Burn or Bury?
A Social Cost Comparison of Final Waste Disposal Methods

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Summary

This paper uses private and environmental cost data for the Netherlands to evaluate the social cost of two final waste disposal methods, landfilling versus incineration using waste-to-energy (WTE) plants. The data only provide some support for the widespread policy preference for incineration over landfilling if the analysis is restricted to environmental costs alone. 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 discussed as well. Proper treatment of and energy recovery from landfills seem to be the most important targets for waste policy. WTE plants are a very expensive way to save on climate change emissions.

Keywords: Waste policy, Project evaluation, Incineration, Landfilling, Climate change

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I 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, next incinerated and, only if nothing else works, landfilled. Landfilling is often considered to be the worst option because it consumes a lot of space and runs a high risk of leakages to air, water and soil. Incineration produces 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. In other words, burning waste in waste incineration plants facilitates compliance with the Kyoto protocol.

However, incineration plants 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. Methane emissions of landfills, the main source of emissions to air, can be reduced by flaring, and can even be used to produce energy as well. 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.

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 proper. 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 NIMBY problems. Consequently, the focus of recent analytical and empirical studies has been on (social) cost comparisons, mainly on the assumption that the cost characteristics of different waste disposal programs would be easier to come by, 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

[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

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. Ontario Ministry of the Environment, 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, our study is the first to present an encompassing empirical analysis of the entire final waste disposal system including the indirect effects of its 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 and reflect (partly revealed) cost estimates of technologies that comply with the strictest waste disposal regulation in the world. Moreover, environmental conditions for final waste disposal are rather poor because the Netherlands is not only densely populated, like Singapore and 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.

Interestingly, our analysis casts serious doubts on the current policy preference for incineration over landfilling. With both private and environmental costs included, we find much lower overall cost for landfilling than for incineration. Only the savings on environmental externalities associated with recovery functions are responsible for lower environmental costs for incineration than for landfilling. But our calculations 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. So a shift from incineration without energy recovery to a WTE plant indeed minimizes social cost, and may also reflect a local minimum. Indeed, proper (environmental) treatment of landfills and using landfill gas for energy production seem to be promising targets for any social cost minimizing waste policy. We also argue that this result generalizes to other European countries and probably to the

available, such as including or excluding energy and materials recovery. Analytically, this requires a Kuhn–Tucker setup of the social cost minimization framework instead of the Lagrangian optimization technique applied by Brisson (1997).

[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).

USA.

The paper is organized as follows. We first illustrate the existing policy preferences for final waste disposal options in several developed, particularly European, countries (Section II). Next we discuss the characteristics of our social cost–benefit analysis of the choice between landfilling and incineration (Section III). Then we present our results for the best available techniques for the Netherlands (Section IV), and we explore the sensitivity of our results in Section V. Section VI 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).

II 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. This belief is often reflected in governmental documents or even in existing environmental law. For instance, the EU confirmed this hierarchy in preparing its recent 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 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:

> 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)

Although there is no explicit preference for incineration in the USA, reuse of materials and use of WTE facilities are undisputed to the EPA as well, again without any explicit reference to private cost.[3]

Existing final waste disposal, however, is still dominated by landfilling, even if stated policy preferences suggest otherwise. The first column of Table 1 shows that fewer than half of 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 all. 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 seem to be 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 is appropriately priced, selling electricity generates resources for the incinerating company, and therefore lowers the net cost of waste

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

[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'.

disposal. Third, WTE saves on electricity production and its associated externalities.

Table 1. Waste incineration characteristics in some developed countries, 2001

	Incinerated waste as a % of total municipal waste	Cultivated land as a % of total land[a]	Population density (people per sq. km)
Europe	33	22	122
Austria	20	43	98
Belgium	55	45	312
Denmark	100	63	126
Finland	5	9	17
France	63	55	107
Germany	72	50	235
Greece	0	68	82
Ireland	0	64	54
Italy	13	53	196
Luxembourg	47	na	166
Netherlands	113	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

a) Figures for 1994.
Sources: Ministry of Finance, 2002; World Bank, 2002; EPA, 2002; World Resources Institute, 1998.

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 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 or heating. WTE plants are thought to be efficient energy plants, because they already produce heat and its use for energy production reduces climate change emissions elsewhere in the economy.

It is still an open question whether modern WTE plants are the best solution or only provide a local optimum. That is, given a suboptimal status quo with waste incineration plants without energy recovery, WTE plants probably reduce the social costs of incineration. 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. As noted in the Introduction, joint production characteristics in terms of energy and materials recovery, as well as the environmental effects associated with all technologies, are important (including landfills with energy recovery).

III The choice between waste disposal options

Our goal is to evaluate existing policy preferences in a general framework that compares the social cost of different incineration techniques with different landfilling techniques. The best option is simply the final waste disposal technology that minimizes social cost at the margin. Obviously, how much waste should be incinerated and/or landfilled depends on the overall 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 social cost perspective, and vice versa.

The social costs for final waste disposal methods include, first of all, the private and direct environmental costs of both disposal technologies. For a given technology, *private costs* split into labor and capital cost for operation and maintenance. Direct *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. Also, 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.[5]

A second issue is the jointness characteristic of waste disposal methods. By choosing a specific technology, the government decides not only on final waste reduction and its environmental cost, but also on

[5] Note that the comparison of the environmental costs of different options follows from their *relative* environmental performance for a given set of shadow prices. 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.

the composition of energy production and the amount of materials recycling in the economy. Final waste disposal technologies differ in their jointly produced useful outputs, and therefore also in their overall environmental externalities. If specific technologies allow for energy and materials recovery, however, the externalities cannot be attributed entirely to the waste function alone. For instance, if a specific technology yields a better recovery performance, this might not only save private costs in producing these outputs but also affect their associated environmental cost.[6] It is also important to recognize that *both* disposal options might contribute to these joint outputs. Like incineration, landfilling might go together with energy production. However, materials such as aluminum can be recovered from the ash after incineration, whereas it would be extremely costly to recover them from a landfill. Therefore, only incineration plants are able to contribute to additional social savings in the production of these materials.

Thus, the government faces a wide variety of final waste disposal technologies, in particular if recycling characteristics (such as energy and materials recovery) are included. 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). In our approach, both private and environmental costs should be considered for each available technology, including the social costs connected to the jointly produced outputs, energy and materials recovery.[7]

Note, once more, that our approach reflects a much wider notion of 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. 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.

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

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[6] If the government selects another waste technology, this indirectly affects not only its energy policy 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.

[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,

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 (Vollebergh, 1997). Therefore, it is more appropriate to measure this contribution by its *social value* in the energy (and materials) system (including *potential* 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).

IV Social cost of waste disposal options in the Netherlands

This section presents a point estimate of the overall 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.[8] 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 direct and indirect private costs and externalities involved. The question of how the different technologies are priced relative to both their private and environmental costs is operationalized as follows. First of all, we derive private cost estimates for both final waste disposal options. Then, we include as much of the relevant physical environmental impacts as possible in order to get comparable results. Using a set of shadow prices, we turn these impacts into monetary values if possible. Finally, we select the relevant substitute as a reference system for each specific joint output. For this reference system, both private and environmental costs are estimated using cost estimates from other sources (see Appendix A for further details).

Our *private cost estimates* for waste *incineration* at a state-of-the-art incineration plant are obtained from a study by the Ministry of Finance (2002). This study calculates the private costs for a newly built waste incineration plant. 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 VI). The calculations use real figures for the best-practice

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and *L* and *K* as the usual labor and capital inputs.

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

incineration plant, accounting for cost savings that would apply if the plant were built in a market environment.[9] According to this study, total private costs equal Euro 103 per tonne for an incineration plant burning 648 ktonne per annum.[10] At this scale, the installation produces 580 kWh of electricity and 315 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.

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 (Ministry of Housing, Land Use and the Environment, 1992).[11] 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 VI), and includes energy recovery investments to generate 122 kWh of electricity per tonne of waste from landfill gas. We 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.

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 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 cost of the incineration plant.

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[9] 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.

[10] 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 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 648 ktonne is the minimum efficient scale, according to expert opinion.

[11] 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 farm land price in 2000, is still small and falls within the 'worst case' on which our figures are based.

Table 2. Private cost estimates for landfilling and incineration (Euro per tonne)

	Landfilling	Incineration
Gross private costs	40	103
Private cost savings:		
- Energy function	4	21
- Materials function	0	3
Net private costs	36	79

We have also been able to include all *direct and indirect environmental costs* categories 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. 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 reflect (expected) emissions of the best final waste disposal technologies currently available.[12] 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. 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.

The shadow prices used to calculate environmental costs reflect marginal abatement 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

[12] 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 V).

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 farm land 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 housebuilding land (Euro 227 per m^2) in 2000. Finally, we use market prices for recycled materials such as aluminum and iron.

Table 3. Environmental cost estimates for landfilling and incineration (Euro per tonne)

	Landfilling	Incineration
Environmental costs:		
- Emissions to air	5.85	17.26
- Emissions to water	0.00	0.00
- Chemical waste	2.63	28.69
- Land use	17.88	0.00
Gross environmental costs	26.36	45.95
Environmental cost savings:		
- Energy function	4.76	22.62
- Materials function	0.00	5.76
Net environmental costs	21.60	17.57

The overall environmental costs of the different functions of the waste disposal options are summarized in Table 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 2 and 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 is not supported by social cost data in a country where this support is most likely. This preference for incineration probably originates in the *overall* environmental cost savings, because incineration without recovery generates much higher 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.

V 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. 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.[13]

Another crucial issue is the extent to which our results depend on particular (shadow) price assumptions. First of all, 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 margin (*ceteris paribus*) 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.

Second, the *environmental cost estimates* are highly sensitive to the shadow prices used, although different prices always affect both options. 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 (see Appendix A). 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 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) could only be obtained with a shadow price for methane (CH_4) that is 7.9 times higher than currently assumed, and the shadow price for NO_x would have to be even 31.7 times higher than assumed currently. Note also that the shadow price for CO_2 should be *lower* than currently assumed in order to produce a TP.

Using the shadow prices of the six different studies presented in European Union (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

[13] 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 not usually a local problem, but its final disposal (at landfills or in closed salt mines) causes potential NIMBY problems elsewhere.

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.

Table 4. Shadow prices in this study compared with other studies (Euro per tonne of emissions)

	CO_2	CH_4	SO_2	NO_x
Our study				
- Initial	34	379	4,701	3,291
- Turning point	-414	2,997	54,727	104,479
Fankhauser, 1992	138	617		
CSERGE, 1993			592	456
Oak Ridge National Laboratory, 1995			1,191	5,162
Externe, 1997	4–140	45–1,583		1,494–15,731
European Union, 2000	4–42	53–2,223	2,100–12,200	4,300–18,340
Intergovernmental Panel on Climate Change, 2001	20–135			

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. 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 European Union (2000) – Euro 713 million per TEQ – results in total environmental costs of only Eurocent 21 per tonne of incinerated waste.

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

A final issue is the calculation of the environmental cost savings of the energy function. Our calculation is based on a gas-using electricity plant because this type of plant delivers the marginal unit of electric ity in the Netherlands. 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.

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.

VI Policy choices and consequences

Our analysis started from the observation that waste policy within the EU is strongly founded on the perception that incineration is preferred to landfilling, in particular in WTE plants. 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 environmental cost, but the difference is small. 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. We first discuss whether our estimates for the best-practice technologies complying with Dutch final waste disposal standards have any wider applicability, in particular within the EU. Then we evaluate current policy preferences with respect to solid waste management.

Wider applicability of our social cost estimates depends on the regulation by the government as well as on local siting conditions. The regulatory conditions imposed by the Netherlands have been an important focal point in drafting the EU directives on incineration and landfills.[14] For instance, Table 5 shows that the recent European standards for incineration plants are close to the Dutch ones, except for dust and acid rain

[14] These directives are published as L 332/91 (2000) and L 182/1 (1999). 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.

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). Without this limit, however, 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.

Table 5. Current environmental regulation for incineration plants in EU countries

	Dioxin (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 = EU (1989)	No limit	30	100	50	300	No limit
B = EU (2000)	0.1	10	50	10	50	200

Note: A and B refer to the EU (1989) and EU (2000) Directives respectively.
Source: Ministerie van Financien, 2002.

The other issue is whether the difference in social costs would be affected by differences in siting conditions between countries (assuming similar standards). This is certainly the case, as local ecological conditions as well as local prices differ, and we only have a point estimate based on average cost. First of all, note again 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) could be exploited at much lower social 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.[15] Note, furthermore, that our environmental cost estimate for landfills is likely to be high judged by European standards. The estimate is dominated by the shadow price for land use (68% of the gross cost), and this price would be (much) lower in less populated areas in Europe. As noted in the previous section, a 22% lower shadow price for land would render landfilling cost-efficient from the environmental perspective alone!

Applying our results to final waste disposal policies, 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.[16] 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.

A related issue is the prevalence of the typical disposal technologies assumed, and whether they are cost-efficient from the social cost perspective. For instance, further improvements in the energy efficiency of waste incineration would be possible using a larger boiler with higher steam pressure and machinery for reheating the steam. Such a so-called high-efficiency incineration plant would improve energy efficiency from the conventional 20% to 30%. The additional capital cost (including the lower availability of the plant due to more maintenance), however, would increase net private cost by Euro 15 per tonne of waste in the Netherlands.[17] According to our calculations, environmental cost would only be reduced by Euro 11 per tonne of waste, which does not compensate for the rise in private cost at the margin. Including energy recovery in

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[15] 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 Ministry of Finance, 2002).

[16] 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.

[17] 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.

landfills also finds strong support from the empirical evidence, even from a private cost perspective alone. 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 2). In fact, the rents are much higher from the social cost perspective because the gross environmental costs without energy recovery would be much higher (see earlier).

Finally, current Dutch and EU policy on final waste disposal, in contrast to the US policy, reflects a 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.[18] 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 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, 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. 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. Differences in regulation may have major impacts on the overall private cost of specific technologies, and therefore also drive potential waste races to the bottom.

Evaluating optimal standard setting within the EU is beyond the scope of our paper,[19] but this example clearly illustrates that our general findings only hold for best available technologies complying with a particular

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[18] This implies a corner solution in a model where the social cost minimizing agent optimizes final waste disposal subject to several constraints describing the jointness characteristics of the waste disposal production process. Interior solutions, i.e. solutions that allow for both incineration and landfilling to be optimal, are possible only if the marginal costs of both options vary enough over certain ranges of waste to be disposed.

[19] 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.

set of standards. Indeed, a large number of French incinerators will be closed, given the newly established legislation in France, which is similar to Dutch standards (see Table 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.[20]

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 are considerable if the focus is on expanding incineration based on these new standards, while a modern landfill is 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.

VII Conclusions

From a social cost minimizing perspective, we find little support for a hierarchical approach towards final waste disposal methods. WTE plants are only a *local* minimum if waste incineration is expanded anyway. Because energy recovery is a joint product obtainable with small additional investments in technology, not only will the net private cost be lower but also the environmental cost of the energy system. However, this strategy does not attain a *global* optimum, given a rather strict set of standards applying to landfills and incineration

[20] 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.

plants.

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. 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 may be a very expensive climate change abatement option. The net private cost 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.

One important caveat remains. Further empirical research is necessary to compare our calculations for the Dutch case with other techniques employed in practice at other locations. Moreover, 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.

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