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Environmental impacts and costs of solid waste: a comparison of landfill and incineration

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The methodology for evaluating the impacts and damage costs ('external costs') due to pollution from waste treatment is described and the results are presented, based on the ExternE project series of the European Commission. The damage costs of landfill and incineration of municipal solid waste are compared, with due account for energy and materials recovery, as well as possible differences in transport distance. We have not been able to quantify the total damage costs of leachates because of the complexity of the environmental pathways and of the long time horizon of some persistent pollutants, but we consider an extreme scenario to show that they are not worth worrying about in the sense that reducing the pollutants in leachates beyond current regulations would bring negligible benefit in comparison with the abatement of other sources of the same pollutants. The damage costs due to the construction of the waste treatment facility are negligible. The damage costs of waste transport, which are illustrated by an arbitrary choice of a 100 km round trip by a 16 tonne truck, are also negligible. The benefits of materials recovery make a small contribution to the total damage cost. The only significant contributions come from direct emissions (of the landfill or incinerator) and from avoided emissions due to energy recovery (from an incinerator). Damage costs for incineration range from about 4 to 21 € $\text{tonne}_{\text{waste}}^{-1}$, and they are extremely dependent on the assumed scenario for energy recovery. For landfill the cost ranges from about 10 to 13 € $\text{tonne}_{\text{waste}}^{-1}$; it is dominated by greenhouse gas emissions because only a fraction of the CH_4 can be captured (here assumed to be 70%). Amenity costs (odour, visual impact, noise) are highly site-specific and we only cite results from a literature survey which indicates that such costs could make a significant contribution, very roughly on the order of 1 € $\text{tonne}_{\text{waste}}^{-1}$.

Introduction

In many countries there has been a lively and at times acrimonious debate about the treatment of waste. In particular, incineration is often vehemently opposed because of its health impacts. However, it would be unwise to reject it in favour of landfill if the total cost (including damage to environment and health) of the latter turns out to be larger than that of incineration. A review of the damage costs (also called 'external costs') of waste treatment was carried out by COWI (2000); however, the damage costs per kilogram of emitted pollutant were based on values in the literature rather than original research. The principal original studies that have quantified the costs per kilogram of pollutant emitted by incineration are the ExternE project series of the EC (ExternE 1998, 2000, 2004) or directly based on its methodology (Rabl *et al*. 1998).

After completion of the first version of this paper we became aware of a very interesting and important study by Dijkgraaf and Vollebergh (2004) who also compare landfill and incineration. In contrast to our paper, they examine the private costs in addition to the external costs. Their results for the external costs are consistent with ours, although they are based on an older study in the Netherlands (CE 1996), whereas ours uses the latest results of ExternE. We will say more on their paper in the conclusion.

The purpose of the present paper is to evaluate and compare the damage costs of landfill and incineration due to pollution, on a consistent basis using the damage costs of the latest phase of ExternE (2004). The methodology involves an impact pathway analysis (see below) rather than contingent valuation

surveys of the perception of waste treatment facilities by the general public (who do not understand the quantitative link between emissions and impacts). The waste to be treated is municipal solid waste (MSW), or more precisely, the fraction of MSW that remains after source reduction and recycling. We show results for typical installations in France, but indicate also how the results could be transferred to other countries.

We do not evaluate amenity impacts (odours, noise, visual intrusion, etc) but only cite the literature survey of Walton *et al*. (2006). Such costs could make a significant contribution, very roughly of the order of 1 € tonne$_{waste}^{-1}$ depending on site and discount rate. We emphasize that the appropriate internalization mechanisms are completely different from those for air pollution damage, and therefore we do not find it appropriate to show them on the same graph. Amenity and land use impacts are limited to the population in the immediate vicinity of an installation and extremely variable from one site to another; the appropriate internalization is by negotiation with the local population before the project is approved. By contrast, air pollution damages are regional (continental scale) for most pollutants and global for greenhouse gases and mercury; their variation with emission site is straightforward (well understood, with simple rules for approximate general trends, see, the section entitled 'Cost per kilogram of pollutant' in the present paper) and they should be internalized by national or international regulations.

To calculate the damage costs one needs to carry out an impact pathway analysis (IPA), tracing the passage of a pollutant from where it is emitted to the affected receptors (population, crops, forests, buildings, etc.). The principal steps of an IPA can be grouped in the following manner.

- Emission: specification of the relevant technologies and pollutants, e.g. kilogram of NO_x per tonne of waste emitted by an incinerator.
- Dispersion: calculation of increased pollutant concentrations in all affected regions, e.g. incremental concentration of ozone, using models of atmospheric dispersion and chemistry for ozone formation due to NO_x (this step is also called environmental fate analysis, especially when it involves more complex pathways that pass through the food chain).
- Impact: calculation of the dose from the increased concentration and calculation of impacts (damage in physical units) from this dose, using a dose–response function, e.g. cases of asthma due to this increase in ozone.
- Cost: monetary valuation of these impacts, e.g. multiplication by the cost of a case of asthma.

The impacts and costs are summed over all receptors of concern; for the numbers presented here this includes the entire European continent. The result of an IPA is the damage cost per kilogram of emitted pollutant; multiplication by the emissions per tonne of waste yields the damage cost per tonne of waste. The steps of the IPA are described in the following section. The work involves a multidisciplinary systems analysis, with inputs from engineers, dispersion modellers, epidemiologists, ecologists, and economists. The present paper is based on the results of the ExternE project series of the EC (ExternE 2004), plus more recent updates for Hg and Pb.

In the spirit of life-cycle assessment (LCA), the IPA is carried out for all stages of the waste treatment process where there are significant differences between landfill and incineration. Thus we consider the construction of the facilities, possible differences in transport distance, emissions from the landfill and incineration stage, and emissions that are avoided thanks to the recovery of energy and materials. In contrast to most conventional LCA, however, we calculate real impacts and costs rather than a set of non-commensurate indices of 'potential impact'. We present results for typical sites and indicate how the results could vary from one site to another.

The uncertainties are large (Rabl & Spadaro 1999, Spadaro & Rabl 2007a) and not all impacts can be quantified adequately. For example, the impacts of leachates from landfill are problematic because they involve scenarios about the future management of the site, more than 30 years after closure (the time limit for the responsibility of the owner under current EU legislation). However, current regulations for leachates are very strict and their impacts are more or less confined to the local zone of a few kilometres; they are unlikely to make a significant contribution to the damage cost (unless a large aquifer can be contaminated). On the other hand, greenhouse gas emissions, especially CH_4, from landfills are appreciable and their cost is comparable with the damage costs of incineration. The comparison turns out to hinge on the emissions avoided by recovery of energy and materials.

Impact pathway analysis
Dispersion of pollutants and exposure

The principal greenhouse gases, CO_2, CH_4 and N_2O, stay in the atmosphere long enough to mix uniformly over the entire globe. No specific dispersion calculation is needed but the estimation of impacts is extraordinarily complex. Here we refer merely to the main authority, the Intergovernmental Panel on Climate Change [IPCC http://www.ipcc.ch]. For most other air pollutants, in particular PM_{10}, NO_x and SO_2, atmospheric dispersion is significant over hundreds to thousands of kilometres, so both local and regional effects are important. ExternE uses therefore a combination of local and regional dispersion models to account for all significant damages. The main model for the local range (< 50 km from the source) has been the Gaussian plume model ISC (Brode & Wang 1992).

At the regional scale one needs to take into account the chemical reactions that lead to the transformation of primary pollutants (i.e. the pollutants as they are emitted) to secondary pollutants, for example the creation of sulfates from SO_2. Here ExternE uses the Windrose Trajectory Model (WTM) (Trukenmüller & Friedrich 1995) to estimate the concentration and deposition of acid species. WTM is a user-configurable Lagrangian trajectory model, derived from the Harwell Trajectory model (Derwent & Nodop 1986). The modelling of
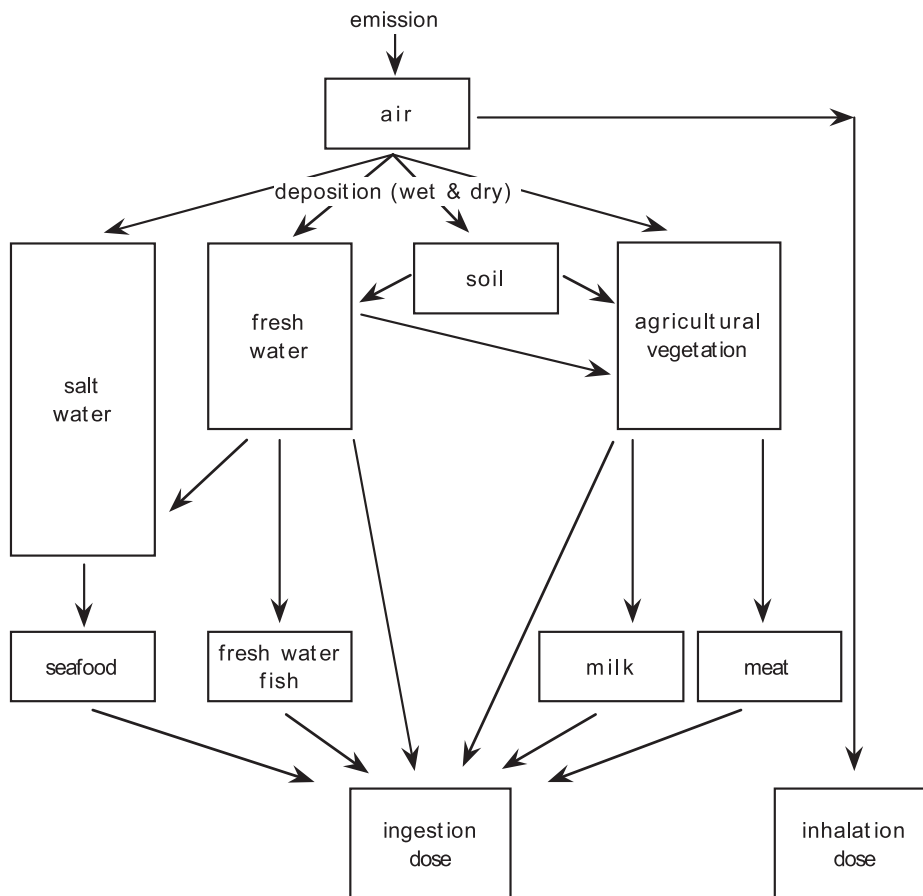
Fig. 1: Pathways taken into account for health impacts of air pollutants. Direct emissions to soil or water are a special case where the analysis begins at the respective 'soil' and 'water' boxes.

ozone is based on the EMEP MSC-W oxidant model (Simpson 1992, Simpson & Eliassen 1997). EMEP is the official model used for policy decisions about transboundary air pollution in Europe (http://www.emep.int).

The calculation of damage costs is carried out by means of the EcoSense software package (Krewitt *et al*. 1995), an integrated impact assessment model that combines these atmospheric models with databases for receptors (population, land use, agricultural production, buildings and materials, etc.), dose–response functions and monetary values. There is also a simplified analysis tool, called RiskPoll (actually a package of several models with different input requirements), developed by J. Spadaro and freely available from www.arirabl.org or www.externe.info. It is based on the interpolation of Eco-Sense dispersion calculations, and with its simplest version yields damage costs that are typically within a factor of two to three of detailed EcoSense calculations for stack heights above 50 m. RiskPoll includes a module for the multimedia pathways of Figure 1.

Several tests have been carried out to confirm the accuracy of the EcoSense dispersion calculations. For example, we have verified the consistency between the Gaussian plume models ISC and ROADPOL (Vossiniotis *et al*. 1996), and we have compared the concentrations predicted by EcoSense with measured data and with calculations of the EMEP program.

Whereas only the inhalation dose matters for PM_{10}, NO_x, SO_2 and O_3, toxic metals and persistent organic pollutants also affect us through food and drink. For these a much more complex IPA is required to calculate ingestion doses. Spadaro & Rabl (2004) have developed a model for the assessment of external costs due to the emission of the most toxic metals (As, Cd, Cr, Hg, Ni and Pb), as well as certain organic pollutants, in particular dioxins. It takes into account the pathways in Figure 1. This model is a generalization of the UWM described in the section entitled 'UWM: a simple model for damage cost estimation' below. It is based mostly on transfer factors published by EPA (1998), with some supplemental data of IAEA (1994 & 2001). These transfer factors account in a simple manner for the transport of a pollutant between different environmental compartments, for example the uptake by agricultural crops of a pollutant from the soil. The uncertainties are large, but at least one has approximate values for the pollutants of concern here.

A general result of this analysis is that when these pollutants are emitted into the air, the collective ingestion dose can be about two orders of magnitude larger than the collective dose by inhalation. As most food is transported over very large distances nowadays, the total dose varies little with the site where these pollutants are emitted into the air (except due to the variation of agricultural crops with emission site). As far as damage is concerned, one has to note that the same

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dose can have a very different effect on the body depending on whether it is inhaled or ingested. Cd, Cr^{VI} and Ni, for instance, are, according to current knowledge, carcinogenic only through inhalation. The impact of metals also depends on the chemical speciation; for example methyl-Hg is far more toxic than Hg vapour.

Dose–response functions
General considerations
The dose–response function (DRF) relates the quantity of a pollutant that affects a receptor (e.g. population) to the physical impact on this receptor (e.g. incremental number of hospitalizations). In the narrow sense of the term, it should be based on the dose actually absorbed by a receptor. However, the term dose–response function is often used in a wider sense where it is formulated directly in terms of the concentration of a pollutant in the ambient air, accounting implicitly for the absorption of the pollutant from the air into the body. The functions for the classical air pollutants (NO_x, SO_2, O_3, and particulates) are typically of that kind, and the terms exposure–response function or concentration–response function (CRF) are often used.

The DRF is a central ingredient in the impact pathway analysis and merits special attention. A particular damage can be quantified only if the corresponding DRF is known. Such functions are available for many of the impacts on human health, building materials and crops, that can be caused by a range of pollutants, in particular primary and secondary (i.e. nitrates, sulfates) particles, ozone, CO, SO_2, NO_x, Benzene, benzo(a)pyrene, formaldehyde, dioxins, As, Cd, Cr, Hg, Ni and Pb. The most comprehensive reference for health impacts is the IRIS database of EPA (http://www.epa.gov/iriswebp/iris/index.html). For application in an IPA that information often has to be expressed in somewhat different form, including additional factors such as the background incidence rate (ExternE 2004, Spadaro & Rabl 2004). Unfortunately, for many pollutants and many impacts the DRFs are very uncertain or not even known at all. For most substances and non-cancer impacts the only available information covers thresholds, typically the NOAEL (no observed adverse effect level) or LOAEL (lowest observed adverse effect level). Knowing thresholds is not sufficient for quantifying impacts; it only provides an answer to the question whether or not there is a risk. The principal exceptions are carcinogens and the classical air pollutants, for which explicit DRFs are known (often on the assumption of linearity and no threshold).

ExternE assumes that all DRFs for health impacts, including neurotoxic effects of Hg and Pb and cancers, are linear at the population level, in view of the lack of evidence for thresholds at current ambient concentrations [however, for Hg we assume a threshold equal to the RfD of EPA (Spadaro & Rabl 2007b)]. By contrast to the homogeneous populations of cloned animals studied by toxicologists, the absence of a no-effect threshold is plausible for real human populations because they always contain individuals with widely differing sensitivities (for example, at any moment about 1% is within

the last 9 months of life and thus extremely frail). Note that for the calculation of incremental damage costs there is no difference between the linear and the hockey stick function (with the same slope), if the background concentration is everywhere above this threshold; only the slope s_{DR} of the DRF matters. For the particles, NO_x, SO_2, O_3 and CO the background in most industrialized countries is above the level where effects are known to occur. Thus the precise form of the DRF at extremely low doses is irrelevant for these pollutants; if there is a no-effects threshold, it is below the background concentrations of interest.

Health impacts
In terms of costs, health impacts contribute the largest part of the damage estimates of ExternE. A consensus has been emerging among public health experts that air pollution, even at current ambient levels, aggravates morbidity (especially respiratory and cardiovascular diseases) and leads to premature mortality (e.g. Wilson & Spengler 1996, WHO 2003). There is less certainty about specific causes, but most recent studies have identified fine particles (PM_{10} or $PM_{2.5}$) as a prime culprit; ozone has also been implicated directly. The most important cost comes from chronic mortality due to particles, calculated on the basis of Pope *et al*. (2002) (this term, chosen by analogy with acute and chronic morbidity impacts, indicates that the total or long-term effects of pollution on mortality have been included, by contrast to acute mortality impacts that are observed within a few days of exposure to pollution). Another important contribution comes from chronic bronchitis due to particles (Abbey *et al*. 1995). In addition there may be significant direct health impacts of SO_2, but for direct impacts of NO_x the evidence is less convincing.

In ExternE the working hypothesis has been to use the CRFs for particles and for O_3 as a basis. The effects of NO_x and SO_2 are assumed to arise indirectly from the particulate nature of nitrate and sulfate aerosols, and they are calculated by applying the particle CRFs to these aerosol concentrations. The uncertainties are, however, large because there is insufficient evidence for the health impacts of the individual components or characteristics (acidity, solubility, …) of particulate air pollution. In particular there is a lack of epidemiological studies of nitrate aerosols because until recently this pollutant has not been monitored by air pollution monitoring stations.

Among the toxic metals the following are considered carcinogenic: arsenic (As), cadmium (Cd), chrome (Cr, in oxidation state VI) and nickel (Ni). We use the DRFs published by the IRIS data base of EPA (http://www.epa.gov/iriswebp/iris/index.html). For Pb we cite the damage cost due to IQ (intelligence quotient) decrement as calculated by Spadaro & Rabl (2004). More recently Spadaro & Rabl (2007b) have estimated the global contribution of IQ loss to the damage cost of Hg (the real damage for a specific site can be different because of local and regional variations that are much more difficult to estimate). Hg has a long residence time in the

atmosphere, of the order of 2 years, and is therefore a globally dispersing pollutant; most of its health impacts arise after its transformation by aquatic organisms to methyl-mercury and its ingestion via seafood. We also consider dioxin, a pollutant emitted from the incineration of municipal solid waste. The calculation is documented in Rabl *et al*. (1998). Note that the damage costs in the present paper are somewhat different from that reference, from Rabl & Spadaro (2002) and from Spadaro & Rabl (2004) because some of the DRFs (e.g. for dioxins) and monetary values have been updated.

Monetary valuation

The goal of the monetary valuation of damages is to account for all costs, market and non-market. For example, the valuation of an asthma attack should include not only the cost of the medical treatment but also the willingness-to-pay (WTP) to avoid the residual suffering. It turns out that damage costs of air pollution are dominated by non-market goods, especially mortality. If the WTP for a non-market good has been determined correctly, it is like a price, consistent with prices paid for market goods. Economists have developed several tools for determining non-market costs. Of these tools contingent valuation (CV) has enjoyed increasing popularity in recent years (Mitchell & Carson 1989). The results of well conducted studies are considered sufficiently reliable for informing public policy.

For the valuation of mortality a crucial parameter is the value of a prevented fatality (VPF; which has usually been called 'value of statistical life', an unfortunate terminology for what is really the 'willingness to pay for reducing the risk of an anonymous premature death'). In ExternE (1998), a European-wide value of 3.1 M€ was chosen for VPF, somewhat lower than similar studies in the USA; this value was chosen as the average of the VPF studies that had been carried out in Europe. The uncertainty is large and one could argue for other values in the range of 1 to 5 M€. Currently ExternE uses 1 M€, based on a recent CV study by the ExternE team (Markandya *et al*. 2004); that is also close to the value recommended for air pollution deaths by the DG Environment of the European Commission (EC 2000b).

A key question for air pollution mortality is whether one should simply multiply the number of premature deaths by VPF, or whether one should take into account the years of life lost (YOLL) per death. The difference is very important because premature deaths from air pollution tend to involve far fewer YOLL per death than accidents (on which VPF is based) (Rabl 2003). The ExternE (2004) numbers, used here, are based on YOLL, by contrast to most previous external cost studies (studies in the USA continue to use VPF, exclusively or in parallel with YOLL). There is considerable uncertainty about the relation between VPF (which has been determined for accidents) and the value of a life year (VOLY) appropriate for air pollution, because it involves the period at the end of life about which valuation studies are only just beginning. In ExternE (1998) the value of a YOLL had therefore been calculated on theoretical grounds by considering

VPF as the net present value of a series of discounted annual values. The ratio of VPF and the value of a YOLL thus obtained depend on the discount rate; it is typically in the range of 20 to 40. More recently a CV study by ExternE found a VOLY of 50 000 € and that has been used for the current damage cost estimates (ExternE 2004).

For cancers the loss of life per premature death is intermediate between accidents and air pollution, but economists also assume a premium because cancers are feared as an especially dreadful form of death. Here we assume a value of 2.0 M€, averaged over fatal and non-fatal cancers. For neurotoxic impacts we take the value of an IQ point to be 10 000 €.

Global warming

The valuation of global warming damages is extremely complex, see, for example, Tol *et al*. (2001). Not only is the task difficult because of the large number of different impacts in all countries of the world that should be taken into account, but as these impacts will occur in future decades and centuries one needs to estimate how these costs will evolve into the distant future. On top of the resulting uncertainties there are controversial ethical issues related to the valuation of mortality in developing countries (where most of the impacts will occur) and the choice of the discount rate for intergenerational costs.

Several major studies have been published with estimates of the damage cost per tonne of CO_{2eq}; the subscript eq indicates that the result can also be used for other greenhouse gases if their masses are multiplied by their global warming potential (GWP). Most of the results are in the range 1 to 50 € t_{CO2eq}^{-1}; the range is so wide because of the large uncertainties. The ExternE team carried out two valuation efforts, the first, in 1998, yielded a range of values with a geometric mean of 29 € t_{CO2eq}^{-1}, the second, in 2000, obtained a much lower value of 2.4 € t_{CO2eq}^{-1} because of more optimistic assumptions and a better accounting for benefits such as increased agricultural production in cold countries. Because of the difficulty of determining the damage cost of CO_2, the current phase of ExternE uses as proxy the abatement cost in the EU implied by the commitment to the Kyoto protocol, 19 € t_{CO2eq}^{-1}. It thus represents an implicit valuation by decision makers of the EU. It is also in effect the cost imposed on the EU by incremental emissions of CO_2 in the EU. Even though this is not the damage cost, the choice appears reasonable in view of the damage cost estimates published in the literature (Tol 2005).

Results per kilogram of pollutant
Cost per kilogram of pollutant

The impacts quantified by ExternE so far are global warming, health, damage to buildings and materials, and loss of agricultural production. Apart from global warming due to CO_2, CH_4 and N_2O, more than 95% of the costs are due to health impacts, especially mortality. Morbidity (above all chronic bronchitis, but also asthma, work days lost, hospital

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Table 1: Impacts evaluated and key assumptions (ExternE 2004).

Atmospheric dispersion models	
Local range	Gaussian plume models ISC (point sources) or ROADPOL (emissions from transport)
Regional range (Europe)	Harwell Trajectory Model as implemented in ECOSENSE software of ExternE
	O_3 concentrations based on EMEP model
Physical impacts	
Impacts on health	
Form of dose-response functions	Linearity without threshold, with slope s_{CR}
Chronic mortality, YOLL (years of life lost)	s_{CR} = 3.9 E–4 YOLL per person per year per μg m^{-3} PM$_{10}$, derived from increase in age-specific mortality (Pope *et al.* 2002) by integrating over age distribution
Acute mortality	For SO_2 and O_3, assume 0.75 YOLL per premature death
Nitrate and sulfate aerosols	Dose response function for sulfates same as for PM$_{10}$
	($s_{CR,sulfate}$ = $s_{CR,PM10}$ = 0.6 $s_{CR,PM2.5}$)
	Dose response function for nitrates: toxicity 50% of PM$_{10}$
	($s_{CR,nitrate}$ = 0.5 $s_{CR,PM10}$)
Micropollutants	Cancers due to As, Cd, Cr, Ni, and dioxins
	DRF for dioxins according to EPA (2000)
	IQ decrement due to Hg and Pb
Impacts on plants	Loss of crops due to SO_2 and O_3
Impacts on buildings and materials	Corrosion and erosion due to SO_2
	Soiling due to particles
Impacts not quantified but potentially significant	Reduced visibility due to air pollution
	Eutrophication and acidification
	Disposal of residues from fossil fuels or incineration
Monetary valuation	
Valuation of premature death	Proportional to reduction of life expectancy, with value of a of life year (VOLY) = 50 000 €
Valuation of cancers	2 M€ per cancer
Valuation of neurotoxicity	10 000 € per IQ point lost
Global warming damage cost	0.019 € kg$_{CO2eq}$$^{-1}$

admissions etc) account for almost a third of the damage cost of PM$_{10}$, NO$_x$ and SO$_2$. The impacts evaluated and the key assumptions are listed in Table 1.

The resulting damage costs in € kg^{-1} of pollutant are shown in Table 2 for typical sources with stack height *h* above 40 m in Central Europe, as well as for road transport in France. The uncertainty is also indicated. Note that the damage costs are somewhat different from those published by previous phases of ExternE, because the methodology has been evolving.

To estimate results for other regions, simply rescale the numbers in proportion to the regional average receptor density ρ (within 1000 km) and the cost per case (if different). As for variation with site and stack height, the following can be stated.

- There is no dependence on site or stack height for globally dispersing pollutants such as CO$_2$. For As, Hg, Pb and dioxins the variation with site, for a given ρ, is in the range of about 0.5 to 2.0, which is small because non-inhalation pathways dominate. Variation with site, for a given ρ, is also small for secondary pollutants, a range of about 0.5 to 2.0 because the formation of the secondary pollutants is slow and occurs mostly far from the source. Variation with

stack height is negligible for non-inhalation pathways and for secondary particles (nitrates and sulfates).
- For primary air pollutants the variation with site and stack height is strong and the result of D_{uni} (see section below entitled 'UWM: a simple model for damage cost estimation') can be improved by using the following correction factors:
  - 0.5 to 5 for site (higher if near big city),
  - 0.6 to 3 for stack conditions (higher for low stacks, up to 15 for ground level emissions in large city).

These correction factors have been derived by evaluating the results of more than a hundred detailed EcoSense calculations. Of course, such rules can only yield rough estimates; site-specific calculations should be carried out when more precise results are needed.

Health risks per kilogram of pollutant
Sometimes a decision-maker may prefer to see results in terms of YOLL (years of life lost) rather than costs in order to make risk–risk comparisons that avoid the uncertainties of monetary valuation. Therefore we also show in Table 2 the dominant health impacts in physical units per kilogram of pollutant.

Table 2: Damage cost and most important health impact (end point) per kg of pollutant. For PM, NO_x and SO_2 morbidity impacts account for about 30% of the damage cost. h = stack height.

Pollutant	€ kg^{-1} [range]	end point	% due to end point	€/end point	Impact kg^{-1}
Traffic, h = 0 m					
PM$_{2.5}$, rural	1.52E+01 [2.8E+00, 2.5E+01]	YOLL	67%	50000	2.1E–04 YOLL kg^{-1}
PM$_{2.5}$, highway	1.15E+02 [2.1E+01, 1.9E+02]	YOLL	67%	50000	1.6E–03 YOLL kg^{-1}
PM$_{2.5}$, Paris	1.58E+03 [2.9E+02, 2.6E+03]	YOLL	67%	50000	2.1E–02 YOLL kg^{-1}
Stacks, h = 100 m		YOLL			
PM$_{10}$, rural	5.2E+00 [9.5E–01, 8.5E+00]	YOLL	67%	50000	7.1E–05 YOLL kg^{-1}
PM$_{10}$, urban	1.2E+01 [2.1E+00, 1.9E+01]	YOLL	67%	50000	1.6E–04 YOLL kg^{-1}
PM$_{10}$, Paris	6.2E+01 [1.1E+01, 1.0E+02]	YOLL	67%	50000	8.5E–04 YOLL kg^{-1}
Cd	3.9E+01 [7.1E+00, 6.4E+01]	cancers	100%	2000000	2.0E–05 cancers kg^{-1}
Cr^{VI}	2.0E+02 [3.7E+01, 3.3E+02]	cancers	100%	2000000	1.0E–04 cancers kg^{-1}
Ni	3.8E+00 [6.9E–01, 6.2E+00]	cancers	100%	2000000	1.9E–06 cancers kg^{-1}
Little h dependence					
SO$_2$, via sulfates	3.5E+00 [6.4E–01, 5.7E–00]	YOLL	68%	50000	4.8E–05 YOLL kg^{-1}
NO$_2$, via nitrates	3.4E+00 [6.1E–01, 5.5E+00]	YOLL	68%	50000	4.6E–05 YOLL kg^{-1}
As	8.0E+01 [7.7E+00, 1.2E+02]	cancers	100%	2000000	4.0E–05 cancers kg^{-1}
Pb	6.0E+02 [5.7E+01, 9.2E+02]	IQ points	100%	10000	6.0E–02 IQ points kg^{-1}
Hg	8.0E+03 [7.7E+02, 1.2E+04]	IQ points	100%	10000	8.0E–01 IQ points kg^{-1}
Dioxins	1.85E+08 [1.0E+07, 2.5E+08]	cancers	100%	2000000	9.3E+01 cancers kg^{-1}
No h dependence					
CO$_2$	1.9E–02 [6.4E–04, 2.3E–02]				

UWM: a simple model for damage cost estimation

A simple and convenient tool for the development of typical values is the 'uniform world model' (UWM), first presented by Curtiss & Rabl (1996) and further developed, with detailed validation studies, by Spadaro (1999), Spadaro & Rabl (1999, 2002). More recently Spadaro & Rabl (2004) have extended it to toxic metals and their pathways through the food chain. The UWM is a product of a few factors; it is simple and transparent, showing at a glance the role of the most important parameters of the impact pathway analysis. It is exact for tall stacks in the limit where the distribution of either the sources or the receptors is uniform and the ratio of surface concentration and deposition (and/or transformation) rate does not vary with location. In practice the agreement with detailed models is usually within a factor of two for stack heights above 50 m. For policy applications one needs typical

values and the UWM is more relevant than a detailed analysis for a specific site.

The UWM for the damage cost D_{uni} in € kg^{-1} of a particular impact due to the inhalation of a primary pollutant is shown in equation (1).

$$D_{uni} = p \, s_{CR} \, \rho / v_{dep} \qquad (1)$$

where p is the cost per case ('price') (€ case^{-1}); s_{CR} is the CRF slope [(cases year^{-1})/(person·μg m^{-3}))]; ρ is the population density (person km^{-2}) averaged over land and water within approximately 1000 km of source; and v_{dep} = deposition velocity of pollutant (dry + wet) (m s^{-1}).

For secondary pollutants the equation has the same form, but with an effective deposition velocity that includes the transformation rate of the primary into the secondary pollut-

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Table 3: Assumptions of the analysis of incineration and landfill of MSW.

Stages taken into account	Construction of landfill or incinerator (negligible) Transport of waste (negligible) Emissions from landfill or incinerator Avoided emissions due to energy recovery Avoided emissions due to materials recovery
Emissions from incinerator	Equal to limit values of Directive EC (2000a)[a]
Avoided emissions due to energy recovery	Equal to limit values of Large Combustion Plant Directive (EC 2001)[a]
Impact pathway analysis	Assumptions and results of ExternE (2004) see Tables 1 and 2.
Impacts that have been quantified	Human health Crops Materials and buildings Global warming Amenity impacts (very site specific, not included in results, only order of magnitude is indicated in text, based on Walton *et al*. (2006)).
Impacts that have not been quantified	Effects of air pollutants on ecosystems Reduction of visibility due to air pollution Soil and water pollution due to leachates (but shown not to be of concern, see section entitled 'Damage cost of leachates') Impacts from residues of incineration (negligible if correctly managed).

[a] In reality the average emissions are usually lower.

ant. With this model it is easy to transfer to the results from one region to another (assuming that CRF and deposition velocity are the same): simply rescale the result in proportion to the receptor density and the cost per case.

Comparison landfill ↔ incineration
Assumptions

As a comparison of different waste treatment options necessitates an LCA, the work begins by choosing the boundaries of the analysis. The most appropriate choice is to start at the point where the waste has been collected and sorted. From here the waste must be transported to the landfill or incinerator; we have included the emissions due to transport by showing a hypothetical round trip distance of 100 km, for the purpose of illustration. In addition to the emission of pollutants from the landfill or incinerator, the emissions avoided by recovery of energy and materials are also taken into account, based on the LCA data of ADEME (2000). The main assumptions of the analysis are summarized in Table 3.

We have explicitly quantified the impacts of the stages in Table 3, with the exception of the construction of the landfill or incinerator. For the latter we merely refer to the LCA of power plants carried out by ExternE, where the emissions from construction (due to materials production) were found to be about three orders of magnitude smaller than those during operation. That holds also for other combustion equipment that is used full time, in particular for waste incinerators. With landfills the impacts of construction are also negligible compared to the utilization stage.

We assume that the incinerator emissions are equal to limit values of Directive EC (2000a); in reality the average emissions are usually lower if compliance is enforced but representative data are difficult to obtain. Our assumptions for the emissions from incinerators are listed in Table 4.

Part of municipal solid waste is of biological origin and its combustion emits CO_2. In the LCA community a special convention has been established according to which such CO_2 emission should not be counted. The accounting framework of ExternE, by contrast, assigns the same damage cost per kilogram of emitted CO_2, regardless of its origin. We follow ExternE, arguing that the convention of current LCA practice is inappropriate because it fails to distinguish different options for reducing the emission of such CO_2. The logic of such a practice would imply absurd conclusions, for instance that the burning of tropical forests be counted the same way as their preservation, and that adding carbon capture and sequestration to a biomass fuelled power plant would have no benefit for global warming (Rabl et al. 2007).

The principal emissions from landfill are CH_4 and CO_2. Figure 2 shows the total greenhouse gas emissions of a municipal solid waste landfill versus time. CH_4 is expressed as equivalent CO_2, using a GWP (global warming potential) of 23 (IPCC 2001). Note that a modern landfill is divided into a large number of individual compartments; they are filled one after another and sealed when full. The data of ADEME (2003) are plotted in Figure 2, where the time is measured from the date that a compartment is sealed. In practice it is impossible to capture all of the CH_4, and capture rates around 70% are commonly assumed (although measured data seem to be difficult to find). Here we assume a capture rate of 70% for the first 40 years, on average, after closure of a compartment; after 40 years we assume that all the remaining CH_4 escapes to the atmosphere.

Damage cost of leachates

There are also emissions to soil and to water. Emissions to soil can occur from slag, from leaking liners under a landfill, and from the storage site of incinerator fly ash. Emissions to water arise from certain types of flue gas treatment and from

Table 4: Assumptions for the emissions from incineration of MSW.

Pollutant	mg Nm^{-3}	g t$_{waste}$$^{-1}$	€ kg$_{pollutant}$$^{-1}$	€ t$_{waste}$$^{-1}$
PM$_{10}$	10	51.5	12	0.62
SO$_2$	50	258	3.5	0.88
NO$_2$	200	1030	3.4	3.61
CO$_2$		861800	0.019	15.33
As (2.8% of 0.5 mg Nm^{-3})	0.014	0.072	80	0.01
Cd (81.2% of 0.05 mg Nm^{-3})	0.0406	0.21	39	0.01
Cr^{VI} (6.5% of 0.2 × 0.05 mg Nm^{-3}) ^{a}	0.00065	0.0033	200	0.00
Hg (0.05 mg Nm^{-3})	0.05	0.26	8000	2.06
Ni (33.8% of 0.5 mg Nm^{-3})	0.169	0.87	3.8	0.01
Pb (22% of 0.5 mg Nm^{-3})	0.11	0.57	600	0.34
Dioxins	1.00E–07	5.15E–07	185000000	0.10

The assumptions are taken as the limit values of the flue gas concentrations, in Directive EC (2000a), assuming 5150 Nm^3 t$_{waste}$$^{-1}$. For metals the directive specifies only 0.5 mg Nm^{-3} for the sum of As + Co + Cr + Cu + Mn + Ni + Pb + Sn + Sb + V, and 0.05 mg Nm^{-3} for the sum of Cd + Tl; for the percentage within these sums we follow ETSU (1996).
^{a} assuming that 20% of Cr from incinerators is Cr^{VI}

the extraction of leachates under a landfill. Fly ash must be stored in specially designed sites for toxic waste, or else it must be stabilized by vitrification or by incorporation into concrete; the resulting health impacts are entirely negligible. Slag is considered sufficiently harmless to be used as construction material, for example for roads, after it has been treated and stabilized.

Landfill emissions to soil are difficult to estimate because they depend on the integrity of the liners in the future. If the landfill is operated according to regulations and if there are no leaks or other mishaps, there are no such impacts during the foreseeable future because the operator has the obligation to maintain and safeguard the facility for 30 years after closure. In any case the impacts would remain limited to the immediate vicinity of the landfill, with the possible exception of sites with large ground water movement.

The quantification of damage costs due to leachates from landfills involves considerations very different from pollutants emitted into the air. No complete IPA has been carried out until now because there are so many difficulties. A few studies have attempted to determine external costs of leachates, but they are not based on an IPA; instead they report the results of contingent valuations (CV) where interviewees are asked how much they are willing to pay to avoid the pollution of their drinking water by leachates [see the review by COWI (2000)]. As the interviewees had no quantitative information on the health impacts, the answers are a lump sum that has no relation to the real damage. Due to such a lack of informa-
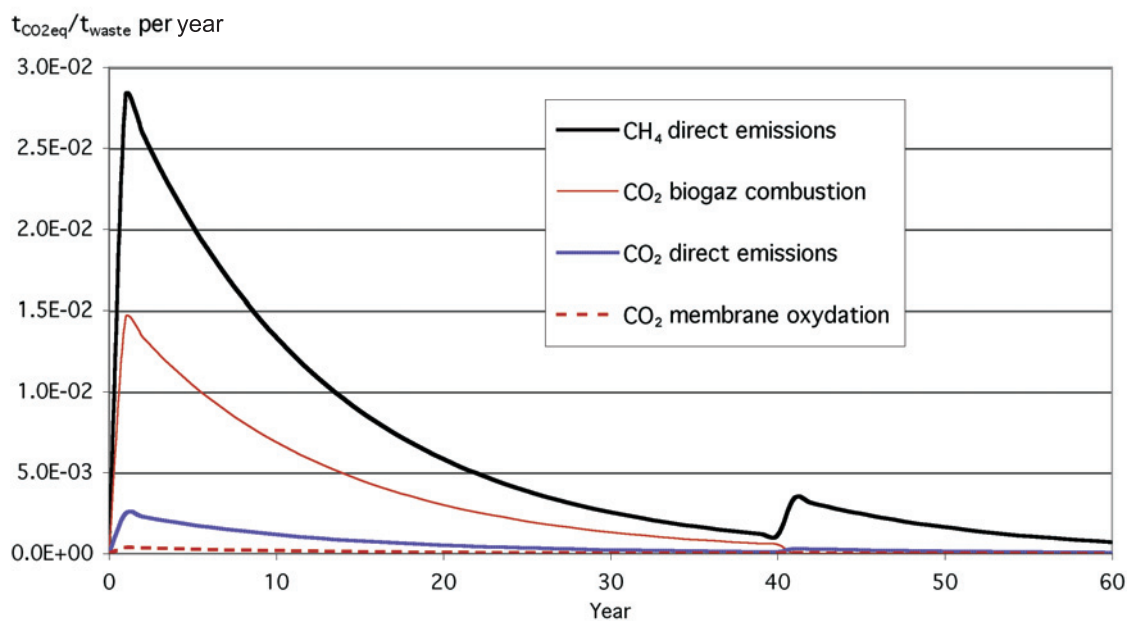

Fig. 2: Greenhouse gas emissions from a municipal solid waste landfill versus time. t$_{CO2eq}$ t$_{waste}$$^{-1}$ if 70% of the CH$_4$ is captured. Based on ADEME (2003).
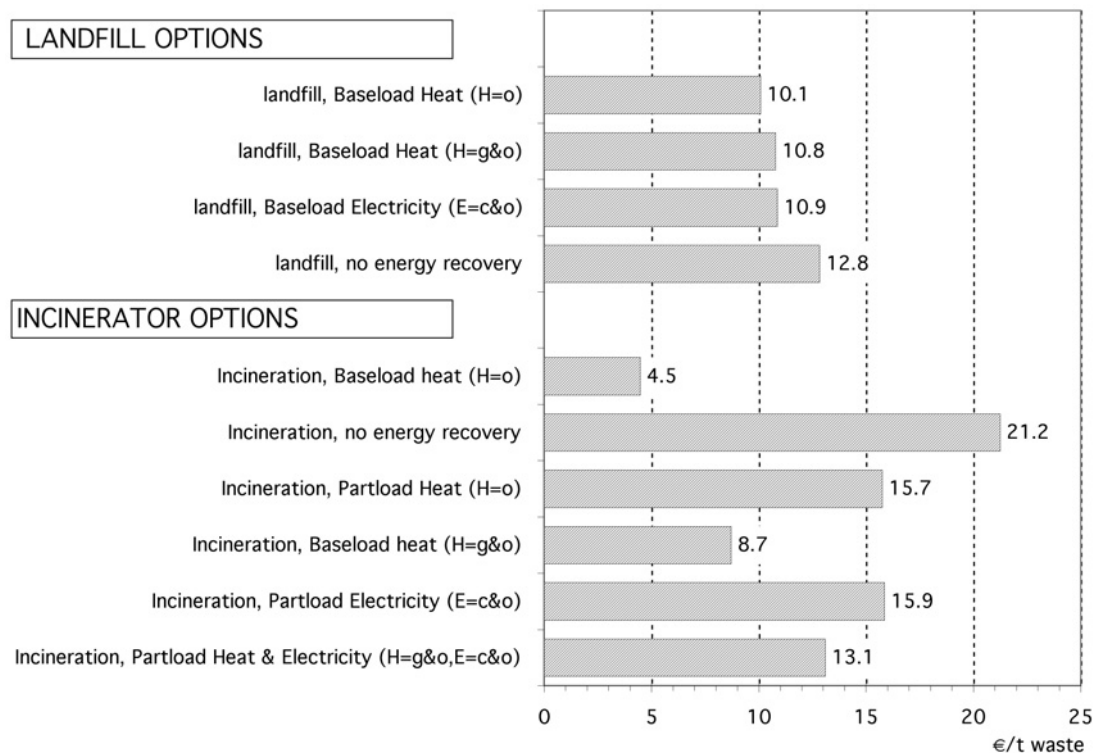
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Fig. 3: Results for total damage cost for all options. If electricity displaces nuclear, damage costs are essentially the same as for the case without energy recovery. Amenity costs are not included; they are very site-specific and could make a contribution on the order of $1 \in t_{waste}^{-1}$.

tion, CV is not an appropriate instrument for this type of impact (and even if one could provide such information, the variety of possible health impacts would be too detailed and complicated for a typical citizen to indicate a meaningful willingness-to-pay).

The problems confronting an IPA of leachates are more fundamental than the formidable technical difficulties of modelling the pathways through the environment into the drinking water. Many of the pollutants have very long life times, and in particular the toxic metals stay in the environment for ever (although only a fraction may become available for intake). Even if the pathways could be analysed in a satisfactory manner, there is no clear solution for the choice of time horizon and discount rate. As emphasized by Rabl (1996), even if there were agreement on the appropriate discount rate, another rate is just as important: the rate at which the damage costs change over time (before discounting a future cost, one must predict what that cost will be). For example, if the progress of medicine renders cancers as harmless as the common cold, it would be absurd to estimate the future damage according to today's assumptions. Thus the quantification involves necessarily subjective judgments about the progress of science and medicine, in addition to subjective assumptions about the future management and integrity of the landfill.

In view of these difficulties we do not attempt an IPA of leachates. Instead we offer an argument of a completely different nature to show that the impacts of leachates are negligible if a landfill conforms to current regulations. In compar-

ison with an IPA (which would be for a specific site and, because of the extreme site-dependence difficult to generalize to typical results needed by most policy applications) such an argument carries the further advantage of being general rather than site-specific.

The argument involves comparisons of a dose from leachates with the dose of the same pollutant from ordinary drinking water. Among inorganic pollutants we choose Pb and As for such a comparison because as a metal their time horizon is unlimited and because among the inorganic pollutants in leachates they have the highest damage cost per kilogram (apart from Hg, discussed in the next section), according to estimates by the ExternE team (Spadaro & Rabl 2004, ExternE 2004). Among organics we look at benzene, because its high carcinogenicity implies a large damage cost.

At the present time the review by Kjeldsen et al (2002) is probably the best source of data on pollutant concentrations in leachate. Data on metal concentrations can be found in Tables 1 and 5 of that reference. Table 1 summarizes values from 14 articles, nine of which date from before 1990; for Pb it shows a range of 0.001 to 5 mg L_{leach}^{-1} per litre of leachate (L_{leach}), and for As a range of 0.01 to 1 mg L_{leach}^{-1}. Table 5 presents measurements at a large number of landfills, published since 1995; for Pb the highest value is 0.188 mg L_{leach}^{-1} and most are much lower (none are shown for As). For example, the average for 106 old Danish landfills is 0.07 mg L_{leach}^{-1}, and the values in 21–30-year-old German landfills range from 0.005 to 0.019 mg L_{leach}^{-1}. EPA states a range of 0.008 to 1.02 mg Pb L_{leach}^{-1}, with a mean estimate of 0.09 mg Pb L_{leach}^{-1};

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Table 5: Concentrations in leachate and drinking water under the extreme scenario where untreated leachate is used as drinking water.

Pollutant	Leachate, range	Leachate, average	Leachate, EPA limit	Drinking water, extreme scenario, upper range	Drinking water, extreme scenario, EPA limit	Drinking water, EU directive
Units	mg L^{-1}	mg L^{-1}	mg L^{-1}	μg L^{-1}	μg L^{-1}	μg L^{-1}
As	0.01–1 [a]		0.05	40	2	10
Pb	< 0.005–0.188 [b]		0.05	7.5	2	10
Benzene	< 0.001–1.63 [c]	0.007 [d]	0.005	65	0.2	1

[a] Table 1 of Kjeldsen *et al*.
[b] Table 5 of Kjeldsen *et al*.
[c] Table 7 of Kjeldsen *et al*.
[d] Average for new landfills, in Table 8 of Kjeldsen *et al*.

Lee *et al*. (1993) report a mean value of 0.5 mg Pb L$_{leach}$$^{-1}$ and a range of 0.1 to 1 mg Pb L$_{leach}$$^{-1}$.

To obtain an upper bound on the resulting concentration in drinking water let us consider an extreme scenario of total liner failure by assuming that all the leachate passes directly into the water supply, without any filtering. For a convenient way of analysing this case, we consider waste production and water consumption per person. We assume a waste density of 800 kg/m^{-3} and a production rate of MSW of 500 kg year^{-1} person^{-1} during 30 years. If the ultimate stacking height of the waste is 10 m, the area used is 1.9 m^2 person^{-1}. For an upper limit on the leachate production rate let us assume that all the precipitation enters the landfill and, after percolating through the waste, passes into the drinking water of the population using the landfill; in other words, we assume total failure of the cover above and of the geomembrane below the waste. In a climate with an average precipitation of 3 L m^{-2} day^{-1} (= 1.1 m year^{-1}) the resulting leachate production is 6 L$_{leach}$ day^{-1} person^{-1}. Since the average household water consumption is around 150 L day^{-1} person^{-1}, the leachate will necessarily be diluted, by a factor of 6/150. Taking the upper limit of the Pb concentration range in Table 5 of Kjeldsen *et al*., the resulting concentration is 188 × (6/150) μg L^{-1} = 7.5 μg L^{-1}. That is lower than the limit of 10 μg L^{-1} allowed by the most recent Water Quality Directive of the EU. For as this argument implies a concentration of 40 μg L^{-1} in the water supply, compared to the regulatory limit of 10 μg L^{-1}, if one takes the upper range of 1 mg L$_{leach}$$^{-1}$ of leachate concentration in Table 1 of Kjeldsen *et al*. (which is probably well above what would be found in newer landfills).

The leachates also contain significant amounts of organic pollutants, for instance solvents. A particularly troubling pollutant is benzene because of its high concentration and because it is highly carcinogenic. Table 6 of Kjeldsen *et al*. shows a leachate concentration range of 0.0002 to 1.63 mg L$_{leach}$$^{-1}$. The aver-

Table 6: Assumptions for energy recovery.

kWh t$_{waste}$$^{-1}$	Electricity	Heat
Part load heat and electricity	202	607
Base load electricity only	270	0
Base load heat	0	1850

Table 7: Assumptions for recovery rates and avoided damage costs for materials recovery from incinerators.

	kg t$_{waste}$$^{-1}$	€ kg^{-1}	€ t$_{waste}$$^{-1}$
Slag	230	0	0
Iron	20.2	0.073	1.49
Aluminum	1.5	0.39	0.58

age values are shown in Table 7 of Kjeldsen *et al*.; for American landfills; they are 0.065 mg L$_{leach}$$^{-1}$ for old and 0.007 mg L$_{leach}$$^{-1}$ for new landfills. The regulatory limit of the most recent water quality directive of the EU is 1 μg L^{-1}, and that is satisfied by our extreme scenario if the leachate concentration is below 0.025 mg L$_{leach}$$^{-1}$, which is certainly the case for the average of new landfills according to Table 8 of Kjeldsen *et al*.

Even though the upper range of the leachate concentrations we have considered would imply some violations of the water quality directive under our extreme scenario, we emphasize that the scenario where the water supply is drawn directly from the leachate is unrealistically pessimistic. In reality the water would certainly be treated and well over 90% of the pollutants would be removed. We note also that in the USA the EPA groundwater protection performance standards require the leachate concentrations of landfills to be below 0.05 mg L$_{leach}$$^{-1}$ for As and Pb and below 0.005 mg L$_{leach}$$^{-1}$ for benzene; at these values the water quality directive would be satisfied by a wide margin. Furthermore, measured leachate production rates are far lower than the extreme scenario of 3 L$_{leach}$ m^{-2} day^{-1} assumed above. For active landfills they range between 0.0005 to 0.02 L$_{leach}$ m^{-2} day^{-1} [Morton Barlaz, http://www.epa.gov/epaoswer/non-hw/muncpl/landfill/bio-work/barlaz-a.pdf]. For covered landfills, the leachate rate is up to two orders of magnitude lower than that.

Therefore we conclude that leachate from MSW landfills is not a problem to worry about, certainly not if the EPA regulations are respected: there are other sources for the ingestion of toxic metals that deserve greater attention. Whereas it is certainly conceivable, even probable, that future regulations for drinking water will be stricter, one should note that one can also expect a decrease of the concentrations in the leachate because of stricter regulations affecting the composition of waste (e.g. by limiting the toxic metal content of bat-

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teries for consumer goods). The numbers of this section are summarized in Table 5.

Mercury

MSW contains appreciable quantities of Hg, from sources such as fluorescent light bulbs, thermometers and batteries (before the transition to Hg-free batteries during the late 1990s). For example the State of Florida has estimated the amount of Hg in MSW of Florida around the year 2000 (Florida 2002), as part of a detailed examination of data on the composition of waste, and the numbers imply a concentration of about 0.25 g Hg t_{waste}^{-1}.

Calculating the damage cost of Hg emissions is extremely difficult and uncertain because of the complexities of the environmental pathways and the lack of sufficient information and data. Only recently has a consensus been emerging among epidemiologists on the magnitude of IQ loss, the best established end point of low chronic exposures to Hg and probably the one with the highest damage cost (e.g. Trasande *et al*. 2005). The toxicity of Hg depends very strongly on its chemical form; methyl-Hg is by far the most toxic. Hg vapour itself is not very toxic, but a small fraction gets converted to methyl-Hg by aquatic organisms and enters the aquatic food chain where bioconcentration leads to relatively high concentrations in fish.

Rice & Hammitt (2005) have estimated the health benefits to be expected in the USA if the proposed new regulation of Hg emissions from power plants is implemented. However, they do not consider the benefits in other countries, benefits that can be a large fraction of the global benefit because Hg vapour has a long residence time in the atmosphere, about 1 to 2 years, and its impact is therefore imposed on the entire hemisphere. For that reason Spadaro and Rabl (2007b) have provided an estimate of the global average damage cost of atmospheric Hg emissions. Their estimate is based on the relation between total global emission and global average ingestion dose of methyl-Hg per person, and therefore it does not account for the possibly large variation with the region where it is emitted. Not having the means for a detailed site-specific IPA of Hg, we use the result of Spadaro and Rabl who find a global average damage cost of about 8000 € kg^{-1} Hg emitted into the air.

Multiplying the Hg concentration in MSW by this damage cost, one finds a cost of

$$0.25 \text{ g Hg } t_{waste}^{-1} \times 8000 \text{ € } kg_{Hg}^{-1} = 2.0 \text{ € } t_{waste}^{-1}$$

if all the Hg escapes into the air (this emission rate is very close to the EC (2000a) limit value of 0.26 g Hg t_{waste}^{-1} in Table 4). That is the case for incineration, unless some of the Hg is removed by active carbon; a technology already used in some incinerators.

For landfills, by contrast, the release rate is quite slow, on the order of 0.1 to 1% per year during the initial operation of a landfill, and the rate seems to slow down even more after closure of the landfill (as expected if the cover is reasonably tight), as indicated by the measurements of Lindberg *et al*. (2005). [In passing we note that Lindberg *et al*. found that a significant fraction of the atmospheric emissions from landfills are in the form of methyl-Hg; however, the resulting inhalation dose is small compared to the average ingestion dose from fish (UNEP 2002), even near the landfill.] Even if the entire Hg would eventually get into the environment, the damage cost would be much lower because of discounting. In view of this situation we conclude that Hg from landfills may be a significant problem that deserves attention, although it is unlikely to make a dominant contribution to the total damage cost.

Recovery of energy and materials

The external costs and the comparison between landfill and incineration turn out to be extremely sensitive to assumptions about energy recovery. For that reason we consider a fairly large number of options, for typical installations in France, according to ADEME (2000). We indicate the options with labels where the letters E and H refer to heat and electricity and the letters c, g, n and o to the fuel (coal, gas, nuclear and oil, respectively) displaced by energy recovery. For example (E = …, H = …) (E = c&o, H = c&o) designates a system where heat and electricity are produced, each displacing a fuel mixture of coal and oil, 50% of each. The options are

for incineration:
- recovery of heat and electricity, (E = …, H = …),
- recovery of electricity only (E = …),
- recovery of heat only (H = …);

for landfill:
- no energy recovery,
- recovery of electricity, by motor (reciprocating engine) (E = …),
- recovery of electricity, by turbine (E = …),
- recovery of heat (H = …).

For each of these options we consider several suboptions:
- the recovered electricity displaces coal and oil fired power plants, 50% each (E = c&o),
- the recovered electricity displaces nuclear power plants (E = n),
- the recovered heat displaces gas and oil fired heating systems, 50% each (H = g&o),
- the recovered heat displaces only oil fired heating systems (H = o).

For recovery of electricity we assume a year-round demand, so all the electricity is used. Likewise for recovery of heat we assume a year-round demand (industrial process heat loads or certain district heating systems with year-round demand, e.g. Paris and Vienna), so none of the heat is wasted. Year-round demand is essential for good recovery rates because the supply of waste tends to be fairly constant. For other load distributions the results can be estimated by rescaling the numbers for energy recovery (see Figure 4).
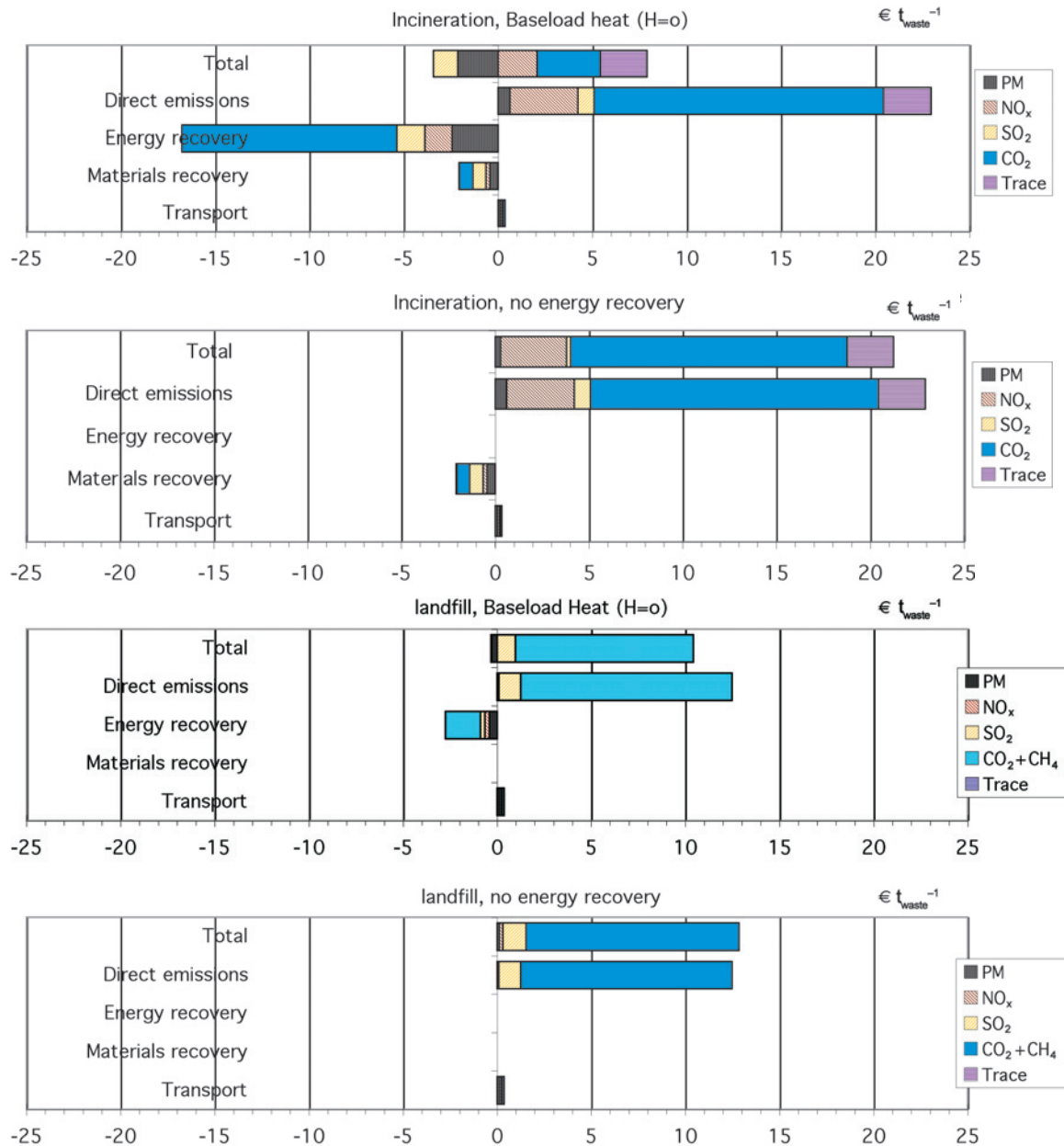
Fig. 4: Some detailed results, by stage and pollutant. 'Trace' = dioxins and toxic metals.

Note that for the purpose of this analysis the benefit of recovered electricity is essentially zero if it displaces nuclear because the damage costs of nuclear are very small compared to those of oil or coal; thus this option is essentially equivalent to no electricity production at all as far as damage costs are concerned. Our assumptions for energy recovery rates are shown in Table 6. The emissions avoided are based on the Large Combustion Plant Directive (EC 2001).

Recovery of materials is practical only for incinerators, not for landfills. Our assumptions for recovery rates and avoided damage costs are listed in Table 7; the damage costs avoided are based on the LCA inventory of Delucchi (2003).

Results for damage cost per tonne of waste

A summary of the total damage cost for all the options is shown in Figure 3. More detailed results for some of the options can be found in Figure 4, showing the contribution of each stage and of the major pollutants (dioxins and toxic metals are shown as 'Trace'). The benefits of materials recovery make a small or negligible contribution to the total damage cost. The damage costs of waste transport, illustrated with an arbitrary choice of 100 km round trip by a 16 tonne truck, are also negligible. The only significant contributions come from direct emissions (of the landfill or incinerator) and energy recovery.

The total cost of incinerator emissions is 22.9 $€$ t_{waste}^{-1} as can be seen by adding the last column of Table 4. Most of that is due to PM, NO_x, SO_2 and CO_2. Toxic metals and dioxins, shown in Figure 4 as 'Trace', contribute only 2.5 $€$ t_{waste}^{-1}, mostly because of Hg and Pb. The contribution of dioxins is negligible, only 0.1 $€$ t_{waste}^{-1}, thanks to the low emission limit of the Directive EC (2000a).

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For landfill the cost is dominated by greenhouse gas emissions because only about 70% the CH_4 can be captured. Energy recovery from a landfill is not very significant (and because of NO_x from the electricity production, this option increases the damage cost if the electricity displaces nuclear). By contrast, energy recovery is crucial for the damage cost of incineration. Under favourable conditions (all heat produced by incinerator displaces highly polluting coal and oil) the total external cost could even be negative, i.e. a net benefit. By contrast to most other countries, in France recovery of electricity does not bring significant benefits, because it is base load power and all the base load power is produced by nuclear; the options where it displaces coal or oil are not realistic in France (except near the border where the power can be exported) because these fuels are used only during the heating season. In any case, electricity production brings far lower benefits than heat because of the poor conversion efficiency of incinerator heat (compared to central station power plants).

Conclusions

We have evaluated and compared the damage costs ('external costs') of landfill and incineration of MSW, based on the latest results of ExternE (2004) and taking into account the relevant life cycle impacts, especially emissions avoided by recovery of energy and materials. The damage cost of incineration ranges from about 4 to +21 € t_{waste}^{-1}, depending on the assumptions about energy recovery. The damage cost of landfills, around 10 to 13 € t_{waste}^{-1}, is mostly due to greenhouse gases, evaluated here with a unit cost of 19 € t_{CO2}^{-1} according to ExternE (2004). In addition there may be amenity costs with an order of magnitude of 1 € t_{waste}^{-1} (highly variable with site and imposed only on the local population, thus to be internalized differently from air pollution). Unlike incinerators, the damage cost of landfill does not vary as much with type of energy recovery because in any case the amount recovered is relatively small.

The benefits of energy recovery from incinerators are largest if the heat can be used directly for process heat or district heating systems with sufficiently large constant load. Electricity production brings far lower benefits than heat because of the poor conversion efficiency of incinerator heat (compared to central station power plants).

The results presented in this paper are for typical conditions in France, but they can be adapted to other sites and other countries if the respective damage costs per emitted pollutant are known. Even without carrying out new calculations using the EcoSense software, one can estimate the damage costs per emitted pollutant using the UWM referred to above together with the simple rules for site variation of given in the section entitled 'Cost per kilogram of pollutant'.

The uncertainties are large (see Table 2) and they have different effects on different policy choices. Comparisons between landfill and incineration are especially sensitive to the uncertainty of greenhouse gases because they play such a large role for landfills. Comparisons between different types of energy recovery for incinerators, on the other hand, also depend on the other pollutants. The emission of greenhouse gases per tonne of waste depends strongly on the type of sorting and pretreatment that is carried out. For example, if paper, cardboard and plastics are removed, the biological fraction of the remaining waste increases and so do the CH_4 emissions from landfill.

In decisions about waste treatment the full social cost (= sum of private costs and external costs) should be taken into account. We have not examined the private costs but refer to Dijkgraaf and Vollebergh (2004) whose Table 2 shows the private costs in the Netherlands. The private costs of landfilling are less than half of those of incineration: for landfilling 40 € t_{waste}^{-1} without and 36 € t_{waste}^{-1} with energy recovery, versus 103 € t_{waste}^{-1} without and 79 € t_{waste}^{-1} with energy recovery for incineration. Since their estimate of the external costs [based on CE (1996)] also indicates lower social costs for landfilling, they question the wisdom of current waste policies in the EU that favour incineration. Our external costs are consistent with theirs, and we agree with their conclusions.

However, we emphasize that costs (private costs + damage costs) are not the only criterion for choosing a treatment option for MSW. There are almost always additional criteria, especially the preferences of the local population, which may be difficult or impossible to express in monetary terms. Land use and land availability are crucial. In many regions of Europe land is so limited that incineration is the preferred choice even if its cost is higher. Such non-monetary criteria can be taken into account by means of a multicriteria analysis, preferably in consultation with the stakeholders, an approach that has been successfully tested at a stakeholder workshop in the SusTools project of the EC (Rabl *et al.* 2004) where the results of this paper were discussed with policy makers, representatives of industry and environmental organizations.

Acknowledgements

Glossary and nomenclature

As	arsenic.
CO_{2eq}	quantity of a greenhouse gas expressed as equivalent quantity of CO_2, using the GWP of the gas.
Cr^{VI}	chromium in oxidation state 6.
CRF	concentration–response function.
CV	contingent valuation.
Discount	rate r that allows comparison of monetary values
	incurred at different times, defined such that an
rate	amount P_n in year n has the same utility as an amount $P_0 \rightarrow P_n(1 + r)^{-n}$ in year 0.
DRF	dose–response function.
ERF	exposure–response function.
EC	European Commission.
EPA	Environmental Protection Agency of USA.

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External costs	costs that arise when the social or economic acti-vities of one group of people have an impact on another for which the first group does not fully account; e.g when a polluter does not compensate others for the damage imposed on them.
GWP	global warming potential.
Hg	mercury.
IPA	impact pathway analysis.
ISC	Industrial source complex Gaussian plume dispersion model.
LCA	life cycle assessment.
N	nitrogen.
Ni	nickel.
NO_x	unspecified mixture of NO and NO_2.

O_3	ozone.
Pb	lead.
PM_d	particulate matter with aerodynamic diameter smaller than d μm.
S	sulfur.
s_{CR}	slope of concentration-response function [cases (person year μg m^{-3})$^{-1}$].
UWM	uniform world model (for simplified estimation of damage costs).
v_{dep}	deposition velocity (m s^{-1})].
VOC	volatile organic compounds.
VOLY	value of a life year.
VPF	value of prevented fatality.
WTP	willingness to pay.
YOLL	years of life lost (reduction of life expectancy).

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