovered from incineration ash or from existing plants for materials production ($G_0 = G + G_L(W_L)$). 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:

$$\begin{aligned} Min\ C = {} & L_P(W_L) + L_M(W_L) + I_P(W_I) + I_M(W_I) + E_P(E) + E_M(E) + \\ & + G_P(G) + G_M(G) + \lambda_1[E_0 - E - E_L(W_L) - E_I(W_I)] + \\ & + \lambda_2[G_0 - G - G_I(W_I)] + \lambda_3[W_0 - W_L - W_I] \end{aligned} \quad \textbf{(1)}$$

Using the Kuhn-Tucker conditions we find as an optimal interior solution (see appendix I for proof):

$$L_P^{'}(W_L) + L_M^{'} - E_{SC,L}^{'} = I_P^{'}(W_I) + I_M^{'}(W_I) - E_{SC,I}^{'} - G_{SC,I}^{'} \quad \textbf{(2)}$$

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planning, siting, contracting and permitting an incineration plant, similarly apply to landfills. There is no reason to assume *a priori* that it would be more difficult to locate an incineration plant compared to a landfill as section II already has clarified.

with:

$$E'_{SC,L} = [E'_P(E) + E'_M(E)]\ E'_L(W_L) \qquad \textbf{(3)}$$

$$E'_{SC,I} = [E'_P(E) + E'_M(E)]\ E'_I(W_I)$$

$$G'_{SC,I} = [G'_P(G) + G'_M(G)]\ G'_I(W_I)$$

This expression has a clear economic meaning. Equation (2) 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. These social cost, however, not only include the private and direct environmental cost of both disposal technologies ($L'_P + L'_M$ vis-à-vis $I'_P + I'_M$), but also the indirect social cost of each method through both the energy and the materials recovery system ($E'_{SC,L}$ vis-à-vis $E'_{SC,I} + G'_{SC,I}$). That is, due to its jointness characteristics landfilling and incineration not only affect the processing of final waste, but also - indirectly - the *private and environmental cost* of energy and materials production. Obviously, if the marginal social cost of landfilling exceed the marginal social cost of incineration ($L'_P + L'_M$ - $E'_{SC,L}$) > ($I'_P + I'_M$ - $E'_{SC,I}$ - $G'_{SC,I}$) for a given range of waste to be processed (0-W_0), the government should incinerate all waste from a social cost perspective and vice versa.[5]

Thus the choice how much waste should be treated by what option also depends on the social opportunity costs per unit recovered energy or material. Indeed, how much waste is incinerated or goes to landfills depends on the location of the total social cost of both options (see figure 1). As is immediately clear from the marginal conditions - given the linearity of both the energy and material recovery functions and the absence of materials recovery in the case of landfilling - energy and/or materials productivity ($E_L(W_L)$ vis-à-vis $E_I(W_I) + G_I(W_I)$) play a crucial role in the total opportunity cost of either waste disposal option. For instance, a technological improvement leading up to a higher energy productivity of some incineration plant directly affects the location of the overall cost of incineration, and therefore also how much waste is treated by each option separately. There is no such effect through prices as the relative social opportunity cost remains unaffected by the amount of energy or materials recovered.

The results obtained reflect a much more general notion of how to choose between landfilling and waste incineration as is usually perceived in both theory and policy. What makes

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5 Given the non-negativity of the choice variables $W_L = 0$ simply implies that $W_I > 0$ (given the constraint $W_0 = W_L + W_I$). By the same reasoning $W_I = 0$ implies $W_L > 0$.

our findings interesting is that they allow for all relevant social cost related to the jointness characteristic of any final waste treatment system. The social cost of landfilling or incineration not only depend on the private and external cost of the landfills and incineration plants themselves but also on the opportunity costs of the recovery production functions. Thus Keeler and Renkow's suggestion (1994, p.210) that waste incineration would have become more favorable in recent years due to the growing attention for energy recovery is wrong. Their suggestion - as well as from many others - takes insufficient notice of the opportunity cost characteristics of available waste disposal programs.

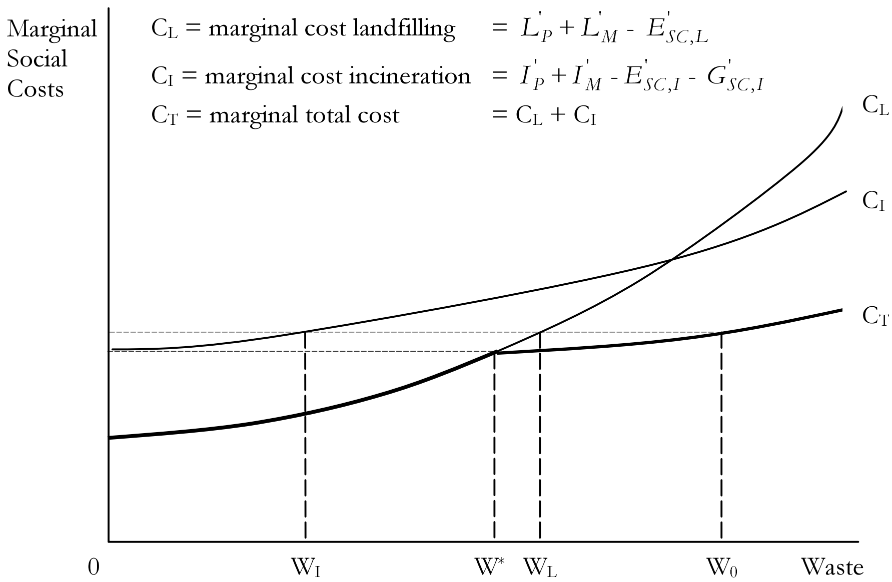

Figure 1. Solution model

Revenues from energy recovery do not decrease *gross* incineration costs - it is even more likely that they rise somewhat due to the need for some additional capital equipment -, but only *net* incineration costs. These private revenues only reflect a net transfer between different consumers, viz. electricity and final waste consumers, usually affected by various institutional factors (Vollebergh, 1997). It is more appropriate therefore to measure these costs by their social value (both private and environmental costs) in the energy system (see also section V). Moreover, it is totally inadequate not to include such energy recovery potential into the comparative analysis for landfilling, as energy recovery can be substantial at landfills as well.

The additivity assumptions imply that we exclude endogenous interactions between incineration and landfilling such as the disposal of burning ash - which is usually quite polluted material - from a waste incineration plant to a landfill.[6] Nevertheless, differences in technology can still be interpreted as changes in the exogenous parameters of the model. The same is true for interactions with recycling programs. Indeed, as Keeler and Renkow (1994) have argued, recycling indirectly influences the final waste disposal choice: with more recycling of materials with high energy content a lower energy production would result which would make landfilling more attractive. However, as our model shows, different qualities of waste being supplied also directly affect the slope of the social costs curves as both the energy and material recovery function are affected as well. Thus, the overall effect depends on much more complicated interactions between the different waste streams including the environmental and opportunity cost which affect the overall cost of both disposal methods.

One example might illustrate this. Assume, the government considers a recycling program for certain materials. Two effects are important according to our model.[7] First, the energy efficiency of incineration (E_I'), depends on the relative energy content of the recycled materials, while the energy productivity of landfilling (E_L') does not directly depend on the energy content of materials but on methane emissions. Second, the recycling program also effects the (physical) externalities, changing environmental costs of both disposal methods (I_M') and L_M'). Thus, our equation (2) shows that recycling might improve the relative attractiveness of incineration only if the change in the environmental disposal costs and social opportunity costs of incineration outweighs the change in these cost for landfilling $\Delta I_M'$ - $\Delta\ E_M'(E_P' + E_M') < \Delta L_M'$ - $\Delta E'\ \ (E_P' + E_M')$.

4. CHOOSING BETWEEN WASTE DISPOSAL OPTIONS IN THE NETHERLANDS

The previous section revealed how a rational government could minimize social cost of its final waste treatment program. Obviously, a large set of data is necessary for an empirical social cost evaluation of final waste treatment. As neither cross-sectional data nor time profiles of final waste production functions are currently available, a proper social evaluation is hardly possible in practice. As a first approximation we consider a point estimate of the social cost minimizing

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6 In a more general framework one would like to include the restriction $W_L = W_{L1} + \alpha W_I$ with $\alpha > 0$ allowing for interactions between different waste flows (see Keeler and Renkow, 1994). Note, however, that this particular interaction is still accounted for exogenously in our analysis as the cost of ash disposal is included in the environmental cost related to incineration.

7 Both private costs and environmental effects of the recycling program itself are not relevant because they are the same for both landfilling and incineration.

problem for the Netherlands for the year 1997.

The Netherlands provides an interesting case study for at least two reasons.[8] First, almost 40% of the waste potential for incineration or 2.8 million tons of waste was actually burned in eleven incineration plants in 1995, a percentage which is likely to grow to 85% in 1997 and up to 100% before 2000. In addition, electricity and heat production from waste incineration are projected to grow by 150% and 250% respectively between 1995 and 2000. Second, the Netherlands is a densely populated country which makes one belief incineration to be the optimal solution for final waste disposal from a social perspective. However, as this section explains in detail, the contrary seems to be the case.

4.1 Private cost

Private cost estimates for waste *incineration* (I'_P (W_I)) are obtained from a recent study of the Dutch Central Waste Agency on the existing waste incineration plants (AOO, 1995).[9] Their figures include current (large) capital investments in both capacity expansion as well as abatement technologies (mainly flue gas) required by environmental regulation such as the EC directives on harmonization of technical standards from 1989. We use their estimate of gross *average* private cost of \$ 134 per ton of waste.[10]

Note that these average cost also include current investment cost in expanding WTE facilities in accordance with the schedule mentioned before. In 1997 the number of WTE plants in the Netherlands is 11 (out of 12) cogenerating 2450 GWh electricity and 1296 GWh thermal heat (CE, 1996, pp.2-13). These figures imply that the energy productivity (E'_I) of the incineration plants in the Netherlands is 567 kWh_e electricity on average out of one ton of waste and 315 kWh_{th} thermal heat per ton waste. The material productivity of these plants (G'_I) is derived from the same study. It is estimated that one ton of ash from incinerated waste produces 1.6 kg aluminum and 34 kg ferro in 1997.

Private costs for landfilling (L'_P) are much harder to obtain. Even no comparable figures are currently available for disposal sites. Therefore we use estimates from a study calculating

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8 See Vollebergh (1997) for a description of the institutional details on final waste management in the Netherlands with special emphasis on the energy potential of WTE plants.

9 Obviously we use average private cost to estimate long run marginal cost. If constant returns to scale would apply in this industry the cost estimates might even be interpreted as short run marginal cost.

10 According to a cross-section for the existing plants overall cost vary between \$ 115 and \$ 173 per ton incinerated waste with, on average, 53% capital and 47% operational cost.

private cost of a landfill with best available technology in the Netherlands (VROM, 1992). Break-even costs for a landfill employing the best techniques as required by recent Dutch legislation against leaking (Stortbesluit Bodembescherming, 1995) and including a cost estimate for energy recovery were estimated at $ 28 per ton waste on average.[11] As differences in scale, prices for land use, etc. account for relative price differentials across the country we also include a typical worst case with $ 42 per ton. It is furthermore assumed that this standard dumping site produces 122 kWh electricity from landfill gas ($E_L^{'}$).

Private (opportunity) cost of energy production ($E_L^{'}$) are simply obtained from private cost estimations for gas and coal fired power plants as fossil fuel inputs account for 92% of the total electricity produced in the Netherlands. On average electricity production costs are $ 3.8 cents per kWh which amounts to an opportunity cost of $ 22 per ton incinerated waste (assuming current electricity productivity of the average WTE-plant). As the energy productivity of landfilling is much lower this amounts to $ 5 per ton landfilled waste. Finally, private (opportunity) cost of materials production ($G_L^{'}$) are simply obtained from world market prices for those materials having a positive recycling price, viz. $ 1.55 per kg aluminum and $ 0.03 per kg ferro (Eurostat, 1992).

TABLE 2. PRIVATE COST ESTIMATES LANDFILLING AND INCINERATION ($/TON)

	Landfilling		Incineration	
	Variable	Cost estimate	Variable	Cost estimate
Gross Private Costs:	$L_L^{'}$	42	$I_P^{'}$	134
Private Opportunity Costs:				
- Energy function	$E_P^{'}$ $E_P^{'}$	5	$E_P^{'}$ $E_I^{'}$	22
- Materials function	-	0 -	$G_P^{'}$ $G_I^{'}$	3 -
Net Private Costs:		37		109

Sources: Gross Private Costs estimates taken from AOO (1992) and CE (1996); private opportunity costs are taken from Eurostat (1992).

Table 2 summarizes the gross private cost estimates for both final disposal methods.[12]

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11 As the same (environmental) regulation applies as in 1992 (with low inflation in between), this calculation is still a good approximation for current overall average costs.

12 Compared to other countries Dutch private costs for incineration and landfilling are very high. For example average private costs in the USA are $ 31 for landfilling and $ 89 for incineration (Miranda and Hale, 1997). Private costs for energy are on a European low level in the Netherlands.

Furthermore it calculates net private cost taking appropriate notice of the jointness characteristics of both waste disposal methods as described before. The result is rather straightforward: landfilling is much less expensive as incineration even if appropriate notice is taken of the differences between both options in terms of their private opportunity cost. Although waste incineration contributes to considerable private cost savings elsewhere in the economy, these savings cannot offset the much higher private cost of the incineration plant. The next subsection explores to what extent environmental costs might change the overall picture.

4.2 Environmental and social costs

Both landfills and incineration plants cause substantial negative environmental externalities, although they differ to what extent they contribute to air, water and soil pollution as well as land use. Again, however, these externalities cannot be attributed to the waste function alone as producing energy and materials recovery produces savings of emissions elsewhere in the economy. Using a recent, well-documented comparative engineering study for both waste disposal options as well as their energy and material productivity (CE, 1996), we were able to calculate the different environmental costs categories as required according to the model of the previous section. These estimates are consistent with the private cost calculations as the energy and material productivity are thus taken from the same source.[13]

CE (1996) produces comparable physical and money values for the environmental externalities in the following way. First of all, they present physical environmental impacts for a whole set of life cycle or LCA-emissions related to landfilling, incineration, energy and material production. Second, environmental costs are calculated weighing these impact with a set of shadow prices for environmental damage. 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 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 the consumption of land. The impacts for landfilling are based on best available techniques against leaking currently available, which is also in accordance with the private cost estimate for landfills. Like the private costs the estimates for incineration, material and energy externalities are based on averages for existing plants. For the estimation of energy externalities not only the direct generation emissions are measured, but also emissions related to extraction and transport.

Shadow prices used to calculate environmental costs are based on local abatement cost calculations for Dutch environmental policy goals for 2010. These policy goals are usually

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13 See appendix II for more details.

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), which can be assumed to reflect current minimum willingness to pay for emission reduction in the Netherlands. They are available for a rather large set of environmental impacts as these aspiration levels include most environmental policy issues. The study also includes a scarcity evaluation of land use. However, this evaluation is based on the current price of farm land ($ 3 per m^2), which clearly underestimates current social scarcity values of land. Therefore we include a more appropriate estimate based on the average price of house building land ($ 286 per m^2).

TABLE 3. ENVIRONMENTAL COST LANDFILLING AND INCINERATION ($/TON)

	Landfilling		Incineration	
	Variable	Cost	Variable	Cost
Environmental Cost:				
- emissions to air		7		22
- emissions to water		0		0
- chemical waste		3		36
- land use		23 +		0 +
Gross Environmental Costs:	L'_M	33	I'_M	58
Environmental Opportunity Cost:				
- energy function	$E'_M\ E'_L$	5	$E'_L\ E'_I$	28
- materials function	-	0 -	$G'\ G'_I$	7 -
Net Environmental Cost:		28		23

The overall environmental cost of the different functions of the waste disposal options are summarized in table 3. Although land consumption by landfilling results in rather high costs of this externality per ton waste, the much larger emissions to air and disposal of chemical waste result in higher gross environmental costs for incineration as in both cases emissions to water are negligible (which in the case of landfilling depends on the best available technique assumed). However, from our model it is clear that not only direct externalities are important for deciding on the optimal waste disposal option, but also the environmental opportunity costs related to the energy and materials functions. Low energy recovery and no material recycling result for landfilling in only $ 5 externality savings per ton waste, whereas incineration saves $ 28 per ton in the energy production system and an additional $ 7 per ton in the material production system. Thus *net* environmental costs are somewhat lower for incineration compared to landfilling, which is in accordance with current policy views on the relative environmental attractiveness of both systems.[14]

Combining both private and environmental cost we find that social cost for waste incineration are $ 132 per ton waste compared to only $ 65 per ton landfilled waste. Even though the environmental cost of incineration are indeed lower compared to landfilling, their private cost difference more than compensates this relatively small environmental cost difference. Whether this result shows irrational behavior from the part of the Dutch government is the subject of the next section.

5. ARE CURRENT WASTE DISPOSAL PREFERENCES IRRATIONAL?

The point estimate for the Netherlands suggest that it is not rational to expand waste incineration as is currently the policy of the Dutch government. Moreover, because environmental cost of land use are of major importance in the case of this densely populated country, one would expect landfilling to be even more attractive in other, less populated countries, such as Denmark or France (see section II). In this section we examine whether this conclusion remains valid in the following cases: i) changes in environmental preferences; ii) recycling precommitments; iii) the effects of differences in environmental policies.

5.1 Changes in environmental preferences

Another important issue with respect to the relative attractiveness of waste incineration versus landfilling are the environmental weights used in our point estimate for the Netherlands. We used shadow prices for externalities typically based on local (Dutch) environmental policy goals and they were mainly responsible for the relatively small environmental cost difference compared to the private cots difference of landfilling vis-à-vis incineration. What would happen to our result that landfilling is preferable at the margin if the shadow prices for environmental externalities change?

One method to explore the sensitivity of our results is to consider the case where the environmental cost difference would make incineration relatively more attractive compared to landfilling. That environmental cost might make such a difference can be explained by the

14 Miranda and Hale (1997) find that the external costs for landfilling for a regular US landfill site are $ 21 per ton waste, whereas incineration would be responsible for $ 9 to $ 25 environmental cost per ton depending on the technology applied. However, these estimates do not include space intensity and effects of chemical waste. Furthermore, their estimates for landfilling are dominated by methane emissions, which were assumed to be very low in the Dutch case because landfill gas was assumed to be used as fuel for electricity production (which is not common practice now).

fact that these cost become more important as a percentage of total social costs even though the relative net environmental cost $\{ L'_M - E'_M \; E'_L \} \, / \, \{ I'_M - E'_M \; E'_I \; - \; G'_M \; G'_I \}$ are constant if all prices are multiplied with a fixed factor. Using the figures for the Dutch case, incineration will only become equally attractive as landfilling at the margin if the shadow prices for the externalities are 16.3 times higher than currently assumed.

Another method is to look at the relative strength of the shadow prices used for the different externalities observed for each final disposal method. Again using the figures for the Dutch case, incineration will only become equally attractive as landfilling at the margin if the shadow prices for climate change emissions (CO_2 and CH_4) are 24.0 times higher than currently assumed. For acid rain emissions (SO_2 and NO_X) this figure is 12.0 and for land use 4.0. As table 4 makes clear, shadow prices are very high in these cases compared to our initial prices and to prices used in other studies. Both for emissions to water and the disposal of chemical waste no turning point is present as higher shadow prices increase the difference in social costs between landfilling and incineration.

TABLE 4. SHADOW PRICES IN THIS STUDY COMPARED WITH OTHER STUDIES

	(Maximum) shadow price in dollar per ton emissions of:			
	CO_2	CH_4	SO_2	NO_x
Our study				
- initial (section IV)	43	478	5,920	4,144
- turning point (section V)	1,042	11,465	71,040	49,728
Nordhaus (1991)	241			
Nordhaus (1992)	19			
Peck & Teisberg (1992)	44			
CEC (1992)			19,800	26,400
Fankhauser (1992)	174	777		
CSERGE (1993)			746	574
ORNL (1995)			1,500	6,500

We can conclude that our result that the current waste disposal hierarchy is not optimal from a social point of view is rather robust. Obviously, it is rather unlikely that shadow prices that would reverse this result fall within the uncertainty range of the current estimates.

5.2 Recycling precommitments

As argued at the end of section III, the assumption that recycling would improve the relative attractiveness of landfilling should be subject to a much richer assessment as is now common. Indeed, calculations for the Dutch case confirm this conclusion.

Assume, first of all, a 50% a recycling program for synthetics. Given the average energy content of synthetics and the percentage of synthetics in current waste composition, energy productivity of the average Dutch waste incineration plant ($E_I^{'}$) would be 16% lower. Given the social opportunity costs of energy, the net social costs for incineration ($E_I^{'}$ ($E_P^{'} + E_M^{'}$)) rise $ 8 per ton waste. Energy production on the landfill site does not depend on the energy content of waste, but on the level of organic carbon (CE, 1996). As synthetics do not contribute to this carbon intensity, the effects of the recycling program on the energy productivity of landfilling ($E_L^{'}$) are equal to zero. As the environmental costs of landfilling are dominated by land use and emissions to air, the effects of the recycling of synthetics on these costs will be very small and is here simply assumed to be zero as well. Therefore, we conclude that recycling of synthetics can make incineration more attractive c.p. if $\Delta\ I_M^{'} + 8 < 0$. However, the environmental cost effect ($\Delta\ I_M^{'}$) is nearly $ 0 per ton incinerated waste. Thus, recycling of synthetics makes landfilling indeed more attractive.

Consider now another type of recycling program, a 100% recycling of paper and cardboard. As the energy content of paper and cardboard is just on the average, the effect on the energy productivity of incineration is only a 1% increase. For the energy productivity of the landfill the story is now completely different. As the organic carbon intensity of paper and cardboard is very high, this recycling program results in a decrease of the production of landfill gas per ton of waste. As less landfill gas can be used for the production of energy the energy productivity will decline with 43%. As the changes in energy productivity correspond to an effect on the opportunity cost of $ 0 per ton waste for incineration and $ 5 per ton for landfilling, equation (2) learns that recycling of paper and cardboard makes incineration more attractive if $\Delta\ I_M^{'} - 0 < \Delta\ L_M^{'} + 5$. The environmental cost effect for landfilling ($\Delta\ L_M^{'}$) is negative and in the order of $ 3 per ton waste, due to the decrease in landfill gas and thus lower emissions to air. In contrast to synthetics, paper and cardboard recycling programs make incineration more attractive, but only if $\Delta\ I_M^{'} < 2$.

However, both recycling programs have no influence on the preferred waste disposal option from a social point of view. Compared to the social cost difference of $ 67 per ton of waste, effects of these recycling programs are rather small.

5.3 The effects of different environmental policies

Intuitively it is clear that, in practice, a relation exists between external and private costs for different disposal techniques. In terms of our model lower environmental limits for a disposal option makes the other option more attractive as long as $\Delta\ I_P^{'} + \Delta\ I_P^{'} > 0$ for incineration and $\Delta\ L_P^{'} + \Delta\ L_P^{'} > 0$ for landfilling. A priori it is not clear what the exact relation is between higher environmental and lower private costs. In fact, a case by case approach is needed to evaluate the effects of different technologies. Based on a regular USA landfill (Miranda and Hale, 1997) the estimated possible decrease in environmental costs is not less than $ 29 if environmental limits would be at the Dutch level. As private costs are $ 14 at the minimum in the USA, lower environmental limits for landfilling would probably make incineration more attractive from a social perspective.

As is shown by our model, not only new environmental limits for landfilling or incineration but also new limits for energy and material production influences the optimal waste disposal strategy. Less tight regulation of emissions in the energy and material production plants makes landfilling more attractive as long as the environmental costs increase is less than the private costs decrease, ($\Delta\ E_P^{'} + \Delta\ E_M^{'} < 0$ and $\Delta\ G_M^{'} + \Delta\ G_M^{'} < 0$) and the energy productivity of incineration is higher than for landfilling ($E_I^{'} > E_L^{'}$).

Less tight emission limits in several developed countries might be responsible for the general believe that landfilling is a worse option compared to incineration. This can be illustrated for the use of landfill gas. In most countries energy production from landfill gas is not common. In the USA only 6% of landfill sites produce electricity. If no electricity is generated ($E_L^{'} = 0$) methane emissions are more than 8 times as high. Furthermore, more electricity has to be generated in the energy production plants. Therefore, no electricity generation would result in \$ 20 higher gross environmental costs ($L_M^{'}$) and \$ 26 higher net environmental costs ($L_M^{'} - E_M^{'}\ E_M^{'}$). Of course private landfilling costs ($L_P^{'}$) would be less than in the original case, but probably this reduction is not more than \$ 5 per ton waste.[15] The net social loss if no electricity is produced with landfill gas is thus at least \$ 21 per ton. Therefore, electricity production is largely responsible for the relative attractiveness of landfilling.

For incineration plants the net social loss if no energy is produced is much smaller compared to dumping sites as energy generation does not result in less methane emission, $\Delta I_M^{'} = 0$. For incineration plants no energy generation only results in a loss of produced heat which has no direct environmental effect. However, given the high environmental opportunity costs ($E_M^{'}$), \$ 28 per ton of waste, energy generation will result in a clear and significant social gain and will thus make incineration more attractive.

6. CONCLUSIONS

In most developed countries current waste policy aims to expand incineration, especially WTE-plants. First of all, incineration would be preferable to landfilling because of the considerable

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15 Because energy production with landfill gas is not compulsory in the Netherlands owners will only invest in energy production if revenues are not less than private costs. Extra private costs will be lower than the opportunity costs because no subsidies are given. In model terms: $\Delta\ L_P^{'} \leq E_L^{'}\ E_L^{'}$.

amount of land consumption for landfilling whereas leakage effects add to environmental costs as well. Second, it is generally thought that energy recovery from incineration plants improves its relative attractiveness as fewer energy resources would be necessary to meet a given amount of energy demand. From a social cost minimizing perspective proper, however, we have been able to show that these arguments - although valid in some circumstances - do not hold in general.

Indeed, if waste incineration is expanded anyway, WTE will probably always be a Pareto improving policy as energy recovery is a joint product with only a small additional investment in technology. When, in contrast, the government is interested in minimizing not only the direct environmental cost of waste reduction, but also the opportunity cost of both energy and material recovery including the alternative of landfilling, landfilling with energy recovery seems to be a very serious alternative. For the Netherlands this turns out to be the optimal policy at the margin. Whether these empirical results generalize is not easy to say as the calculations for the Dutch case start from currently best available techniques for both options. In practice landfills do not always recover energy while leaking can be a serious problem, especially for older landfill sites. Nevertheless, our model is helpful for evaluating any final waste disposal technique and has as a main advantage over previous approaches that also the indirect effects are included in such evaluations.

APPENDIX I

We have the following minimand expression:

$$\begin{aligned} Min\ C = {} & L_P(W_L) + L_M(W_L) + I_P(W_I) + I_M(W_I) + E_P(E) + E_M(E) + \\ & + G_P(G) + G_M(G) + \lambda_1[E_0 - E - E_L(W_L) - E_I(W_I)] + \\ & + \lambda_2[G_0 - G - G_I(W_I)] + \lambda_3[W_0 - W_L - W_I] \end{aligned} \qquad \textbf{(I-1)}$$

The Kuhn-Tucker conditions are:

$$\frac{\partial C}{\partial W_L} = L_P'(W_L) + L_M'(W_L) - \lambda_1 E_L'(W_L) - \lambda_3 \geq 0 \qquad \textbf{(I–2)}$$

$$\frac{\partial C}{\partial W_I} = I_P'(W_I) + I_M'(W_I) - \lambda_1 E_I'(W_I) - \lambda_2 G_I'(W_I) - \lambda_3 \geq 0 \qquad \textbf{(I–3)}$$

$$\frac{\partial C}{\partial E} = E_P'(E) + E_M'(E) - \lambda_1 \geq 0 \qquad \textbf{(I–4)}$$

$$\frac{\partial C}{\partial G} = G_P'(G) + G_M'(G) - \lambda_2 \geq 0 \qquad \textbf{(I–5)}$$

$$\frac{\partial C}{\partial \lambda_1} = E_0 - E - E_L(W_L) - E_I(W_I) \geq 0 \qquad \textbf{(I–6)}$$

$$\frac{\partial C}{\partial \lambda_2} = G_0 - G - G_I(W_I) \geq 0 \qquad \textbf{(I–7)}$$

$$\frac{\partial C}{\partial \lambda_3} = W_0 - W_L - W_I \geq 0 \qquad \textbf{(I–8)}$$

plus the nonnegativity and complementary-slackness conditions.[16]

If $W_L > 0$ and $W_I > 0$ the nonnegativity constraints are not binding. It follows that the partial derivatives (I-2 to I-8) must be strictly equal to zero. The solution now becomes:

16 See for an outstanding explanation of non-linear programming Chiang (1984).

$$(I-2)' \quad L_P^{'}(W_L) + L_M^{'}(W_L) - \lambda_1 E_L^{'}(W_L) - \lambda_3 = 0$$

$$(I-3)' \quad I_P^{'}(W_I) + I_M^{'}(W_I) - \lambda_1 E_I^{'}(W_I) - \lambda_2 G_I^{'}(W_I) - \lambda_3 = 0$$

$$(I-4)' \quad E_P^{'}(E) + E_M^{'}(E) - \lambda_1 = 0$$

$$(I-5)' \quad G_P^{'}(G) + G_M^{'}(G) - \lambda_2 = 0$$

$$(I-4)'' \quad \lambda_1 = E_P^{'}(E) + E_M^{'}(E)$$

$$(I-5)'' \quad \lambda_2 = G_P^{'}(E) + G_M^{'}(E)$$

$$(I-4)'' \rightarrow (I-2)' \rightarrow (I-2)''$$

$$L_P^{'}(W_L) + L_M^{'}(W_L) - [E_P^{'}(E) + E_M^{'}(E)]\, E_L^{'}(W_L) = \lambda_3$$

$$(I-5)'' \rightarrow (I-3)' \rightarrow (I-3)''$$

$$I_P^{'}(W_I) + I_M^{'}(W_I) - [E_P^{'}(E) + E_M^{'}(E)]\, E_I^{'}(W_I) - [G_P^{'}(G) + G_M^{'}(G)]\, G_I^{'}(W_I) = \lambda_3$$

(I-2)'' = (I-3)'' gives exactly equation (2) in the paper.

As the objective function (I-1) is differentiable and convex, the constraint functions are linear, this condition is necessary-and-sufficient for a minimum.

APPENDIX II

In this appendix we show how we arrived at our external cost estimates. As not otherwise indicated figures are from CE (1996). CE (1996) produces money values for the environmental externalities in the following way.

First, externality sources are defined; emissions to air, 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 II-1).

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 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 the consumption of land.

Third, emissions are taken from the national Environmental Effects Reportings, 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 II-1 to II-12.

Fourth, shadow prices for the different externalities are derived, 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), which can be assumed to reflect current minimum willingness to pay for emission reduction in the Netherlands. They are available for a rather large set of environmental impacts as these aspiration levels include most environmental policy issues. The study also includes a scarcity evaluation of land use. However, this evaluation is based on the current price of farm land (\$ 3 per m^2), which clearly underestimates current social scarcity values of land. Therefore we include a more appropriate estimate based on the average price of house building land (\$ 286 per m^2). Shadow prices are given in table II-13 for the most important (in terms of environmental costs) externalities.

Last, physical emissions are multiplied by the shadowprices to arrive at environmental costs (see tables II-1 to II-12).

TABLE II-1 LANDFILL GAS PRODUCTION

Landfill gas	Total production[1]	148	m^3 per ton per year
	- Exploitation[2]	78	m^3 per ton per year
	- Flaring	36	m^3 per ton per year
	- Emissions	34	m^3 per ton per year

1 Total production is estimated with the following formulae: Gas = $1.86 * \alpha * CO * k1 * e^{-k1*t}$
With $\alpha = 0.58$, CO = 143, k1 = 0.094.

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

TABLE II-2 ENERGY PRODUCTION FROM LANDFILL GAS

1	Exploited landfill gas	78	m^3/ton	see table II-1
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 II-3 LANDFILLING LAND USE; PHYSICAL VALUES, PRICES AND COSTS

	Quantity	Price (\$ per m^2)	Costs (\$ 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	285.71	22.50

TABLE II-4 LANDFILLING: EXTERNALITY PHYSICAL VALUES, PRICES AND COSTS

	Emissions to air, water and production of chemical waste in kg per ton waste				Price in $ per kg	Cost[1]
	Direct	Flaring	Gasmotor	Total	Total	Total
Air						
CO_2	0.00	0.00	0.00	0.00	0.04	0.00
CH_4	13.30	0.00	0.00	13.30	0.48	6.35
SO_2	0.00	0.00	0.00	0.00	5.92	0.00
NO_x	0.00	0.01	0.23	0.24	4.14	1.01
Water						
As	4.8E-06			4.8E-06	0.33	0.00
Cd	1.44E-07			1.44E-07	0.00	0.00
Cr	9E-06			9E-06	0.01	0.00
Cu	2.4E-06			2.4E-06	0.01	0.00
Ni	6E-06			6E-06	0.00	0.00
Pb	2.7E-06			2.7E-06	0.23	0.00
Hg	9E-07			9E-07	0.74	0.00
Chem. Waste	2			2	1.66	3.31

1. Environmental costs in $ per ton waste.

TABLE II-5 HEATING VALUE WASTE

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

TABLE II-6 ENERGY AND MATERIALS RECOVERY INCINERATION

1	Heating value	10364	MJ/ton	see table II-5
2	Energy contents	2851	kWh/ton	1 kWh = 3.59 MJ
3	Electrical conversion yield	19.9	%	given
4	Delivered electricity	567	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 ferro	34	kg/ton	given
8	Recycled aluminum	1.6	kg/ton	given

TABLE II-7 INCINERATION; EXTERNALITY PHYSICAL VALUES, PRICES AND COSTS

	Direct emissions in kg per ton waste	Price in $ per kg	Environmental costs in $ per ton waste
Air			
CO_2	447.89	0.04	19.45
CH_4	0.03	0.48	0.01
SO_2	0.20	5.92	1.21
NO_x	0.25	4.14	1.05
Water			
As	5E-08	0.33	0.00
Cd	5.2E-06	0.00	0.00
Cr	4.5E-06	0.01	0.00
Cu	5.5E-06	0.01	0.00
Ni	2.1E-05	0.00	0.00
Pb	1.2E-05	0.23	0.00
Hg	7.2E-08	0.74	0.00
Chemical waste	21.80	1.66	36.13

TABLE II-8 ENVIRONMENTAL OPPORTUNITY COSTS EMISSIONS TO AIR ENERGY FUNCTION; INCINERATION

	Electricity g/kWh	Extra heat[1] g/kWh	Winning, generation, transport g/kWh	Extra CaO g/kWh	Total[2] kg/ton	Price $/kg	Costs $/ton waste
CO_2	503.03	105.50	64.86	3.39	383.73	0.04	16.67
CH_4	0.00	0.00	3.28	0.01	1.87	0.48	0.89
SO_2	0.19	0.00	0.23	0.00	0.24	5.92	1.43
NO_x	0.62	0.00	0.08	0.00	0.40	4.14	1.65

1. As the average incinerator produces relative more heat than the average energy plant, a heat production boiler is added to make the two energy producers comparable.
2. Assumed 1 ton waste produces 567 kWh electricity and 299 kWh heat.

TABLE II-9 ENVIRONMENTAL OPPORTUNITY COSTS CHEMICAL WASTE ENERGY FUNCTION; INCINERATION

	Production g/kWh	Re-use %	Total[1] kg/ton	Price $ per kg	Environmental costs $ per ton waste
Bottom ash	15.3	50	4.34	1.66	7.19
F1y ash	14.4	100	0.00	1.66	0.00
Plaster	6.75	100	0.00	1.66	0.00
Mine waste	87.8	0	49.78	0.00	0.00
Nuclear waste	0.021	0	0.01	1.66	0.02

1. Assumed 1 ton waste produces 567 kWh electricity and 299 kWh heat.

TABLE II-10 ENVIRONMENTAL OPPORTUNITY COSTS EMISSIONS TO AIR ENERGY FUNCTION: LANDFILLING

	Electricity g/kWh	Winning, generation, transport g/kWh	Extra CaO g/kWh	Total[1] kg/ton	Price $/kg	Costs $/ton waste
CO_2	503.03	64.86	3.39	69.70	0.04	3.03
CH_4	0.00	3.28	0.01	0.40	0.48	0.19
SO_2	0.19	0.23	0.00	0.05	5.92	0.31
NO_x	0.62	0.08	0.00	0.09	4.14	0.36

1. Assumed 1 ton waste produces 122 kWh electricity and 0 kWh heat.

TABLE II-11 ENVIRONMENTAL OPPORTUNITY COSTS CHEMICAL WASTE ENERGY FUNCTION: LANDFILLING

	Production g/kWh	Re-use %	Total[1] kg/ton	Price $ per kg	Environmental costs $ per ton waste
Bottom ash	15.3	50	0.93	1.66	1.55
F1y ash	14.4	100	0.00	1.66	0.00
Plaster	6.75	100	0.00	1.66	0.00
Mine waste	87.8	0	10.71	0.00	0.00
Nuclear waste	0.021	0	0.00	1.66	0.00

1. Assumed 1 ton waste produces 122 kWh electricity and 0 kWh heat.

TABLE II-12 ENVIRONMENTAL OPPORTUNITY COST EMISSIONS TO AIR MATERIALS FUNCTION: INCINERATION

	Total[1] kg/ton	Price $/kg	Costs $/ton waste
CO_2	51.01	0.04	2.22
CH_4	0.11	0.48	0.05
SO_2	0.79	5.92	4.67
NO_x	0.08	4.14	0.31

1. Assumed 1 ton waste produces 34 kg ferro and 1.6 kg aluminium

TABLE II-13 SHADOW PRICES EXTERNALITIES

	Price in $ per kg
Air	
CO_2	0.04
CH_4	0.48
SO_2	5.92
NO_x	4.14
Water	
As	0.33
Cd	0.00
Cr	0.01
Cu	0.01
Ni	0.00
Pb	0.23
Hg	0.74
Waste	
Chemical	1.66
Bottom ash	1.66
Fly ash	1.66
Plaster	1.66
Mine	0.00
Nuclear	1.66
Land-use	286

APPENDIX III

In this appendix we explain our estimates for: i) changes in environmental preferences; ii) a recycling program for synthetics and a recycling program for paper and cardboard.

i) Changes in environmental preferences

We calculate the shadow prices for the case where incineration will become equally attractive as landfilling at the margin, given the physical environmental efffects, using an Excel procedure.

iia) Recycling program synthetics

With a 50% recycling program of synthetics, the loss in energy contents is 16.2% (see table III-1, compare table II-5). As the effect on the net social cost for incineration can be calculated by $E_I^{'}\,(E_P^{'} + E_M^{'})$ and $E_P^{'}$ and $E_M^{'}$ are not effected, the effect on net social cost can be calculated by multiplying the loss in energy contents with the net social cost per ton with no recycling program (50$ per ton, see table 2 and 3). The effect of the 50% recycling program of synthetics is thus 16.2% of 50$ = $ 8 per ton incinerated waste.

TABLE III-1 HEATING VALUE INCINERATED WASTE, 50% RECYCLING SYNTHETICS

		Heating value (GJ/ton)	
	Part in total waste (%)	Component	Total
Paper and cardboard	24.4	10	2.4
Wood	7.4	14	1.0
Synthetics	7.3	33	2.4
Ferro	5.8		
Non-ferro	0.8		
Glass	3.0		
Organic	33.3	3	1.0
Rocky	0.9		
Other combustible	17.3	10	1.7
Total	100		8.6

Above the effects trough the change in energy production, emissions will change because of the change in waste composition. However, the change in environmental cost of incineration due to this change is nearly zero (see below).

The effects on CO_2 emissions are

- part synthetics in total waste before program * emissions per ton

0.136 * 350 = 47.60

- part other waste in total waste before program * emissions per ton

0.864 * 464 = 401.19

Total emissions CO_2 per ton waste before program

447.89

- part synthetics in total waste after program * emissions per ton

0.073 * 350 = 25.55

- part other waste in total waste after program * emissions per ton

0.927 * 464 = 430.13

Total emissions CO_2 per ton waste after program

455.68

Difference in CO_2 emissions per ton waste

7.79

Price per kg CO_2 in $ 0.04

Total effect program on cost per ton waste due to difference in CO_2 emissions

$ 0.31

The effects on SO_2 emissions are

- part synthetics in total waste before program * emissions per ton

0.136 * 0.370 = 0.05

- part other waste in total waste before program * emissions per ton

0.864 * 0.174 = 0.15

Total emissions SO_2 per ton waste before program

0.20

- part synthetics in total waste after program * emissions per ton

0.073 * 0.370 = 0.16

- part other waste in total waste after program * emissions per ton

0.927 * 0.174 = 0.03

Total emissions SO_2 per ton waste after program

0.19

Difference in emissions per ton waste

-0.01

Price per kg SO_2 in $ 5.92

Total effect program on cost per ton waste due to difference in SO_2 emissions

$ -0.06

Effects of this synthetics program on other emissions are nearly zero.

iib) Recycling program paper and cardboard

A 100% recycling program of paper and cardboard has only a very minor effect on the energy productivity of incineration as the energy content of paper and cardboard is just on the average of the energy content of an average ton of waste (see table II-5). The effects on the energy productivity of landfilling is very big, a 43% decrease. This is mainly caused by the decrease in landfill gas. The life cycle production of landfill gas from one ton of waste is estimated by the formula given in footnote 1 of table II-1. The recycling program of paper and cardboard changes the organic carbon (CO) content from 143 kg per ton of waste to 82. This results in 34 m^3 per ton waste less exploited landfill gas. As the generated net electricity depends directly on this amount, the total delivered energy is 52 kWh per ton less, a 43% decrease in energy productivity. Given the total social opportunity cost of energy production, the effects on the opportunity cost can be calculated.

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