Environmental Impacts and Costs of Energy

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4.1 Introduction

Air pollution causes considerable damage to human health, flora and fauna, and materials. The damage costs are externalities to the extent that these costs are not reflected in the prices of goods. When making decisions that affect the emission of pollutants, the damage costs, also called external costs, should be considered. The external costs can be internalized via taxes, tradable permits, or other environmental regulations..

In recent years, there has been much progress in the analysis of environmental damage costs due mostly to several major projects evaluating the external costs of energy in Europe (ExternE 1995, 1998, 2000, 2004) and also in the United States (Lee 1994; Rowe et al. 1995). Of these projects the ExternE (External Costs of Energy) Project of the European Commission has the widest scope and is the most up to date. This paper, authored by participants of all ExternE Projects since 1992, presents an overview of the methodology and the results.

The quantification of damage costs has many important applications:

 Guidance for environmental regulations (e.g., determining the optimal level of the limit for the emission of a pollutant)

- Finding the socially optimal level of a pollution tax
- Identifying technologies with the lowest social cost (e.g., coal, natural gas, or nuclear power for the production of electricity)
- Evaluating the benefits of improving the pollution abatement of an existing installation such as a
 waste incinerator
- Optimizing the dispatching of power plants
- "Green accounting," i.e., including corrections for environmental damage in the traditional accounts of GNP

4.2 Methodology

4.2.1 Impact Pathway Analysis

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 as follows, as shown in Figure 4.1:

1. Emission: specification of the relevant technologies and pollutants, e.g., kg of NO_x per GWh_e emitted by power plant

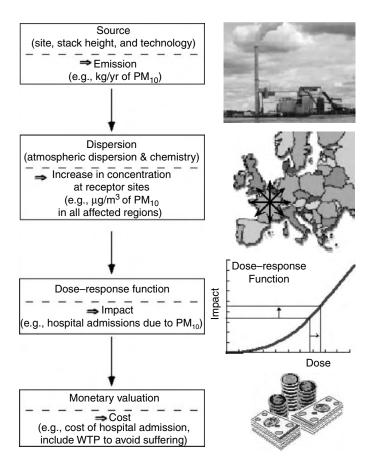


FIGURE 4.1 Impact pathway analysis.

- 2. 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 (DRF), e.g., cases of asthma due to this increase in ozone
- 4. Cost: economic 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. The work involves a multidisciplinary system analysis, with inputs from engineers, dispersion modelers, epidemiologists, ecologists, and economists. The result of an IPA is the damage cost per kg of emitted pollutant, as shown in Figure 4.5 of Section 4.3. The steps of the IPA are described in the following sections.

The reader may wonder about the relationship between an IPA and an environmental impact study (EIS) that is required before the approval of a proposed installation (factory, power plant, incinerator, etc.). The purpose of an EIS is to ensure that nobody is exposed to an unacceptable risk or burden. Because the highest exposures are imposed in the local zone, it is sufficient for an EIS to focus on local analysis, up to 10 km depending on the case. Thus, an EIS provides the possibility of a veto if a proposed installation is considered unacceptable. By contrast, the calculation of total damage costs requires an IPA where the damages are summed over all affected receptors. Regarding most air pollutants emitted in Europe the affected receptors are the entire continent, and in the case of greenhouse gases (GHGs) it is the entire globe. Damage costs are needed primarily by decision makers at the national or international level, or generally by anyone concerned with total impacts.

For many environmental choices, one needs to look not only at a particular source of pollutants but also must take into account an entire process chain by means of a life cycle assessment (LCA). For example, a comparison of power generation technologies involves an analysis of the fuel chain sketched in Figure 4.2. Whether an IPA of a single source or an LCA of an entire cycle is required depends on the policy decision in question. When finding the optimal limit for the emission of NO_x from an incinerator, an IPA is sufficient, but the choice between incineration and landfill of waste involves an LCA.

In principle, a site-specific IPA would evaluate the damages and costs for each pollution source in the life cycle. In practice, however, most LCAs have taken the shortcut of first summing the emissions over all stages and then multiplying the result by site-independent impact indices. Additionally, most practitioners of LCA reject the concept of monetary valuation, preferring instead to use approximately ten nonmonetary indicators of "potential impact" that are based on expert judgment.

4.2.2 Dispersion of Pollutants

The principal GHGs, 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. However, the calculation of impacts is extraordinarily complex and the reader is referred 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} (particulate matter with diameter less than $10~\mu m$), NO_x and SO_2 , atmospheric dispersion is significant over hundreds to thousands of km, so both local and regional effects are important. ExternE uses 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 and 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 when studying the creation of sulfates from SO₂, ExternE uses the Windrose trajectory model (WTM) (Trukenmüller and 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 and Nodop 1986). The modeling of ozone is based on the EMEP MSC-W oxidant model

→ Real impacts for each stage (site specific)
Goal: evaluate the entire matrix

$ \begin{array}{c} \text{Steps of impact pathway analysis} \\ \rightarrow \end{array} $	Emission	Dispersion	Exposure response function	Economic valuation
Stage of fuel chain ↓				
Fuel extraction				
Fuel transport				
Power plant				
Transmission of electricity				
Management of wastes				

Life cycle assessment:	First sum over emissions	
	↓ Then	
	$\Sigma \longrightarrow \times$ Multiplication by	
	"potential impact" indices	

FIGURE 4.2 Relation between impact pathway analysis and current practice of most LCA, illustrated for the example of electricity production. (From Spadaro, J. V. and Rabl, A. 1999. *International Journal of Life Cycle Assessment*, 4(4), 229–243. With permission.)

(Simpson 1992; Simpson and Eliassen 1997). EMEP is the official model used for policy decisions about transboundary air pollution in Europe.

The calculation of damage costs is carried out by using 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.), DRFs and monetary values. J. V. Spadaro also has developed a simplified analysis tool called RiskPoll (actually a package of several models with different input requirements) that is freely available from www.arirabl.org or www.externe.info. This tool is based on the interpolation of dispersion calculations by EcoSense. Its simplest version yields results 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 4.3.

Several tests have been done to confirm the accuracy of the results. The tests have checked the consistency between ISC and ROADPOL, and the concentrations predicted by WTM were compared with measured data and with calculations of the EMEP program, the official program for the modeling of acid rain in Europe.

Whereas only the inhalation dose matters for PM_{10} , NO_{∞} , SO_2 and O_3 , toxic metals and persistent organic pollutants affect humans also through food and drink. These pollutants require a much more complex IPA to calculate ingestion doses. Spadaro and 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. The model takes into account the pathways in Figure 4.3. The output of this model is the damage per kg of pollutant, as a function of the site and conditions (for emissions to air: stack height, exhaust temperature, and velocity) of the source. The model 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, like the uptake by agricultural crops of a pollutant from the soil. The uncertainties are large, but at least approximate values for the pollutants of concern are available.

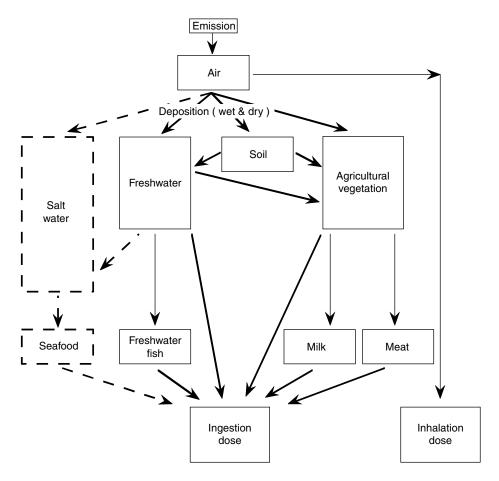


FIGURE 4.3 Pathways taken into account for health impacts of air pollutants. Direct emissions to soil or water are a special case where the analysis begins at the respective "soil" and "water" boxes. In the present version of the model seafood is not yet included.

It is not yet possible to have all of the elements for calculating the dose due to ingestion of seafood. The dose may be potentially large because of bioconcentration and the fact that most fish are oceanic rather than freshwater. Even if the concentration increment in the sea is very small, the collective dose from seafood could be significant if the removal processes (sedimentation) are slow, and the analysis has no cutoff in time.

A general result of this analysis is that when these pollutants are emitted into the air, the ingestion dose can be approximately two orders of magnitude larger than the dose by inhalation. Because nowadays most food is transported over very large distances, the total dose varies little with the site where these pollutants are emitted into the air. As far as damages are concerned, one has to note that the same 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 carcinogenic only through inhalation according to current knowledge.

4.2.3 Dose-Response Functions: General Considerations

The 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 narrowest sense of the term, it should be based on the dose actually absorbed by a receptor. However, the term DRF 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 this kind, and the terms exposure–response function or concentration–response function (CRF) are often used.

The DRF is a central ingredient in the IPA, thus meriting special attention. Damage can be quantified only if the corresponding DRF is known. Such functions are available for the impacts on human health, building materials, and crops caused by a range of pollutants such as primary and secondary (i.e., nitrates, sulfates) particles, ozone, CO, SO₂, NO₃, benzene, dioxins, As, Cd, Cr, 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 the application in an IPA, the information often has to be expressed in somewhat different form, due to accounting for additional factors such as the incidence rate (ExternE 1998; Spadaro and Rabl 2004). Unfortunately, the DRFs are very uncertain or not known for many pollutants and many impacts. For most substances and noncancer 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).

By definition, a DRF starts at the origin, and in most cases it increases monotonically with dose, as sketched schematically in Figure 4.4. At very high doses, the function may level off in S-shaped fashion due to saturation, but that case is not of interest here. DRFs for health are determined from epidemiological studies or laboratory studies. Because the latter are mostly limited to animals, the extrapolation to humans introduces large uncertainties.

A major difficulty lies in the fact that unless the sample is very large, relatively high doses are needed to obtain observable nonzero responses; such doses are usually far in excess of typical ambient concentrations in the EU or North America. Thus there is a serious problem of how to extrapolate from the observed data towards low doses. Figure 4.4 indicates several possibilities for the case where the point P corresponds to the lowest dose at which a response has been measured. The simplest is the linear model, i.e., a straight line from the origin through the observed data point(s). The available evidence suggests that a DRF is unlikely to go above this straight line in the low dose limit. However, the straight-line model does appear to be appropriate in many cases, especially for many cancers. In fact, most estimates of cancers due to chemicals or radiation assume this linear behavior.

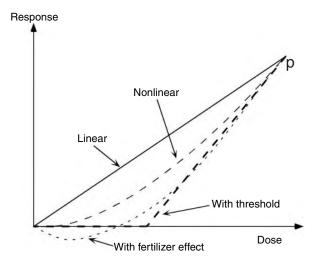


FIGURE 4.4 Possible behavior of dose–response functions at low doses. If *P* is the lowest dose where a nonzero impact has been observed, the extrapolation to lower doses is uncertain but values higher than linear are unlikely.

Another possibility is the "hockey stick": a straight line down to some threshold, and zero effect below that threshold. Thresholds occur when an organism has a natural repair mechanism that can prevent or counteract damage up to a certain limit.

There is even the possibility of a "fertilizer effect" at low doses, as indicated by the dashed line in Figure 4.4. This effect can be observed in the DRFs for the impact of NO_x and SO_2 on crops: a low dose of these pollutants can increase the crop yield. In other words, the damage is negative. Generally a fertilizer effect can occur with pollutants that provide trace elements needed by an organism.

In practice, most DRFs used by ExternE, particularly all the ones for health, are assumed to be linear (without threshold). However, for the calculation of incremental damage costs, there is no difference between the linear and a hockey stick function with the same slope, if the background concentration is everywhere above this threshold; only the slope matters. For the particles, NO_x, SO₂, O₃, 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-effect threshold, it is below the background concentrations of interest.

4.2.4 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 and Spengler 1996; ERPURS 1997; the reports of the World Health Organization, available at http://www.who.int, see Table 4.1). There is less certainty about specific causes, but most recent studies have identified fine particles as a prime culprit; ozone has also been directly implicated. 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

TABLE 4.1 Air Pollutants and Their Effects on Health

Primary Pollutants Secondary Pollutants		Impacts
Particles (PM ₁₀ , PM _{2.5} ,		Mortality
black smoke)		Cardiopulmonary morbidity (cerebrovascular hospital admissions, congestive heart failure, chronic bronchitis, chronic cough in children, lower respiratory symptoms, cough in asthmatics)
SO_2		Mortality
_		Cardiopulmonary morbidity (hospitalization, consultation of doctor, asthma, sick leave, restricted activity)
SO_2	Sulfates	Like particles?
NO_x		Morbidity?
NO_x	Nitrates	Like particles?
$NO_x + VOC$	Ozone	Mortality
		Morbidity (respiratory hospital admissions, restricted activity days, asthma attacks, symptom days)
CO		Mortality (congestive heart failure) Morbidity (cardiovascular)
PAH (diesel soot, benzene, 1,3-		Cancers
butadiene, dioxins) As, Cd, Cr ^{VI} , Ni		Commen
As, Cu, Cf , NI		Cancers Other markidity
II pl.		Other morbidity
Hg, Pb		Morbidity (neurotoxic)

comes from chronic bronchitis due to particles (Abbey et al. 1995). In addition, there may be significant direct health impacts of SO₂, but for direct impacts of NO_x, the evidence is less convincing.

In ExternE, the working hypothesis has been to use the DRFs for particles and for O_3 as basis. Effects of NO_x and SO_2 are assumed to arise indirectly from the particulate nature of nitrate and sulfate aerosols, and the effects calculated by applying the particle DRFs to these aerosol concentrations. But the uncertainties are large because there is insufficient evidence for the health impacts of the individual components or characteristics (acidity, solubility, etc.) of particulate air pollution. In particular there is a lack of epidemiological studies of nitrate aerosols because until recently air pollution monitoring stations did not monitor this pollutant. In view of the lack of evidence for thresholds at current ambient concentrations, all DRFs for health impacts have been assumed linear at the population level. By contrast to the homogeneous populations of cloned animals studied by toxicologists, the absence of a no-effect threshold is plausible for real populations because they always contain individuals with widely differing sensitivities (for example, at any moment about 1% is within the last nine months of life and thus extremely frail).

4.2.5 Monetary Valuation

The goal of the monetary valuation of damages is to account for all costs, market and nonmarket. 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. If the WTP for a nonmarket good has been determined correctly, it is like a price, consistent with prices paid for market goods. Economists have developed several tools for determining nonmarket costs; of these tools contingent valuation (CV) has enjoyed increasing popularity in recent years (Mitchell and Carson 1989). The basic idea of a CV is to ask people how much they would be willing to pay for a certain good if they could buy it. The results of well-conducted CV studies are considered sufficiently reliable.

It turns out that nonmarket goods dominate damage costs of air pollution, especially the valuation of mortality. The single most important parameter is the so-called "value of statistical life" (VSL). This term often evokes hostile reactions from people who think that economists try to measure the value of life. The value of life is limitless. To save an individual in danger no means are spared. In reality VSL is the "willingness to pay for avoiding a small risk of an anonymous premature death". In ExternE (2000), a European-wide value of 3.4 M€ was chosen for VSL, consistent with similar studies in the USA; this value was chosen as average of the VSL studies that had been carried out in Europe.

A crucial question for air pollution mortality is whether one should simply multiply the number of premature deaths by VSL, 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 VSL is based). In fact, for air pollution the appropriate measure is the value of a YOLL due to air pollution, called VOLY (value of a life year), whereas the true number of premature deaths due to pollution cannot even be determined, as shown by Rabl (2003).

There is considerable uncertainty because until recently there have been no studies to determine VOLY. ExternE (1998, 2000) calculated VOLY on theoretical grounds by considering VSL as the net present value of a series of discounted annual values. The ratio of VSL and the value of a YOLL thus obtained depends on the discount rate; it is typically in the range of 20–30. More recently ExternE carried out a CV study for VOLY and is now using a value of $50,000 \in$ for a year of life lost due to air pollution. As for cancers, ExternE assumes $0.45 \text{ M} \in$ for nonfatal cancers, and $1.5-2.5 \text{ M} \in$ for fatal cancers (depending on the YOLL for each cancer type).

4.2.6 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 also 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 cost per tonne of CO_{2eq} (the subscript eq indicates that the result can also be used for other GHGs if their masses are multiplied by their global warming potential (GWP)). Most of the results are in the range of 1–50 $\[\in \]$ / $t_{CO_{2eq}}$, the range being 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 \[\in \]$ / $t_{CO_{2eq}}$, the second, in 2000, obtained a much lower value of $2.4 \[\in \]$ / $t_{CO_{2eq}}$ because of more optimistic assumptions and a better accounting for benefits such as increased agricultural production in cold countries. The current phase of ExternE uses the value of $19 \[\in \]$ / $t_{CO_{2eq}}$ because that is the abatement cost in the EU implied by the commitment to the Kyoto Protocol. It 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. The choice of this value appears reasonable in view of the estimates published in the literature.

4.2.7 Other Impacts

Air pollution damage to materials and agricultural crops has been found to make a relatively small contribution to the damage costs, only a few percent of the total. Estimations of ecosystem impacts, other than agricultural losses, have remained extremely uncertain, if they have been attempted at all. Some estimates have been made of the costs of forest decline due to acid rain, but more recently doubts have been raised about their validity. In general there is a lack of information on ecosystem impacts and their economic valuation.

However, air pollution at typical ambient concentrations in the EU does not seem to have significant direct (i.e., not acid rain) impacts on ecosystems. At first glance this claim may appear surprising because human health impacts are significant and one might indeed expect similar impacts on animals as on humans. The explanation lies in what is valued: society values ecosystem impacts at the level of a population, human impacts at the level of the individual. Concentrations of air pollutants are generally so small that the incremental mortality is at most a small percentage of the natural rate. Furthermore, most of the deaths from air pollution occur among individuals well beyond reproductive age. If a small percentage of animals die prematurely after having produced and raised offspring, the effect on the ecosystem is negligible. But if any human dies prematurely, society cares a great deal.

The situation is different for certain aquatic impacts. Acidification of rivers and lakes has been shown to be detrimental to aquatic life. A river can collect much of the air pollution from a large region, leading to relatively high concentrations in the water, quite apart from direct emission of pollutants to water.

4.2.8 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 and Rabl (1996) and further developed, with detailed validation studies by Spadaro (1999), and Spadaro and Rabl (2002). More recently Spadaro and Rabl (2004) 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 IPA. It is exact for tall stacks in the limit where the distribution of either the sources or the receptors is uniform and the key atmospheric parameters do 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 \in /kg of a particular impact due to the inhalation of a primary pollutant is shown in Equation 4.1:

$$D_{\rm uni} = \frac{p s_{\rm CR} \rho}{v_{\rm dep}},\tag{4.1}$$

where p is the cost per case ("price") [\in /case], s_{CR} is the CRF slope [(cases/yr)/(pers·(μ g/m³))], ρ is the average population density [pers/km²] within 1000 km of source, and v_{dep} is the deposition velocity of pollutant (dry+wet) [m/s].

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 pollutant. 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.

Results for Cost per Kilogram of Pollutant 4.3

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 CO₂, CH₄, and N₂O, more than 95% of the costs is due to health impacts, especially mortality. Morbidity (especially chronic bronchitis but also asthma, hospital admissions, etc.) account for almost 30% of the damage cost of PM, NO_x, and SO₂. The impacts evaluated and the key assumptions are listed in Table 4.2. The resulting damage costs in €/kg of pollutant are shown in Figure 4.5 for typical sources with stack heights above 50 m in Central Europe. As for variation with site and stack height, the following rules can be recommended for modifying factors:

Impacts Evaluated and Key Assumptions TABLE 4.2

Atmospheric Dispersion Models

Gaussian plume model ISC (point sources) or ROADPOL Local range:

(emissions from transport)

Regional range (Europe): Harwell trajectory model as implemented in EcoSense software of ExternE

Ozone impacts based on EMEP model

Linearity without threshold, with slope S_{CR}

Physical Impacts

Impacts on Health

Chronic mortality

Impacts on plants

Form of dose-response functions

(DRF)

S_{CR}=4.1E-4 YOLL (years of life lost) per person per year per μg/m³ derived from increase in age-specific mortality due to PM_{2.5} [Pope et al 2002], by integrating

over age distribution

For SO₂ and ozone, assume 0.75 YOLL per premature death Acute mortality

Nitrate and sulfate aerosols Dose–response functions for nitrates: $50\% \times PM_{10}$

 $(PM_{10} \text{ slope}=60\% \times PM_{2.5} \text{ functions})$

Dose–response functions for sulfates same as for 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

Loss of crops due to SO2 and ozone

Impacts on buildings and materials

Corrosion and erosion due to SO2 and soiling due to particles Impacts not quantified Reduced visibility due to air pollution

but potentially significant 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}

Source: From ExternE. 2004. Project NewExt "New elements for the assessment of external costs from energy technologies." European commission DG Research, Contract No. ENG1-CT2000-00129. Coordinated by R. Friedrich, IER, University of Stuttgart. Final report. http://www.externe.info. With permission.

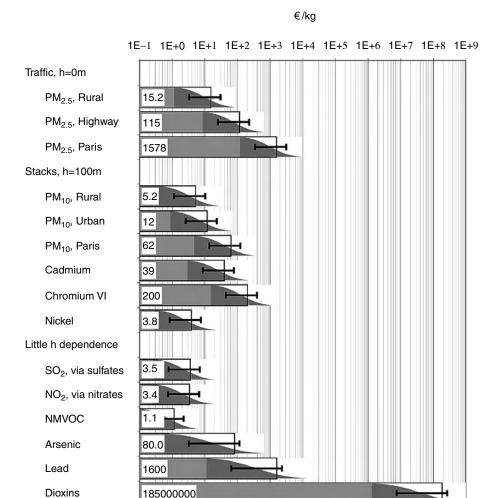


FIGURE 4.5 Results for damage costs of the most important air pollutants (typical values, for LCA applications in the EU15). The error bars indicate the 68% confidence interval; on the logarithmic scale, they are symmetric around the median (equal to the geometric mean of lognormal distribution). The broad hollow bars and the numbers are the mean that is larger than the median. The gray S-shaped curve indicates the probability that the true cost is above a specified value, h= stack height. (From ExternE, 2004. New results of ExternE, after the NewExt project, http://www.externe.info.)

- No variation for globally dispersing pollutants such as CO₂
- Weak variation for As, Pb, and dioxins because noninhalation pathways dominate: approximately 0.7–1.5
- Weak variation for secondary pollutants: approximately 0.5-2.0
- Strong variation for primary pollutants: approximately 0.5–5 for site, and approximately 0.6–3 for stack conditions (up to 15 for ground-level emissions in a large city)

Of course, such rules can only yield rough estimates; site-specific calculations should be carried out when more precise results are needed.

An analysis of the uncertainties is crucial for the credibility of the results. Uncertainties can be grouped into different categories, even though there may be some overlap:

- Data uncertainty (e.g., slope of a DRF, cost of a day of restricted activity, and deposition velocity of a pollutant)
- Model uncertainty (e.g., assumptions about causal links between a pollutant and a health impact), assumptions about form of a DRF (e.g., with or without threshold), and choice of models for atmospheric dispersion and chemistry
- Uncertainty about policy and ethical choices (e.g., discount rate for intergenerational costs, and "VSL")
- Uncertainty about the future (e.g., the potential for reducing crop losses by the development of more resistant species)
- Idiosyncrasies of the analyst (e.g., interpretation of ambiguous or incomplete information)

The first two categories (data and model uncertainties) are scientific in nature. They are amenable to analysis by statistical methods, combining the component uncertainties over the steps of the impact pathway, to obtain formal confidence intervals around a central estimate. For this an approach based on lognormal distributions and multiplicative confidence intervals was followed. The error bars in Figure 4.5 show the results of this analysis; they are one-geometric standard deviation intervals around the median estimate. The largest sources of uncertainty lie in the DRFs for health impacts and in the value of a life year. Details can be found in Rabl and Spadaro (1999).

Quantifying the sources of uncertainty in this field is problematic because of a general lack of information. Usually one has to fall back on subjective judgment, preferably by the experts of the respective disciplines. For ExternE a survey of experts has been done in an informal manner. Also all the relevant information that could be found in the literature was used.

4.4 Results for Energy Production

Once the cost per kg has been determined, multiplication by the emitted quantities of the pollutants yields the cost per activity, for instance per kWh of electricity produced by a power plant. A complete accounting of the damage costs should involve a LCA, i.e., a complete inventory of emissions over the entire chain of processes involved in the activity. The total damage cost per kWh of electricity should include impacts upstream and downstream from the power plant, such as air pollution from the ships, trucks, or trains that transport the fuel to the power plant. That has been done for the fuel chain results of ExternE. A few results for France are shown in Figure 4.6.

For the fossil fuel chains, the lion's share of the external costs comes from air pollutants emitted by the power plant, the main impact categories being global warming and public health. Air pollutants from upstream and downstream activities make a relatively small contribution (roughly 10% of the GHGs). Apart from CO₂, the damage cost is mostly due to health impacts, especially mortality.

The emission of toxic metals is highly uncertain and variable from one source of fuel to another; the numbers shown are not necessarily typical. However, they are in any case so small that their contribution to the total damage cost is negligible.

Recently updated and more complete results for power production in Europe have been calculated during the ExternE-Pol phase of the ExternE series (Rabl et al. 2004); they are shown in Figure 4.7 and Figure 4.8. The numbers are not completely comparable with those in Figure 4.6 because the methodology has been evolving. In particular, the numbers in the following are based on detailed LCA inventories provided by the ecoinvent database. Also the methodology for nuclear damage costs in Figure 4.7 is different from the one in Figure 4.6.

Results obtained for new and current technologies on the basis of ecoinvent are discussed in the remainder of this section. Figure 4.7 presents the results for current and advanced electricity systems, with the external costs per kWh in part a) and the contributions of the individual pollutants in part b). Likewise Figure 4.8 summarizes the results for the different heating systems.

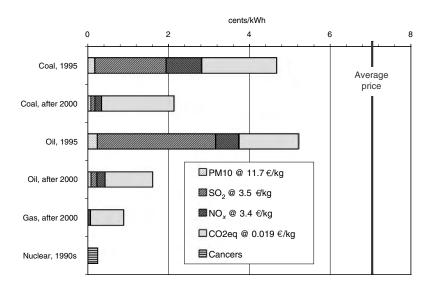


FIGURE 4.6 Typical damage costs for electric power plants in France, for plants operating during the mid-1990s and for plants that respect the new EU regulations issued in 2000. Costs for nuclear are upper bound (0% "effective discount rate"); 90% of cancers from nuclear occur only after the first 100 yr. Retail price of electricity is about 7 cents/kWh; production cost of base load electricity about 3 cents/kWh.

Current fossil systems exhibit the highest external costs, in the range of 1.6–5.8 c€/kWh (Figure 4.7). Introduction of advanced technology (CC and PFBC) substantially reduces the external costs of fossil systems, but they still remain in the range 1–2 c€/kWh. This also applies to cogeneration, for which gas technology generates external costs one-third lower than diesel technology. Greenhouse gas (GHG) contribution to total costs prevails over other species for advanced fossil technologies, making about 80% of total external costs for CC and PFBC. Current averages of coal and oil plants show still high contributions from SO₂, depending on the extension of installation of scrubbers in UCTE.

Nuclear external costs are below 0.19 c€/kWh of which 70% is radioactivity dependent. However, with discounting this contribution would strongly decrease, because most of the calculated damages from radiation are either related to very long-term emissions (e.g., radon from uranium mill tailings) or to very long-lived isotopes giving very small doses. On the other hand, the present estimation of external costs from ionizing radiation is based on a preliminary calculation using the disability-adjusted life years (DALY) concept, a rough attribution of cost/DALY, and an incomplete but important subset of isotope releases from the ecoinvent database. It is recommended to rework the estimation of damage factors from radioactive emissions in future projects of the ExternE series. The nuclear power plant itself contributes 5% or less to external costs from the nuclear chain.

Wind onshore with nearly 0.09 c€/kWh performs slightly better than wind offshore with 0.12 c€/kWh. Monocrystalline silicon photovoltaic (PV) panels of European fabrication, installed in Southern Europe cause nearly 0.28 c€/kWh, which would mean 0.41 c€/kWh for the average yield of 800 kWh/kW_{peak} yr in Central Europe. Assuming improvements in manufacturing technology of crystalline silicon, improved cell efficiency, and an expanded PV market, 0.21 c€/kWh has been estimated for future (2010) systems. External costs associated with imported panels may differ due to different manufacturing technology and electricity supply. Due to the relatively high material intensity of PV and wind, the contribution from heavy metals is about 15% and nearly 25%, respectively. Hydropower exhibits the lowest external costs of all systems, below 0.05 c€/kWh, but costs may increase on sites where higher direct emission of GHG from the surface of reservoir occur.

In general, gas boilers have lower external costs than boilers burning light oil: approximately $0.6 \text{ c} \in /\text{kWh}_{\text{th}}$ vs. $0.94 \text{ c} \in /\text{kWh}_{\text{th}}$ (Figure 4.8). The upstream chain of gas and light oil contributes

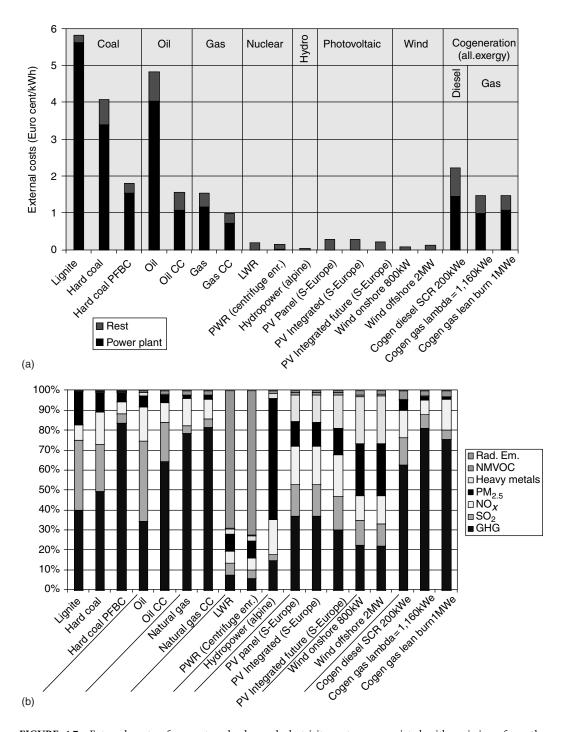


FIGURE 4.7 External costs of current and advanced electricity systems, associated with emissions from the operation of power plant and with the rest of energy chain. (a) The costs in €cent/kWh. (b) The contribution of the individual pollutants.

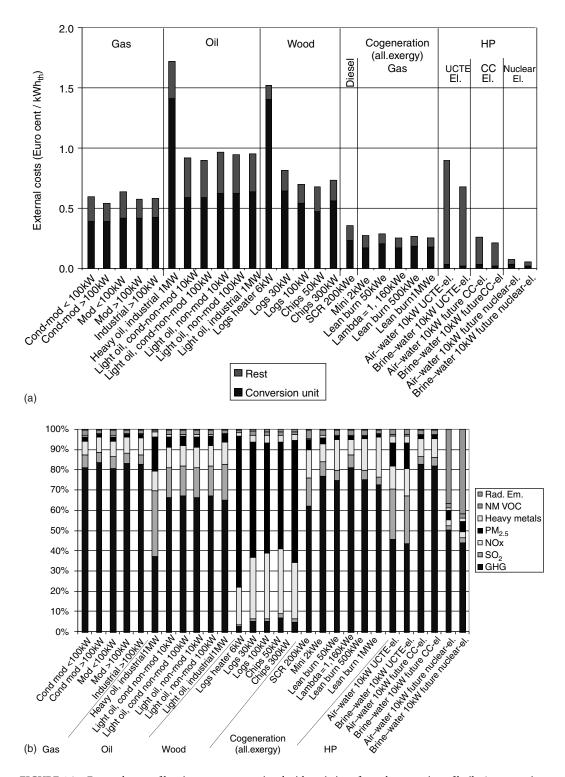


FIGURE 4.8 External costs of heating systems, associated with emissions from the operation of boiler/cogeneration unit and with the rest of energy chain. (a) The costs in €cent/kWh. (b) The contribution of the individual pollutants.

roughly one-third to the total external costs. GHG contribute two-thirds of the total external costs for oil, over 80% for gas boilers. Burning heavy oil gives the highest damages with over $1.7 \text{ c} \text{-/kWh}_{th}$, where SO_2 makes about 33% and GHG 38% of the damages. A range of about 0.7– $0.8 \text{ c} \text{-/kWh}_{th}$ has been calculated for wood boilers, where the upstream chain contributes 20%–30% to total damages. Particles and nitrogen oxides emissions contribute most, i.e., nearly 60% and about 30%, respectively, to total damages. The modern fireplace gives more than $1.5 \text{ c} \text{-/kWh}_{th}$, mostly due to the high particle release. GHG contribute 7% or less to total external costs for modern wood systems, because the CO_2 from wood combustion is compensated by tree sequestration.

Cogeneration plants perform well when allocation is based on exergy: $0.36 \text{ c} \text{-/kWh}_{th}$ for diesel and an average of $0.27 \text{ c} \text{-/kWh}_{th}$ calculated for the gas units. The magnitude of external costs of heat pumps (HP) is controlled basically by two factors: the seasonal performance factor (SPF) and the energy supply source. For current systems and average UCTE electricity mix the external costs are nearly $0.7 \text{ c} \text{-/kWh}_{th}$ and $0.9 \text{ c} \text{-/kWh}_{th}$ for the air–water HP and brine–water HP, respectively. Due to the fact that about 26% of the UCTE electricity mix is from coal systems, damages from SO_2 contribute nearly one quarter to the total external costs. For future HP technologies and electricity delivered by gas CC or nuclear, these costs go down to $0.26 \text{ c} \text{-/kWh}_{th}$ and $0.21 \text{ c} \text{-/kWh}_{th}$, or nearly $0.08 \text{ c} \text{-/kWh}_{th}$ and $0.06 \text{ c} \text{-/kWh}_{th}$, respectively, for the two heat pump systems and the two electricity supply cases (Figure 4.8).

4.5 Comparison Landfill ↔ Incineration

Because waste can be considered a source of renewable energy, the paper also examines the role of energy recovery from landfill and incineration, the main approaches to waste treatment. The results reported are based on Rabl, Spadaro and McGavran (1998), Rabl and Spadaro (2002), and Zoughaib and Rabl (2004).

4.5.1 Assumptions

Because the comparison 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; for the purpose of illustration, the emissions due to possible differences in transport distance by showing a hypothetical distance of 100 km are included. 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 assumptions of the analysis are summarized in Table 4.3.

The principal emissions from landfill are CH₄ and CO₂. Figure 4.9 shows the total GHG emissions of a municipal solid waste landfill versus time. CH₄ is expressed as equivalent CO₂, using a GWP of 20. 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 time in Figure 4.9 is measured from the date that a compartment is sealed.

TABLE 4.3 Assumptions of the Analysis of Incineration and Landfill of Municipal Waste

Stages taken into account	Transport of waste; 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 (2000), in reality the average emissions are usually lower
Impact pathway analysis	Assumptions and results of ExternE (2004)
Impacts taken into account	Human health; crops; materials and buildings; global warming
Impacts not taken into	Effects of air pollutants on ecosystems; Soil and water pollution due to leachates
account	(probably negligible); Impacts from residues of incineration; Amenity impacts
	(visual intrusion, noise, odor)

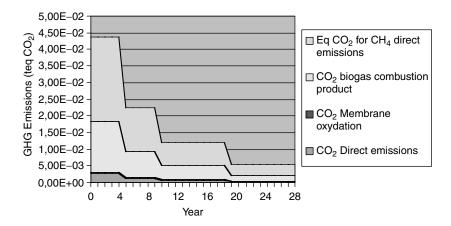


FIGURE 4.9 Total greenhouse gas emissions from a municipal solid waste landfill versus time, t_{CO2eq}/t_{waste} if 70% of the CH₄ is captured. (This figure was prepared from a table with averages over four or more years; in reality, the evolution is smooth rather than in steps).

There may also be emissions to soil and to water. Emissions to soil can occur from slag, leaking liners under the landfill, and the storage site of incinerator fly ash. Emissions to water arise from certain types of flue gas treatment and from the extraction of leachates under a landfill. Emissions to soil are difficult to estimate because they depend on integrity of the liners in the future. If the landfill is operated according to regulations and if there are no 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 their impacts would remain limited to the immediate vicinity of the landfill, with the possible exception of sites with sufficient ground water movement. In the present version impacts of emissions of leachates to soil or water have not been considered.

The external costs and the comparison between landfill and incineration turn out to be extremely sensitive to assumptions about energy recovery. For that reason, a fairly large number of options (indicated in the figures by labels such as "E=c+o") is condered:

For incineration:

- Recovery of heat and electricity, for typical installations in France, according to ADEME (2000) (E=..., H=...)
- Recovery of electricity (E=...)
- Recovery of heat (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 recovery of electricity, a year-round demand is assumed such that all the electricity is used. Likewise, for recovery of heat, a year-round demand is assumed (industrial process heat loads or certain district heating systems with year-round demand, e.g., Paris and Vienna), such that 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 each of these options, several suboptions (indicated by the labels in the captions) are considered:

• 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 example, (E=c+o, H=c+o) designates a system where heat and electricity are produced, each displacing a fuel mixture of coal and oil. 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 external costs are concerned.

4.5.2 Results

A summary of the total damage cost for all the options is shown in Figure 4.10. More detailed results for some of the options can be found in Figure 4.11, showing the contribution of each stage and of the major pollutants (dioxins and toxic metals are shown as "other"). 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-ton truck, is also negligible. The only significant contributions come from direct emissions (of the landfill or incinerator) and energy recovery.

For landfill the cost is dominated by GHG emissions because only about 70% the $\mathrm{CH_4}$ can be captured. Energy recovery from a landfill is not very significant. By contrast, energy recovery is crucial for the damage cost of incineration. Under favorable conditions (all heat produced by incinerator displaces coal and oil) the total external cost can even be negative, i.e., a net benefit. In contrast to most other countries, in France

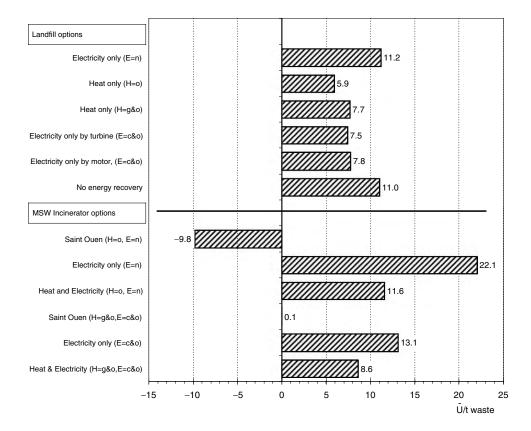


FIGURE 4.10 Results of total damage cost for all options. If electricity displaces nuclear, damage costs are essentially the same as for the case without energy recovery.

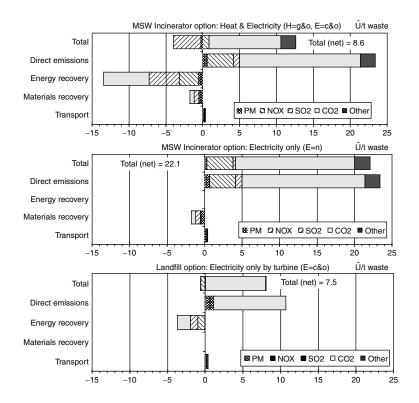


FIGURE 4.11 Some detailed results, by stage and pollutant. "Other" refers to dioxins and toxic metals.

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.

The uncertainties are very large (see Figure 4.5) and they have different effects on different policy choices. Some comparisons, in particular those between landfill and incineration, are especially sensitive to this uncertainty because GHGs play such a large role for landfills.

4.6 Conclusions

The most relevant damages caused by air pollution can be quantified and monetized using the methodology of ExternE. However, damage cost estimates still show large uncertainties, and the reader may wonder whether it is meaningful to use them as basis for decisions. The first reply is that even a threefold uncertainty is better than infinite uncertainty. Second, in many cases the benefits are either so much larger or so much smaller than the costs that the implication for a decision is clear even in the face of uncertainty. Third, if in the other cases the decisions are made without a significant bias in favor of either costs or benefits, some of the resulting decisions will err on the side of costs, others on the side of benefits. Rabl, Spadaro, and van der Zwaan (2005) have examined the consequences of such unbiased errors and found a very reassuring result. The extra social cost incurred because of uncertain damage costs is remarkably small, less than 10%–20% in most cases even if the damage costs are in error by a factor three. But without any knowledge of the damage costs, the extra social cost (compared to the minimal social cost that one would incur with perfect knowledge) could be very large.

A key argument for the necessity of quantifying damage costs has emerged from recent epidemiology. Whereas in the past there was a general belief that it would be sufficient to reduce the concentration of air

pollutants below their no-effect thresholds, epidemiological studies have not been able to find evidence for such no-effect thresholds at the population level, and linear DRFs appear increasingly plausible for low concentrations of most air pollutants. If there are no safe levels of air pollution, then policy makers have no natural criterion for deciding how far to reduce the emission of pollutants. Because the cost per kg of avoided pollutant increases sharply as emissions are reduced, there is a serious risk of spending too much on the fight against air pollution. This could in fact lead to a deterioration of public health if the money thus spent is not available for more cost-effective measures. A comparison of costs and benefits is needed for rational policy making.

Gradually the results of ExternE are diffusing into the world of decision makers. For example, ExternE is recognized as the reference for comparative risk assessment by agencies such as the International Atomic Energy Agency. In the EU, ExternE is increasingly used as input to environmental decisions, e.g., via cost–benefit analyses that justify tighter regulations for the emission of pollutants from incinerators or from power plants.

Glossary and Nomenclature

As Arsenic

CBA Cost-benefit analysis

CO Carbon monoxide

CO_{2eq} Quantity of a greenhouse gas expressed as equivalent quantity of CO₂, using the GWP of the gas

Cr^{VI} Chromium in oxidation state 6

CRF Concentration–response function

CV Contingent valuation

Discount rate Rate that allows comparison of monetary values incurred at different times, defined such that an amount P_n in year n has the same utility as an amount $P_0 = P_n(1+r)^{-n}$ in year zero

DRF Dose-response function

EC European Commission

EPA Environmental Protection Agency of USA

External costs Costs that arise when the social or economic activities 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

O₃ Ozone

PAH Polycyclic aromatic hydrocarbons

Pb Lead

 PM_d Particulate matter with aerodynamic diameter smaller than $d \mu m$

S Sulfur

 S_{CR} Slope of concentration–response function [cases/(person yr $\mu g/m^3$)]

UCTE Union for the Coordination of Transmission of Electricity

UWM Uniform world model (for simplified estimation of damage costs)

v_{dep} Deposition velocity [m/s]

VOC Volatile organic compounds

VOLY Value of a life year

VSL Value of statistical life

WTP Willingness to pay

YOLL Years of life lost (reduction of life expectancy)

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