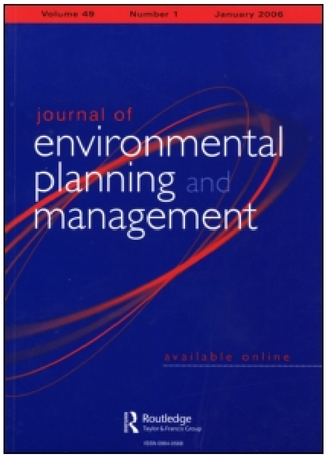

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Ari Rabl [a]; Mike Holland [b]
[a] Ecole des Mines de Paris, Paris, France
[b] EMRC, Reading, UK

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Environmental Assessment Framework for Policy Applications: Life Cycle Assessment, External Costs and Multi-criteria Analysis

ARI RABL* & MIKE HOLLAND**
*Ecole des Mines de Paris, Paris, France
**EMRC, Reading, UK

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ABSTRACT *The paper presents a framework for the analysis of external costs of environmental burdens, namely an impact pathway analysis, often coupled with the inventory stage of life cycle assessment (LCA). The ground rule is: quantify as much as possible in terms of burdens (pollutant emissions, etc.), impacts, and their monetary equivalent, then use multi-criteria analysis (MCA) for any remaining impacts that are considered to be too uncertain or defy quantification through to monetization. Although MCA could be used directly on estimates of burdens or impacts, monetary valuation provides a mechanism for consistent weighting of impacts categories based on assessment of public preference. Further advantages of extending LCA through detailed impact assessment combined with monetary valuation are that it greatly simplifies MCA by combining a large number of different environmental impact categories, thereby avoiding an unmanageably large number of criteria, and also facilitates cost benefit analysis (CBA). The risks are noted of inappropriate use of the tools or interpretation/use of the results, and recommendations are made for improved practice. These points are illustrated with examples. The key messages are: (1) that policies should be targeted correctly to give a clear signal which source of a burden should be reduced by how much; (2) that analysts should take into account the needs of policy makers and the link between the analysis and possible policy applications; and (3) that current LCA practice gives limited guidance in both areas, largely through a lack of consideration of the relative and absolute importance of different types of impact. However, this is precisely the strength of external costs analysis, particularly when used with MCA.*

Introduction

To optimize the allocation of resources one needs reliable information about the costs and benefits of proposed actions. For the sake of equity and justice one also has to take into account their external costs, i.e. costs imposed on others who do not participate in a given market transaction, in particular the damage costs of pollution

Correspondence Address: Ari Rabl, Ecole des Mines de Paris, 60 boul. St.-Michel, F-75272 Paris, France.
Email: ari.rabl@gmail.com

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and other environmental burdens (the term 'burden' is more general than pollution, including impacts such as noise, land use and visual intrusion). The quantification of damage costs is needed for many important applications, often via its use in cost-benefit analysis (CBA), for example:

- formulating environmental regulations (e.g. determining the optimal limit for the emission of a pollutant);
- determining the socially optimal level of a pollution tax[1] or of the quantity of tradable permits;
- identifying technologies with the lowest social cost (e.g. choosing between coal, natural gas and nuclear for the production of electricity);
- evaluating the benefits of improving the pollution abatement of an existing installation such as a waste incinerator;
- socially optimal dispatching of power plants, for example through purchase requirements for renewable energy technologies;
- 'Green accounting', i.e. complementing the traditional GNP accounts with corrections for environmental damage.

The optimal pollution level is the one that minimizes the sum of damage cost and abatement cost.

In recent years the subject of external costs has attracted a great deal of attention in the EU with the acknowledgment by the European Commission (EC) that the internalization of external costs is a key tool for the attainment of sustainable development. 'Getting the price right' has become a catchphrase to express this idea. A large number of recent and current research projects of the EC concern this subject.

However, there is much confusion about the relation between external costs and their use for environmental policy. Therefore, the purpose of the present paper is to present the general framework for the analysis of external costs[2] of environmental burdens, to show how results may be used and to point out the risks of inappropriate interpretation or use of the results. These points are illustrated with examples.

To begin, the paper notes confusion in the use of the term 'external cost' because there are at least two possible definitions:

- costs imposed on non-participants, that are *not taken into account* by the participants in a transaction;
- costs imposed on non-participants, that are *not paid* by the participants in a transaction.

According to the first definition a damage cost is internalized if the polluter reduces the emissions to the socially optimal level, for example, as a result of a regulation that imposes an emission limit. In addition, the second definition requires that the polluter compensates the victims for any damage, either directly or indirectly (for example, by paying a pollution tax). In either case, the resulting level of emissions is equal to the social optimum. But the corresponding damage cost is external only according to the second, not the first definition. Many economists accept only the first definition, and often the term 'relevant externality' is used to designate the portion (if any) of the damage cost that is greater than the social optimum. However, in practice 'relevant externalities' are extremely difficult to evaluate because they require a CBA to determine the social optimum; that is doubly uncertain because the uncertainties of the damage cost are compounded by those of the abatement cost. The authors have not found any study that has tried to determine 'relevant externalities' for pollution. Furthermore, all the damage costs calculated by the leading externality programme (ExternE, 1998, 2000, 2004) and related projects in the EU have been presented simply as 'external costs', even though they are the total damage cost, rather than the portion beyond the optimum. To avoid confusion the use of the term 'damage cost' is preferred here.

The calculation of the damage cost of a pollutant requires an impact pathway analysis (IPA), i.e. an evaluation of the chain emission → exposure → impact → cost. It involves tracing the passage of the pollutant from where it is emitted to the affected receptors (population, crops, forests, buildings, etc.), quantifying the impacts and evaluating their costs. A very brief description of IPA is given in the next section (for a detailed description of the methodology, see the publications of the ExternE project series at www.externe.info, in particular the report *Methodology 2005 Update* of the MAXIMA project (ExternE, 2005)). The result of an IPA is the damage cost per burden, e.g. the cost per kg of PM_{10} for a specific site and stack height.

For many environmental decisions several stages of a process or product need to be taken into account. In such cases a life cycle assessment (LCA) of these stages is needed to establish an inventory of burdens, followed by an IPA of each of these burdens. Whether an IPA of a single stage or an LCA of several stages is required, depends on the policy decision in question. For example, for regulations concerning emissions from coal-fired power plants only an IPA is needed; an LCA of the entire fuel chain would be an unnecessary complication. By contrast, the choice between coal and nuclear involves a full cradle-to-grave LCA.[3]

Monetary valuation of the impacts offers two crucial advantages: it reduces the otherwise incommensurate categories to a common measure, which is consistent with the market. One objection, that separate categories have the advantage of preserving the different nature of the impacts and avoiding the illusion of substitutability, can be answered easily by recommending that a breakdown of the damage costs by categories be shown in addition to the total. Another frequent objection to monetary valuation is based on the view that monetary values should not be assigned to goods such as a beautiful landscape, the existence of a rare animal or human life. However, this is a misunderstanding of the problem: the monetary valuation is based on the willingness-to-pay (WTP) to avoid losing the item in question; it has nothing to do with the intrinsic value. For example, our WTP (including ability to pay) to avoid the risk of an anonymous premature death is finite, even if we feel that the value of life is unlimited. If the WTP for a non-market good has been determined correctly, it is like a price, consistent with the price of market goods. If stakeholders or decision makers disagree about the valuations applied in any analysis they are of course free to substitute their own (provided they can demonstrate it to be justified) to see if it would make any difference to the conclusions reached, provided that the original analysis has been reported with sufficient transparency.

Apart from the acknowledgment that external costs (based on WTP) should be internalized, there are no more ethics in the analytical framework of this paper than in any ordinary market transaction. But of course, costs are not the only criterion for decisions; for example, the protection of vulnerable groups and distributional issues (who gains, who loses) are important. Such issues can be treated, explicitly or implicitly, by multi-criteria analysis (MCA).

Framework for the Analysis

Impact Pathway Analysis (IPA)

The principal steps of an IPA applied to the emission of a pollutant to land, water or air can be grouped as follows:

- Specification of the site, height and stack conditions of the source of the pollutant (e.g. kg of NO_x per GWh emitted by the stack of a power plant);
- Calculation of the change in 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);
- Exposure of the sensitive receptors such as people, buildings or ecosystems due to the increased concentration;
- Calculation of the resulting impacts (damage in physical units), using an exposure-response function[4] (e.g. cases of asthma due to this increase in ozone);
- 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 some burdens, e.g. visual intrusion, the passage from burden to cost is more direct, without intermediate steps. The result of an IPA is the damage cost per burden. If the results are for a specific source of the burden, that should be indicated. IPA is the logically correct approach, but the details of the implementation differ between different studies.

For many policy applications representative values are more appropriate than site specific ones. The most exact way of obtaining representative values is to average over a sufficient number of site-specific ones, with suitable weighting factors. In practice a very simple approximation can be obtained by means of the 'uniform world model' (UWM) of Spadaro and Rabl in conjunction with correction factors for the respective source conditions such as stack height (see Section 11.2 of ExternE (2005)). The UWM has been validated by comparing it with well over 100 detailed site-specific EcoSense[5] calculations for sites on four continents: it has been found to reproduce the results of the detailed calculations within a factor of two[6] for most power plants (Spadaro, 1999; Spadaro & Rabl, 2002; additional calculations by Spadaro) (Ecosense, the software of ExternE, was developed by Krewitt *et al.*, 1995). A third method is the multi-source EcosSense software of ExternE. Simple typical results can be found with the EcoSenseLE (EcoSense lookup edition) software at www.externe.info.

It would be highly desirable to establish a catalogue of burdens and typical costs per burden. The beginnings of such a catalogue are available in the tables of damage cost per kg of pollutant published by ExternE (2005) and in the BeTa database (Holland *et al.*, 2006).

LCA and Cost per Product or Process

LCA is a widely used tool for the analysis of environmental problems (Barnthouse *et al.*, 1998). In recognition of that, ISO has developed the 14000 family of standards which is primarily concerned with environmental management (http://www.iso.org/iso/en/prods-services/otherpubs/iso14000/index.html). The assessment of impacts in LCA is evolving and there are different variations (Jolliet *et al.*, 2005), but most practitioners have refused monetary valuation. Instead they report the impacts in terms of approximately 10 different impact categories, with no attempt to provide a synthesized view on which option of those considered performs best overall. That is awkward for decision makers because the categories are incommensurate and thus necessitate a multi-criteria analysis (MCA) that may be performed formally, or simply left to the judgment of the decision makers. This latter situation presumes that the decision makers are in a better position to evaluate the relative merits of each category than the environmental/technical experts performing the analysis; being implicit to the decision made, the rationale for preferring one option over another also lacks transparency. As example of trying to make LCA more policy-relevant, Hertwich & Hammitt (2000) are cited.

In principle the damages and costs for each pollution source in the life cycle should be evaluated by a site-specific IPA. But in practice most LCA studies have taken the shortcut of first summing the emissions over all stages and then multiplying the result by site-independent impact indices. This is illustrated in Figure 1 for the example of electricity production. However, the subject of life cycle impact assessment (LCIA) is undergoing very active research and development, in an attempt to obtain more realistic results that recognize the importance of site (Jolliet *et al.*, 2005]. There are several different methods for LCIA, for example Eco-indicator 99 (http://www.pre.nl/eco-indicator99/default.htm) and Hofstetter (1998). For an interesting comparison of three LCIA methods, see Dreyer *et al.* (2003).

Given the current state of LCIA, it is strongly recommended using only the inventory phase of LCA and calculating the impacts and cost per burden according to ExternE. Combining the inventory with the cost per burden, it is possible to readily find the cost per product or process.

LCA Boundaries and Policy Applications

The appropriate boundary of the LCA stages depends on the policy choices under consideration. For example, for decisions concerning the provision of natural gas the boundary of stages should exclude the utilization of the gas, to avoid unnecessary complications (because it is the same for all options), but it should include the impacts from exploration, extraction, purification and transport. There are many different policy instruments that can be used for internalizing external costs, and the information needed for setting their optimal level depends on the instrument. Regulations for specific pollutants require the damage cost per kg of each pollutant. On the other hand, more aggregated information is needed for more complex

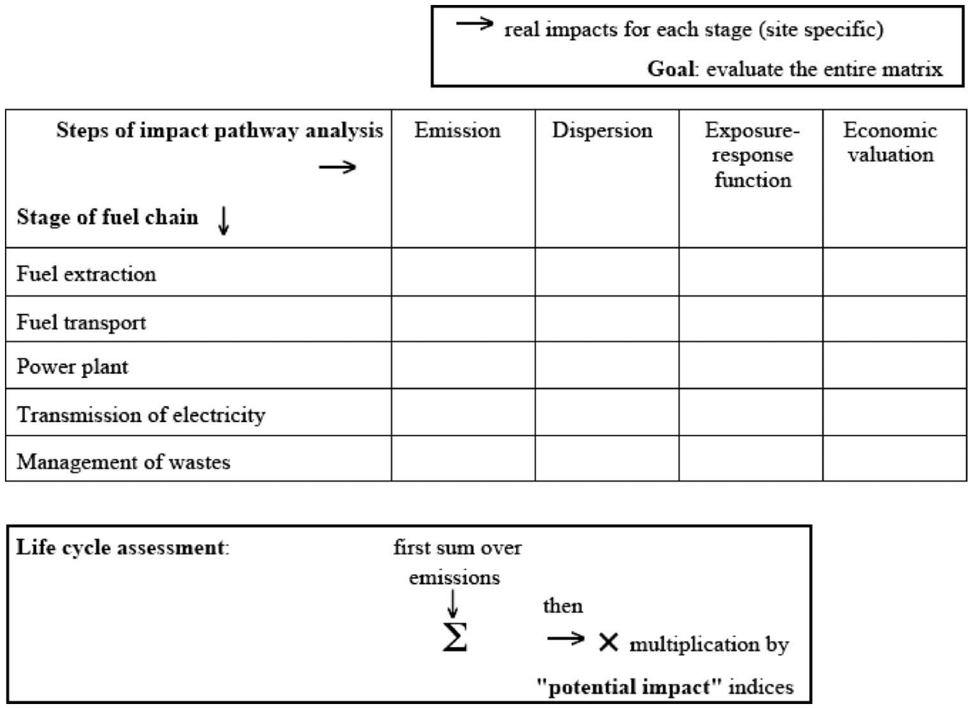

Figure 1. Relation between impact pathway analysis and current practice of most LCA, illustrated for the example of electricity production. *Source*: From Spadaro & Rabl (1999). In recent years the impact assessment of LCA has evolved, but most LCA studies still do not use realistic exposure-response functions and monetary valuation.

instruments, for example, a regulation that obliges electricity companies to have in their portfolio a minimum market share for renewable energy.

Since it is not possible to foresee all probable uses of the results of an externality study, care should be taken to provide sufficient detail for all the applications of probable interest. Therefore, all damage costs associated with a product should not be blindly added and only the total reported as external costs. Otherwise it would be possible to get the perverse result of a pollution tax for electricity producers that includes impacts of coal mines, without any information about the different contributions to the tax. The principle is 'polluter pays', not 'polluter pays for someone else's damage'. Regulations are needed that target each polluter and each pollutant correctly, for example, regulations for the pollution from coal mines. Of course, regulations could be tried that encourage power plants to buy coal from a better-run mine, but that would be far less effective than regulations that target each mine operator directly.

Whatever the policy decision, the basic ingredient is always the damage cost per burden for all the burdens affected by the decision; only the aggregation over burdens and/or LCA stages will differ. In this context it is of interest to mention that the current phase of ExternE, the NEEDS project (2004–08), will link external cost analysis to LCA and MCA.

An Example

At the start of the analysis it is necessary to define the scope by listing the burdens and impacts that might be significant. Usually many of the minor impacts can be omitted after brief reflection or back-of-the envelope calculations because they cannot be significant. Then the detailed analysis will focus on the priority impacts, i.e. the items that seem most important.

The identification of burdens and impacts that are likely to be significant is illustrated in Table 1 with the example of waste incineration. The damage costs are reported in units of €/unit burden, e.g. €/kg of pollutant. For some pollutants the variation with stack height can be neglected because of global dispersion, ingestion pathways, or creation of secondary pollutants. The stages Transport of Waste and Materials and Energy Recovery are relevant for a comparison with other waste

Table 1. Impacts of waste incineration

Burden	€/kg of pollutant	Transport of waste	Construction of incinerator	Incineration	Materials & energy recovery
SO_2	3.5[b]	x		X	X
NO_x	3.4[b]	x		X	X
PM_{10}	12[a]	x		X	X
VOC	1.1[b]	x		x	
CO_2	0.02[c]	x		X	X
CH_4	0.4[c]			x	
N_2O	6[c]			x	
As	80[b]			x	
Cd	39[a]			x	
Cr^{VI}	200[a]			x	
Hg	8000[b]			X	
Ni	3.8[a]			x	
Pb	600[b]			X	
Dioxins	1.85E + 08[b]			x	
Visual intrusion				Site-dependent	
Traffic		?	? Site-dependent		
Noise		X Site-dependent	X Site-dependent	Site-dependent	
Odour				Site-dependent	
Bottom ash				?	
Fly ash				?	
Waste water				?	

Notes: X = priority impacts; x = not very large for new waste incinerators;
? = no current information, should be looked at, perhaps significant;
blank = not significant. The numbers show typical ExternE damage costs in €/kg of pollutant emitted in France, also usable for much of central Europe.
[a]Strong variation with site or stack height (numbers are for urban sites with stack height around 50 m).
[b]Negligible variation with stack height, slow variation with site.
[c]No variation with site or stack height.

treatment technologies (transport is relevant only if the systems being compared involve different transport distances). Almost no significant impacts are shown from the construction of an incinerator or landfill, referring 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 conclusion holds for other combustion equipment that is used more or less full time, in particular for power plants and waste incinerators (but not for passenger cars). Combining the €/kg of pollutant with an emissions inventory one readily finds the cost per kg of waste.

The potential for damage caused by fly ash and bottom ash is very strongly dependent on the legislation in force in the location where the ash is generated. In situations where these wastes are treated in a way that prevents exposure to their harmful substances, there is no externality, assuming that disposal routes for long-lived toxins are secure over time.

Of course, the existence of legislation does not automatically mean that it will be complied with; it must be policed closely enough to prevent non-compliance.

With regard to the choice of the functional unit, for processes that provide multiple services the question arises which output to choose. For waste incineration with energy and materials recovery a tonne of waste is an appropriate choice because waste disposal is the primary purpose. However, in the agricultural sector there is often no clearly dominant service and the most appropriate unit is the monetary value of the total output of the activity under consideration; in other words, the damage cost is to be stated as percentage of the value of the output.

Policy Instruments and Information Requirements

Types of Policy Instruments

Internalization of external costs requires government intervention. Table 2 lists the principal types of regulations that the government can use for this purpose. Some of these policy instruments act directly on the emissions, others such as eco-labels and portfolio standards can affect emissions indirectly by reducing the consumption of materials or energy. The last column indicates to what extent the analysis necessitates an LCA.

Who is Responsible, Who should be Targeted?

Whereas the 'polluter pays' principle seems straightforward, in practice the question of who is the polluter and who should be targeted by an internalization instrument can be more complex. For example, in the case of pollutants that a consumer emits into the waste water (e.g. estrogen from hormone replacement therapy, or pollutants from cleaning products), the home owner can hardly do anything about it, but the sewage company and/or the product manufacturer can. If they are forced to do it, they will pass the cost on to the consumer, so all can be automatically and correctly internalized, at least on average.

Similar considerations apply to consumer products such as batteries. Here one of the policy options is to oblige the producer of batteries to use technologies without

Table 2. Policy instruments for reducing pollution

Type	Examples	Tool(s) for analysis
Limits on emission of pollutants for specific technologies	Max. mg SO_2 per m^3 of flue gas; max. g CO per km driven by cars	IPA (+LCA in some cases[a])
National emission ceilings	Limit on tonnes of SO_2, NO_x, VOCs, CO_2, etc. emitted by a country	IPA
Mandatory technologies	Usually by demanding 'Best Available Technology' (BAT), e.g. flue gas desulphurization for coal or oil fired boilers	IPA (+LCA in some cases[a,b])
Major technology choices	Decision in Germany to phase out nuclear power	IPA + LCA
Subsidies for clean technologies	Tax credit for wind and solar in California during 1980s; Feed-in tariffs to force electricity company to buy from renewable sources	IPA + LCA
Eco-labels	'printed on recycled paper'; 'no chlorine used'; 'energy star' label for computers	IPA + LCA
Pollution taxes	€/tonne of a pollutant	IPA
Tradable permits	Government sets cap on number of permits (e.g. tonne of SO_2), polluters can trade these permits	IPA
Portfolio standards	Government sets minimum % for the market share of a clean technology, e.g. 'zero emission' vehicles in California, or 'green kWh' from solar energy, and industry adjusts the prices to achieve these goals.	IPA + LCA

Notes: [a]In case of impacts upstream or downstream, e.g. lime production for flue gas desulphurization.
[b]Depending on specifics of each case.

toxic components (Cd, Ni and Hg) even though that increases the cost; of course, the cost will be passed on to the consumer. If this option is more cost-effective than alternatives such as recycling, it is to be preferred.[7] The key consideration in such cases is 'who can repair or eliminate the problem for the lowest cost?' Low administrative costs and ease of verification must also be taken into account in choosing the most appropriate regulations. For example, unlike large power plants, monitoring of lawn mower emissions is not practical, but emission standards for new lawn mowers are.

Note that in any case it is crucial to aim the policy correctly at the right target, namely the specific pollutant that causes the problem. For example, an aggregated tax per kWh does not tell a power plant which pollutant contributes how much of the tax, so the power plant does not know how to allocate its abatement expenditures in an optimal manner. By contrast, a tax per kg of each pollutant does give a clear and correct signal.

Spatial and Temporal Boundaries of the Analysis

Spatial and Temporal Boundaries

In principle all impacts of a burden should be included in the analysis. This implies that the region of concern should be the entire globe (e.g. for mercury and greenhouse gases) and the time horizon should extend from now to infinity (e.g. for toxic metals). For most air pollutants both local and regional impacts are important and should be accounted for by means of appropriate dispersion models. That is not a serious problem for the classical air pollutants because essentially their entire impact is significant only in the short term and in a region on the order at most a few thousand km from the source. The relation between range of analysis and resulting damage cost is illustrated in Figure 2. For example, the ExternE project series includes impacts on the entire European continent in its assessment of air pollution damage costs due to sources in Western and Central Europe. Since the analysis is funded by the European Commission which needs to take a pan-European perspective, and one that looks ahead to 2020 and beyond, it uses the same monetary values for the entire continent. The time dimension is important here, as investment made by the European Union in new member states should advance their economies significantly in the coming years. To apply valuation based on current conditions would imply either that these investments will not bring any advance to economies, or that policy can be made instantly and have an instantaneous effect on people, neither of which is true.

However, for pollutants with global range or very long persistence in the environment such requirements pose several dilemmas. Evaluating the damage cost of global warming is especially problematic, because much, if not most, of the damage will be imposed on developing countries in future decades and centuries. On top of the uncertainties of the physical impacts there are controversial ethical issues related to the valuation of mortality in developing countries (where most of the impacts are likely to occur) and the choice of the discount rate for intergenerational costs.

Impacts beyond the present generation pose a problem, not only in the choice of the appropriate discount rate (e.g. Rabl, 1996). Most of the discussions about intergenerational discounting have been unrealistically one-sided, focusing only on the choice of an intergenerational discount rate, while neglecting the evolution of the costs. The latter is just as important because before discounting a future cost it must be predicted what that cost will be. For example, many carcinogens (e.g. dioxins, PCBs, As and long lived radionuclides) persist in the environment for a long time (some even more than a thousand years) and, furthermore, cancers develop slowly.

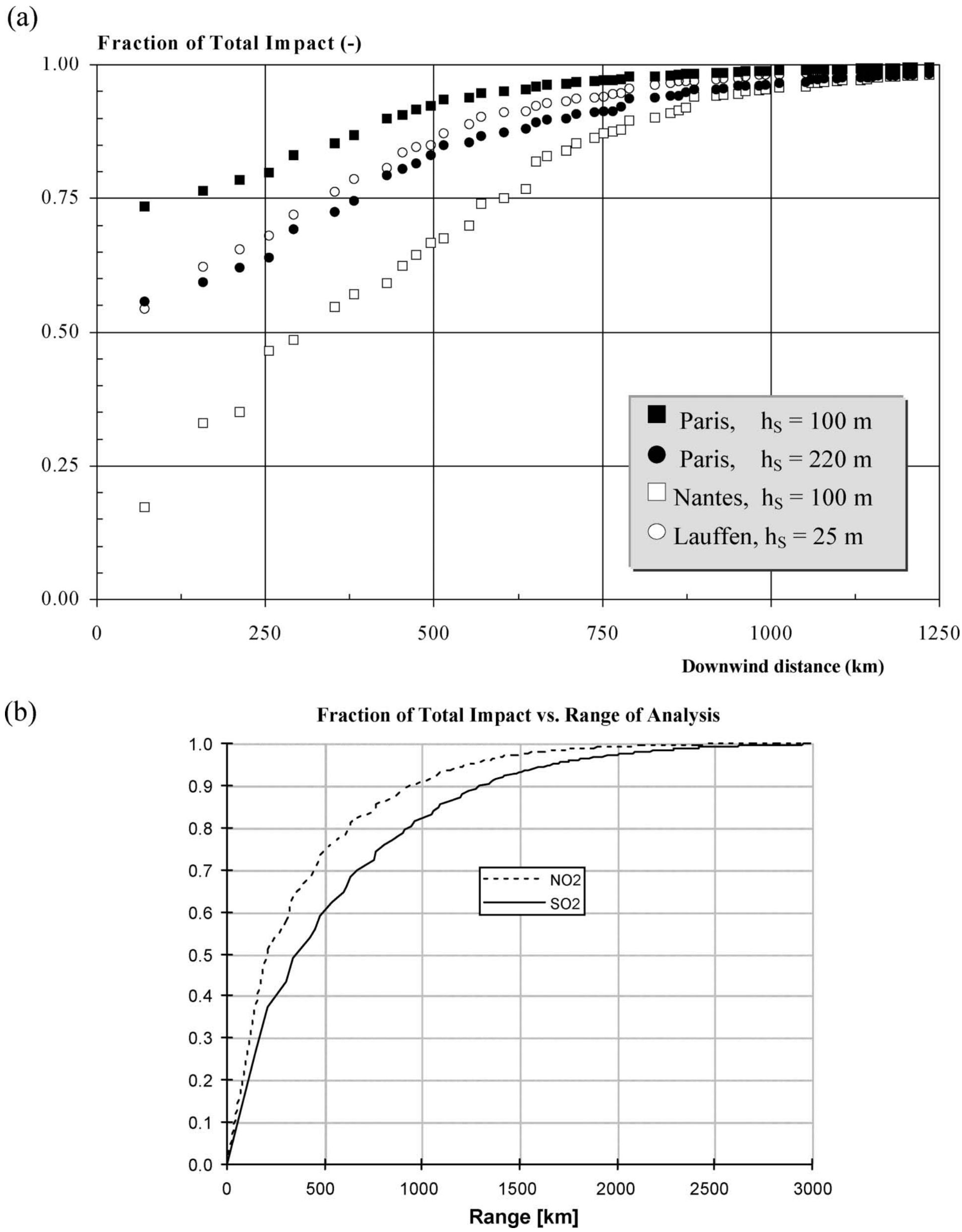

Figure 2. Fraction of total damage as function of radius of region over which damages are included. The exposure-response function (ERF) is assumed to be linear without threshold.
(a) For SO_2 as a primary pollutant, emitted near Paris, Nantes (North Atlantic coast of France) and Lauffen ($\sim$25 km north of Stuttgart, Germany). Calculated by Spadaro (1999) using ECOSENSE. h_s = stack height.
(b) For secondary pollutants due to emission of NO_2 or SO_2, assuming uniform receptor density and linear ERF without threshold, based on EMEP data. Wiggles are due to discretization. *Source*: From Curtiss & Rabl (1996).

For an emission today the impact will not be realized until many years into the future and it is necessary to (1) predict that future cost and (2) discount it. Only an extreme pessimist without any sense of history would evaluate the future impacts of a carcinogen according to the current severity and cost of cancers. Future centuries and millennia will almost certainly enjoy spectacular progress in medicine, and it is likely that cancers will become much easier to treat or prevent. So the future cost will be greatly reduced, quite independent of the discount rate. Thus the assessment of far future impacts involves subjective judgments about the future evolution of mankind and its capabilities.

Furthermore, impacts beyond national boundaries or the near future may be awkward for policy makers. For example, to what extent should policy makers of the EU take into account global warming damages imposed on developing countries a century from now?

Obviously there is no simple solution that would be accepted by all critics. Instead, it is recommended to present a breakdown of impacts and costs according to separate spatial and temporal categories. Table 3, for the damage costs of the nuclear fuel chain, provides an illustration. Depending on the intended user, the breakdown may be provided by country or region, and by near future (present generation, say 25 years) and far future.

The problems, be they analytical or political, should not be taken as excuse for not attempting an estimation of impacts. Any scientific estimation (together with an assessment of uncertainties) is better than either the neglect of a potentially important impact or an excessive preoccupation with an impact that would turn out insignificant. Even if the uncertainties are large, it provides at least a structured approach for thinking about possible environmental policies.

Table 3. Breakdown by time and space of damage costs of nuclear fuel chain (at 0% discount rate) for technologies of 1990 in France

	mEuro/kWh
Short term (< 1yr)	
Local	0.068
Regional	0
Global	0
Medium term (1 – 100 yr)	
Local	0.084
Regional	0.06
Global	0.19
Long term (100 – 100 000 yr)	
Local	0.026
Regional	0.002
Global	2.1
Total	2.52

Source: Rabl & Dreicer (2002).

Local Impacts

Burdens such as noise, odour and visual intrusion are limited to the local zone. The policy instruments for their internalization tend to be different from those for regional or global air pollutants. In particular, negotiation with local stakeholders is appropriate. Monetary values are not needed here, if the negotiations do not involve financial compensation. In particular in the case of health impacts the affected population often rejects direct financial compensation as unacceptable because they consider their health non-negotiable. However, that does not undermine the usefulness of economic valuation in situations where risks are to the general population rather than a specific group within it.

For air pollutants the local impacts tend to make only a small or negligible contribution to the total damage costs, except for ground level emissions of primary pollutants in large cities, where a significant fraction of the total can be imposed within the first kilometers around the source. Nevertheless, it should be emphasized that local impacts should be evaluated and presented even if they are small compared to total damage costs, for the sake of protecting the rights of the local population.

A comment is added about the relation between an IPA and an environmental impact assessment (EIA) that is required before a proposed installation (factory, power plant, incinerator, etc.) can be approved. The purpose of an EIA is to ensure that nobody is exposed to an 'unacceptable' risk or burden (usually by showing that environmental quality standards or no-effect thresholds are not exceeded or that the lifetime risk of cancers due to the installation is less than 10^{-6} per person). Since the highest exposures are imposed in the local zone, it is sufficient for an EIA to focus on a local analysis, up to perhaps 10 km depending on the case. Thus an EIA provides the possibility of a veto if a proposed installation is considered unacceptable or would impose excessive risks on vulnerable groups. By contrast the calculation of total damage costs requires an IPA where the damages are summed over all affected receptors. Damage costs are needed primarily by decision makers at the national or international level, or generally by anyone concerned with total impacts.

Difficulties for the Analysis

Lack of Data

Lack of exposure-response functions. A damage can be quantified only if the corresponding exposure-response function (ERF) is known. For air pollutants, such functions are available for impacts on human health, building materials and crops, caused by a range of pollutants such as primary and secondary (i.e. nitrates, sulphates) particles, O_3, CO, SO_2, NO_x, 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). But even IRIS does not provide sufficient information, especially since for most impacts it shows only threshold information, rather than a usable ERF (see below). For many impacts the epidemiological literature has to be consulted directly.

For application in an IPA the available information often has to be expressed in a somewhat different form, accounting for additional factors such as the reference incidence rate (ExternE, 1998; Spadaro & Rabl, 2004). There are also cases where the available information has to be supplemented by expert judgements that introduce a subjective element. An especially important example is the question of the health effects of individual components of the PM mixtures in ambient air, a question about which very little is known. For example, how toxic are nitrate particles compared to sulphates and to primary combustion particles? Most of the current health impact assessments (e.g. WHO, 2003; Abt Associates, 2004) assume that all components are equally harmful. But if nitrates are harmless (a possibility that cannot be excluded on the basis of current knowledge), the damage cost of NO_x emissions would be very much smaller than current estimates and that of other PM components would be higher. This problem has been discussed in detail in the reports on the methodologies used by ExternE (2005) and the European Commission's Clean Air For Europe Programme (CAFE, see Hurley *et al.*, 2005). The current version of ExternE distinguishes different components by assuming that (here PM_{10} and $PM_{2.5}$ designate the average composition of ambient air):

- the toxicity of nitrates is 0.5 times that of PM_{10};
- the toxicity of sulphates is equal to that of PM_{10} (or 0.6 times $PM_{2.5}$);
- the toxicity of primary particles from power stations is equal to that of PM_{10};
- the toxicity of primary particles from vehicles is equal to 1.5 times the toxicity of $PM_{2.5}$.

Effects of O_3 are considered independent of PM and added, whereas direct effects of CO, SO_2 or NO_x are not taken into account.

Unfortunately, for many pollutants and many impacts the ERFs 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 ERFs are known (often on the assumption of linearity without threshold). Pennington *et al.* (2002) have proposed a promising method of using LOAEL or NOAEL data for estimating ERFs, but their results are not yet sufficiently elaborated for implementation. Note that the required ERFs are at the population level, not the level of individuals; the former can be without threshold even if there are thresholds for individuals (because real populations include a wide range of individual sensitivities and background exposures).

A particularly troubling example of substances without known ERF are pesticides. Some studies of the damage cost of pesticides can be found in the literature, but they are simply based on willingness-to-pay (WTP) to avoid pesticides, determined by contingent valuation. Since people do not know the real damage, their WTP expresses a vague fear rather than an informed judgement about the real damage, unknown because of the lack of ERFs.

Sometimes general principles such as sustainable development or the precautionary principle are invoked when damages cannot be quantified. Unfortunately, they provide no guidance for specific actions. Interpreting the precautionary principle as saying that the emission of a substance must be avoided if it can harm human health can lead to consequences that cause far more damage. For example, lowering the limit for the allowable emission of dioxins from waste incinerators will avoid some cancer deaths, but people will have to pay more for waste disposal. Such costs induce effects elsewhere in the economy. For example, in the USA, Keeney (1995) has shown that for each $5 to 10 million of cost imposed by a regulation there will be on average one additional premature death due to this cost. Poverty kills, a finding confirmed by numerous studies. Therefore, when evaluating a decision, it is necessary to consider the consequences of alternatives and of unintended effects.

Lack of data for monetary valuation. There is no problem for impacts on market goods because only market prices are needed. But for several potentially important impacts sufficient information is not available for the monetary valuation. Particularly important examples are biodiversity and, in developing countries, the valuation of premature mortality.

Lack of data for future emissions. LCA data are retrospective but prospective data are needed because the most important applications concern choices for the future—it is not possible to change the past. In reality the emission of most pollutants (except CO_2) by large industries has been reduced by a factor of 3 to 10 during the past 10 years, a trend that is continuing. Yet even the EcoInvent database (Frischknecht *et al*., 2004), probably the most comprehensive and up-to-date, is for technologies and emissions in the year 2000. For a review of life cycle inventory databases in the world, see Curran (2006). Unfortunately, few if any LCAs have tried to take the evolution of emissions into account, and their results can be quite misleading in cases where such evolution is different for different technologies. It is necessary as a minimum for policy purposes not to consider even current burden levels, but those that will exist once current legislation is fully in place, for example, in the years 2020 or 2030. Failure to consider this far ahead could lead to a recommendation that further action is necessary when that action has already been agreed to. As starting point for estimating future emissions, the limit values of the regulations that are expected to be in force can be taken: they are an upper limit if the regulations are enforced (but the difference between limit value and real emissions can vary with pollutant and technology).

Non-linearity of Impacts

Some external costs are strongly non-linear functions of the burden, for example, O_3 damages due to NO_x emissions. Non-linearities are also important for ecosystem impacts, e.g. eutrophication. The correct procedure for treating non-linearities is somewhat technical and complicated; so far it has not been implemented correctly in any externality studies.

The goal is to estimate marginal damage costs because the socially optimal level of pollution control corresponds to the point where the sum of marginal damage cost and marginal abatement cost equals zero. However, if this seemingly simple statement is interpreted carelessly it could lead to absurd policy recommendations for impacts that are a non-linear function of the emission. To illustrate this problem, consider Figure 3 which shows a pollutant whose damage increases with emission at low emission levels but decreases again if the emission is high. Such a situation actually occurs with O_3 impacts as a function of one of the precursor emissions, NO (note that most NO_x is emitted as NO). The case of O_3 damage due to NO is the most extreme (complicated even more by the strong dependence of the curve on the other precursor VOC), but the problem also occurs in milder form with aerosols created by NO_x and SO_2 emissions.

With a careless literal interpretation of 'marginal damage cost' a negative marginal damage (tangent at the current emission level E_1) would be found, implying that the policy response should be to encourage even greater emission of this pollutant. Such a policy response would miss the real optimum at E_{opt}. It would be trapped in a bad local optimum instead of finding the much better global optimum at E_{opt}.

Since the policy goal should be to bring us to the global optimum, CBA should be based on the marginal costs at the optimum. Therefore, the appropriate marginal damage cost is the tangent at the point (E_{opt}, D_{opt}). Finding this point is not easy. Both the damage cost and the abatement cost can vary with emission site, and so does the optimal emission level; furthermore, the abatement costs vary with time and evolution of the technologies. Ideally a policy maker should know the entire cost curves for marginal damage and abatement at each site. In the case of NO_x, SO_2 and VOC the damage costs are complicated site-dependent functions of not only the pollutant under consideration but also the simultaneous emission of several other pollutants with due consideration of all of their respective emission sites. The optimization requires the solution of the coupled optimization equations.

Of course this poses a problem in practice since policy makers want simple numbers rather than complicated functions, to say nothing of the computational difficulties of determining the complete functions. If a single number is required, it should be reasonably close to the value at the optimum. This begs the question since the optimum is not known.

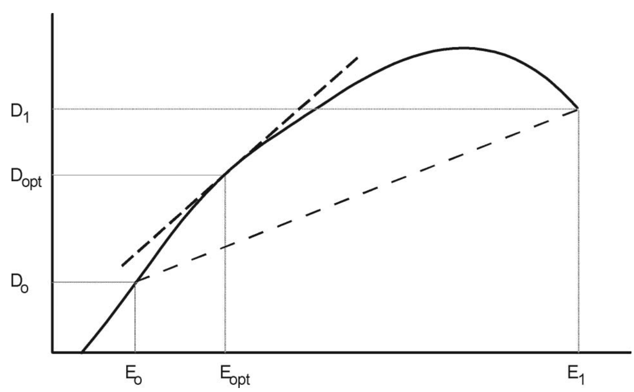

Figure 3. Pollutant whose damage D increases with emission E at low levels but decreases again if the emission is high. Slope of tangent at optimum (E_{opt}, D_{opt}) is the correct marginal damage; the tangent at current emission E_1 is not. Since the optimum is not known, the slope of the chord from pre-industrial (E_0, D_0) to current (E_1, D_1) can be taken as first approximation.

The best that can be done is to proceed iteratively. With an initial guess of the optimal emission levels, a first estimate of the appropriate marginal damage costs can be derived. Comparing them with the abatement costs the estimation of the optimal emission levels can be improved. In view of the uncertainties of the abatement costs (if they are known at all in the required range) the estimates of the optimal emission levels are likely to remain very rough, with the ensuing additional uncertainties of the appropriate marginal damage costs. Fortunately, there seems to be a fair amount of tolerance to errors in the determination of the optimal emissions, as shown by Rabl *et al.* (2005), so even an initial estimation of the optimum may suffice for the purpose of calculating the damage costs for non-linear impacts.

As a starting point for applications in the EU, the estimates could be taken of optimal emission levels by Rabl, Spadaro & van der Zwaan, who find that the emissions of NO_x and of SO_2 should be reduced to levels between 0.2 and 0.8 times (depending on pollutant and country) their level in 1998. But the estimation of optimal levels would have to be extended to VOC and NH_3, with due attention to the coupling between VOC and NO_x through their contributions to O_3 formation.

The optimal NO_x emissions are much more uncertain than those for SO_2, for several reasons. Not only is the damage cost due to nitrate aerosols uncertain because of the lack of information on their toxicity, but the optimum depends also on the damage costs due to O_3, because the optimization for NO_x involves the total marginal damage cost, not the individual cost components due to nitrates or ozone. The O_3 damage due to NO_x depends in turn on the background emissions of VOC. So far the optimal emission levels for VOC have not been estimated, and in any case iterations would be needed because of the coupled nature of the equations. Perhaps the best compromise for NO_x is to set the marginal damage cost equal to the slope of the line from pre-industrial emissions E_o to the current emissions E_1.

Figure 4 illustrates another example of non-linearities; it can arise in ecosystems, for example, with eutrophication of water bodies. Two features are very different from most other impact types, in particular from health impacts of air pollutants:

- there is a threshold zone above which the impacts increase sharply with the burden (e.g. emission of nitrates to water), reaching a plateau at high levels of the burden;
- to bring the system back to the original low impact level, the burden must be reduced to the threshold for decreases which is lower than the threshold for increases, a phenomenon known as hysteresis.

Again the calculation of external costs requires knowledge of the social optimum: the correct marginal damage cost is the slope of the tangent at the optimum. Since that is not known, a possible first approximation is the slope of the line from pre-industrial conditions to the current state.

Problematic Impacts and the Use of MCA

For many impacts fairly reliable damage cost estimates are available and for others progress is being made, but some impacts defy quantification, as indicated in

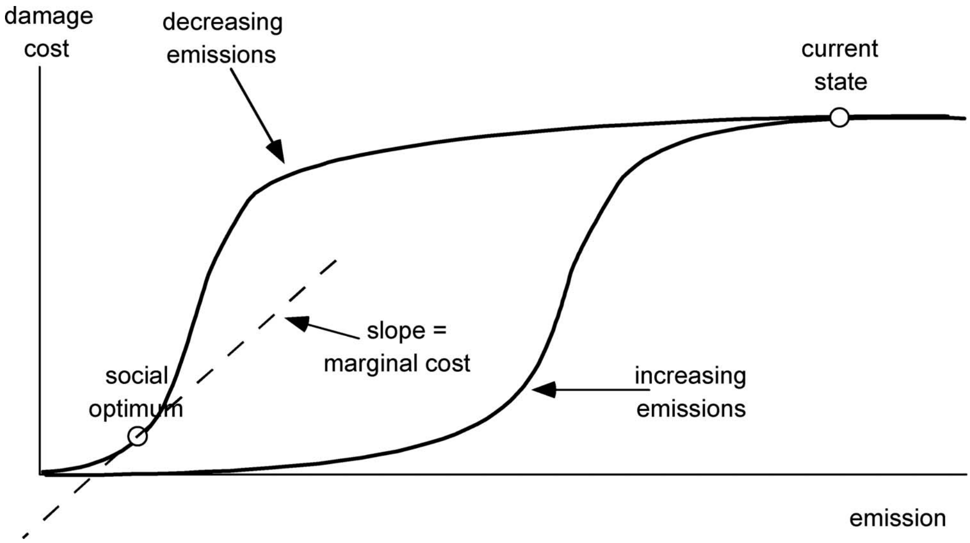

Figure 4. Non-linearities that can arise, for example, in ecosystems:

- There is a threshold zone above which the impacts increase sharply with the burden (e.g. emission of nitrates to water), reaching a plateau at high levels of the burden;
- To bring the system back to the original low impact level, the burden must be reduced to the threshold for decreases that is lower than the threshold for increases, a phenomenon known as hysteresis.

The appropriate marginal damage cost for policy applications is the slope of the tangent at the social optimum.

Table 4. Particularly troubling are the risks of nuclear power: the risk of a large accident, the storage of high level waste, and the risks linked to nuclear proliferation and terrorism are especially controversial. They are the reason why debates about nuclear power tend to remain inconclusive, despite the recognition by most experts that the impacts from normal operation of the nuclear fuel chain are negligible, especially when compared with fossil fuels. The attitudes are a reflection of optimism or pessimism about mankind's ability to manage this technology.

Risks of damage from storage of toxic waste, including nuclear waste, are extremely uncertain because they depend on the future management of the storage site. In principle they can be avoided completely if the site will be permanently maintained in safe condition without any leaks, and the waste is stored in a retrievable manner so the waste can be reprocessed and rendered less harmful once future technologies allow it. Thus the analysis requires the assumption of scenarios, and the results are of the 'what if' type rather than simple definite numbers.

Impacts that are potentially important but have not been quantified must not be forgotten in the final report. They have to be listed with a sufficiently detailed description in order to allow decision makers to take them into account in a multi-criteria analysis (MCA). For a test of MCA for environmental burdens, see for

Table 4. Impact categories and availability of damage cost estimates

Impact categories	**Status**
Global warming	Available, but very uncertain
Health impacts	Available for the most important pollutants
Damage to buildings and materials	Available for PM, NO_x, SO_2 and O_3
Loss of agricultural production	Available for NO_x, SO_2 and O_3
Acidification and eutrophication	Preliminary estimates for NO_x
Reduction of visibility	No reliable monetary values for Europe
Noise	Good estimates available for many sites
Visual intrusion	Extremely site-specific
Odour	Very site-specific
Depletion of resources	Already internalized?
Land use	Very site-specific
Soil erosion	No reliable monetary values for Europe
Storage of waste	Depends on assumptions about the future
Supply security (e.g. of oil)	Some estimates available, but uncertain
Accidents (non-nuclear)	Impacts on workers: already internalized by work contracts? Impacts on public: data available
Accidents (nuclear)	Some estimates available, but controversial
Nuclear proliferation and risks of terrorism	Too difficult

example the SusTools project of the EC, DG Research (Rabl *et al.*, 2004). For other examples of the application of MCA, see e.g. Hokkanen & Salminen (1997), Haastrup *et al.* (1998) and Vaillancourt & Waaub (2002).

Uncertainties

The uncertainties of damage cost estimates are large. Rabl & Spadaro (1999; Spadaro & Rabl, 2007) have carried out a formal analysis of the uncertainties and found that they can be characterized by lognormal distributions with multiplicative confidence intervals. For the classical air pollutants they correspond to a geometric standard deviation of about three, meaning that the 68% confidence interval ranges from a third of the median estimate to three times the median estimate. For pollutants such as toxic metals and for greenhouse gases the uncertainties are even larger.

In view of the large uncertainties the reader may wonder whether it is meaningful to use the damage costs as basis for decisions. The first reply is that even a threefold uncertainty is better than infinite uncertainty in the absence of analysis. Second, the uncertainties should not be looked at on their own. Rather, it should be asked what effect the uncertainties have on the choice of policy options, the key question being 'how large is the cost penalty if the wrong choice is made because of errors or uncertainties in the cost or benefit estimates?' In a recent paper Rabl *et al.* (2005) have examined the uncertainties from this perspective and their findings are very encouraging: 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

Table 5. Some of the problems in the calculation, presentation, interpretation or use of damage costs

Problem	Examples
(1) LCA inventories are based on past data, and results that were calculated for past technologies are applied to decisions about future versions of these technologies (see sub-heading *Lack of data for future emissions*, p. 95).	Damage cost estimates published by ExternE in 1995 and 1998 were based on technologies of the early or mid-1990s, but they were applied without adjustments to evaluate future options.
(2) Site-specific results are applied as if they were representative for the policy option in question.	Most of the results of ExternE before 2004 (with the exception of France) were based on specific sites, not chosen to be representative. Site dependence is extreme for hydro, very strong for impacts of primary pollutants, and significant for secondary pollutants.
(3) Inappropriate aggregation over burdens or over LCA stages.	Many reports show only the aggregated damage costs per kWh without breakdown for the contribution of each pollutant or breakdown by stage of the process chain (that is also the case in many LCA studies).
(4) Non-linearities.	Negative damage costs (i.e. benefits) have been reported for O_3 creation due to NO_x emissions. For the correct method (see *Non-linearity of impacts*, p. 95).
(5) Misunderstanding of the changes of damage cost estimates due to changes in methodology (progress of the science) and due to changes in the technologies.	The wide range of damage cost estimates in the literature is taken to imply that the results are too unreliable to be used (e.g. Stirling, 1997).
(6) The large uncertainty of damage cost estimates.	Some readers take it to imply that the studies are worthless, instead of looking at the effect of uncertainty on policy choices (for example of which see Holland *et al.*, 2005; Rabl *et al.*, 2005).
(7) The fact that quantification has not been possible for some impacts is either overlooked or, at the other extreme, taken to imply that the study is useless.	For many potentially toxic substances no ERFs are available (e.g. pesticides), and some impacts defy quantification (see sub-heading *Lack of exposure-response functions*, p. 93).

social cost (compared to the minimal social cost that one would incur with perfect knowledge) could be very large.

Furthermore, there are quite a few situations where the magnitudes of costs and benefits are so different that the conclusion of a CBA is not affected by the uncertainties. As an example, analysis under the CAFE programme found very little overlap between the probability distributions of the costs and the benefits for several scenarios, and thus a very high probability of achieving a net benefit (Holland *et al.*, 2005).

Frequent Problems in the Use of the Results

Table 5 lists some frequent problems that we have encountered in the literature of LCA and of external costs and in their application. Many of the problems arise from the all too human tendency to over-simplify and lose track of crucial detail. There is a dangerous fascination with simple global numbers such as 'the external cost of a kWh', instead of asking which information is relevant for which policy decision. Avoiding this type of problem is difficult; it is a real challenge for the communication of the results of damage cost studies: too little detail and readers misuse the results, too much detail and readers do not bother to look. Fortunately hypertext can help.

We have added some comments and suggested solutions have been added for some of the problems in Table 5. With regard to point 3, there has been much confusion about the stages to be included in an LCA and about the appropriate aggregation of the results. It is important to keep policy applications in mind when presenting the results. Since the analyst cannot foresee all applications, he/she should present the results with sufficient detail rather than merely showing aggregated results.

Points 5 to 7 lead many people to conclude that the damage cost estimates in the literature are too unreliable to be useful or even worse, that any quantification is meaningless. With regard to uncertainties, there is a comment about that objection in the previous section. To minimize the risk that results will be misunderstood, both the analyst and the potential user need to pay attention to important details. The analyst must be careful to provide sufficient documentation on assumptions and results, including an indication of uncertainties and other limitations (e.g. what is not included). It should also be remembered that policy makers may have very little scientific guidance unless some level of quantification is performed.

Conclusion

The objective of this paper has been to emphasize the need to take into account the link to policy applications when evaluating the impacts of environmental burdens. Starting with a discussion of impact pathway analysis (IPA) and LCA, the paper examined what type of information is necessary for what type of environmental decision. Numerous examples have been cited where the existing studies have not provided the appropriate information, and recommendations have been made for improved practice. Above all, the authors of damage cost studies should be careful to present and document the assumptions and the results in sufficient detail, including a breakdown by pollutant and process stage; uncertainties and other limitations should also be indicated. Users of such studies need to pay attention to such details, noting that policies should be targeted correctly to give a clear signal as to which source of a burden should be reduced by how much. A thorough and well-documented damage cost analysis can help provide a systematic assessment of the consequences of a decision before it is too late, and by clearly exposing the assumptions, it facilitates informed discussion of disagreements.

With regard to a general framework for environmental assessments, the following procedure is recommended:

(1) Define the options for the policy issue under consideration and identify the stakeholders who will be affected.
(2) Define the boundaries of the analysis appropriate for the policy issue.
(3) Use LCA, with the boundaries thus defined, to assemble an inventory of the burdens imposed by the activities of each option.
(4) Identify the impacts that may be significant.
(5) Carry out an IPA to quantify, as much as possible, the physical impacts (e.g. years of life lost) and the damage cost ('external costs') for each of these burdens.
(6) Examine options that reduce the burdens, and obtain data for their costs (abatement costs, as well as induced costs such as impact on consumer spending).
(7) Estimate uncertainties, and identify impacts and costs that defy quantification.
(8) Compare the quantified costs and benefits of the options (distinguishing short term and long term costs, identifying winners and losers, and examining distributional effects).
(9) Develop indicators for impacts and costs that have not been quantified (e.g. sustainability indicators).
(10) Involve the concerned stakeholders to estimate weighting factors for the various indicators to be used in the MCA.
(11) Carry out an MCA to derive aggregate scores for the considered options by combining the developed indicators with the weighting factors provided by the stakeholders.
(12) Select the most preferred option resulting either from the average score or by applying a majority rule, or through a constructive negotiation procedure between stakeholders.

The ground rule is: quantify as much as possible, then use MCA for any remaining impacts that are too uncertain or defy quantification entirely.

This approach has been tested in the recent SusTools project of the EC (Rabl *et al.*, 2004), with a case study on the impacts of nitrogen fertilizer (von Blottnitz *et al.*, 2006) and a case study on waste treatment (Rabl *et al.*, 2007). The results were discussed with stakeholders at workshops for the respective subject areas and the stakeholders were asked to assign weighting factors to the different criteria. The stakeholders were very pleased to participate in the workshops. The main problems encountered were (1) the difficulties many stakeholders had in choosing their weighting factors, and (2) getting a sample of stakeholders that would be a fair representation of the interests of society. Whereas the first can usually be alleviated by clear explanations of the impacts and extensive discussions among the workshop participants, the difficulties of getting a representative sample can be very great indeed. The very definition of what is representative poses problems: to what extent should individuals who do not care much about an issue be represented with equal votes as those who feel very concerned? How can one even decide whether a particular group of stakeholders is 'representative'?

In any case we note that the state-of-the-art is sufficiently well advanced for many of the most important impact categories, to render the results valuable for policy decisions, despite the large uncertainties. Quantification of impacts and damage costs is essential to help make public decisions more consistent between different types of decisions and different sectors (e.g. traffic safety and protection from pollution) and to make sure that the preferences of the population are correctly taken into account.

Finally, we emphasize that in many, if not most, environmental decisions the quantified costs and benefits are not the only criteria. For example, decisions about nuclear power need to address the risks of proliferation and accidents. For another example, if the internalization of the damage costs of power plants leads to price increases for consumers, some of the poor may no longer be able to heat their residence sufficiently; in such a case considerations of equity may necessitate a modification of a proposed regulation. Such criteria lie outside the analysis of external costs, but they should be taken into account at the decision stage by means of an MCA (multi-criteria analysis), preferably with involvement of the stakeholders.

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Notes

1 The present paper defines the socially optimal pollution tax in the narrow sense where it is equal to the marginal damage cost, but it is emphasized that the tax could be quite different with broader criteria, for example, a pollution tax might be set higher to bring in additional tax revenue (replacing less desirable taxes) or to bring about more rapid change of behaviour.

2 Sometimes the question of external benefits is raised. The benefits of the activities or goods themselves are internalized; for example, the driver of a car pays the (private) cost of driving because he/she receives the benefit. Certain activities can entail benefits that are not yet internalized, but they are rare, the most important example being agriculture, which can render a landscape more attractive if managed appropriately. Certain pollutants can provide minor benefits, for example NO_x emissions can provide fertilizer for agricultural crops; however, such benefits tend to be small compared to the overall harm of the respective pollutants.

3 The link between policy decision and analysis has frequently been overlooked. A typical example has been the reporting and use of external costs for electricity, when only a single aggregate number for cost per kWh (including all pollutants from the entire fuel chain) is cited or used. Such a number does not tell the operator of a power plant how much to reduce the emission of a particular pollutant, and it penalizes the power plant for pollutants emitted upstream or downstream.

4 Also known as dose-response function or, in the case of air pollutants, as concentration-response function.

5 With regard to the validation of EcoSense, the crucial elements are the calculations of atmospheric dispersion and chemistry; here comparisons with measured data have demonstrated sufficiently good agreement (see Section 4.4 of ExternE, 2000).

6 Such agreement is quite sufficient in view of the large overall uncertainties of any calculation of environmental external cots. The UWM is especially useful for a scoping analysis. In many cases sufficiently detailed data for an EcoSense calculation are not available.

7 Similar considerations hold for radon in buildings even though it is of natural origin and thus not an 'external cost' in the strict sense. Several different actors as well as the occupant could be asked to do something to reduce this risk, and in any case the occupant will ultimately bear the cost.

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