REPORT

External costs in the transport sector: A litterature review

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1 Introduction

The internalisation of external costs is in accordance with the polluter pays principle (PPP) which was adopted by the OECD already in 1972. The PPP is nowadays a cornerstone of both OECD and EU environmental policy in the sake of internalising external costs. However, the estimation of external costs is not an easy task based on one hand on the lack of consensus related to the methods to be used and on the other hand on the uncertainties related to the estimated values. This study is a literature review on external costs related to goods transport where a smorgasbord of methods, and models as well as some of the derived external costs are presented to give insight on the levels of these costs linked to health, the ecosystems as well as the built environment.

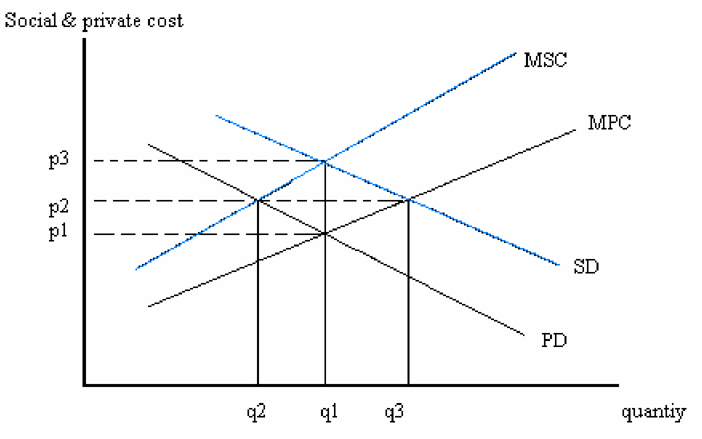
A further purpose of this study is to give guidelines on which approaches, methods as models to be used to estimate the external costs of goods transport for a number of companies.

This study is organised as follows: Section 2 is about the sources of externalities, section 3 presents the impact pathway method. Sections 4 and 5 are discussions of monetary valuation and estimation of external costs. In sections 6, 7, 8 and 9 congestion, accidents, noise as well as corrosion are presented together with methods and some estimates related to their external impacts. Furthermore, the ecosystem is discussed in section 10. Section 11 concludes.

2 External costs

According to Economics (2007), an externality is "an effect of a purchase or use decision by one set of parties on others who did not have a choice and whose interests were not taken into account". Externalities may be positive or negative. Negative externalities arise when an action by an individual or a group implies harmful effects on others such as air pollution effects on health, forest growth or fish reproduction. A positive externality may be the result of actions by an individual or a group benefiting others such as technological spill-over, which for instance can be generated by foreign direct investments in a developing country. The positive externality may also lead to higher social benefit, being the profit of an activity to a whole society, including not only the benefit to those members of the society being directly involved in the activity, but also the benefits to all other members. In the case of positive externalities, the social benefit is larger than the private benefit while the opposite applies for negative externalities. In general, economic inefficiency in resource allocation would be the result of a divergence between private benefits (costs) and social benefits (costs). When negative externalities are generated they should be internalized into the market economy. By internalizing i.e., by including the costs (or benefit) of the externality, environmental costs (or benefits) such as air pollution effects on human health and ecosystem, the externalities are allocated to the pollution sources and included in the economics of the activities causing the problem (e.g. industry, traffic, agriculture, energy production). Figure 1 illustrates this discussion.

Figure 1 A simple illustration of the effects of internalising positive and negative externalities with the assumption that the relations between prices, marginal costs and quantities are linear.



In the case of a beneficial externality, assume that at marginal social cost MSC defined here as the incremental cost to society and private demand PD, the quantity produced is q2 and the corresponding price is p2. Since this production generates positive externalities (e.g. vaccines), the production is expected to increase to the point where MSC equals the social demand SD i.e. q1. Observe that the price has been increased to p3 illustrating the higher willingness to pay for the product if the externalities are internalised to the market.

For the case of a negative externality, assume the production of for example a polluting activity, e.g. production of cement at quantity q3 with the price p2, reflecting only the marginal private cost i.e. the additional cost to the firm producing cement. In order to reach optimality the SD should equal MSC. This is done at quantity q1 and price p3. Hence, the result is a decrease in production being imposed by the corresponding increase in price which is the result of internalising the negative externality by way of a tax for example.

The illustration above shows the problem of negative externalities that often arise because of market and government failures. Market failures occur because markets for environmental goods and services do not exist, or when the markets do exist, the market prices underestimate their social scarcity values. However, markets can exist and function efficiently only when property rights on goods and services exchanged are well defined and transaction costs of exchange are small. According to economic theory the problem of externalities would not occur if property rights were properly defined for both private and public goods. In the case of public goods, this procedure would be impossible or rather impractical such as in the case of the European air, waters and ecosystems.

Abiotic resources

When resources are scarce such as fossil fuels, greater current use diminishes future opportunities. Scarcity imposes an opportunity cost referred to as marginal user cost (MUC). The MUC is the present value of these forgone opportunities at the margin. The MUC may be estimated as the present worth of the cost of replacing the depleted asset at some future date.

When resource depletion is of concern, the marginal social cost discussed above may be augmented by the MUC leading the marginal opportunity cost (MOC) of a resource such as:

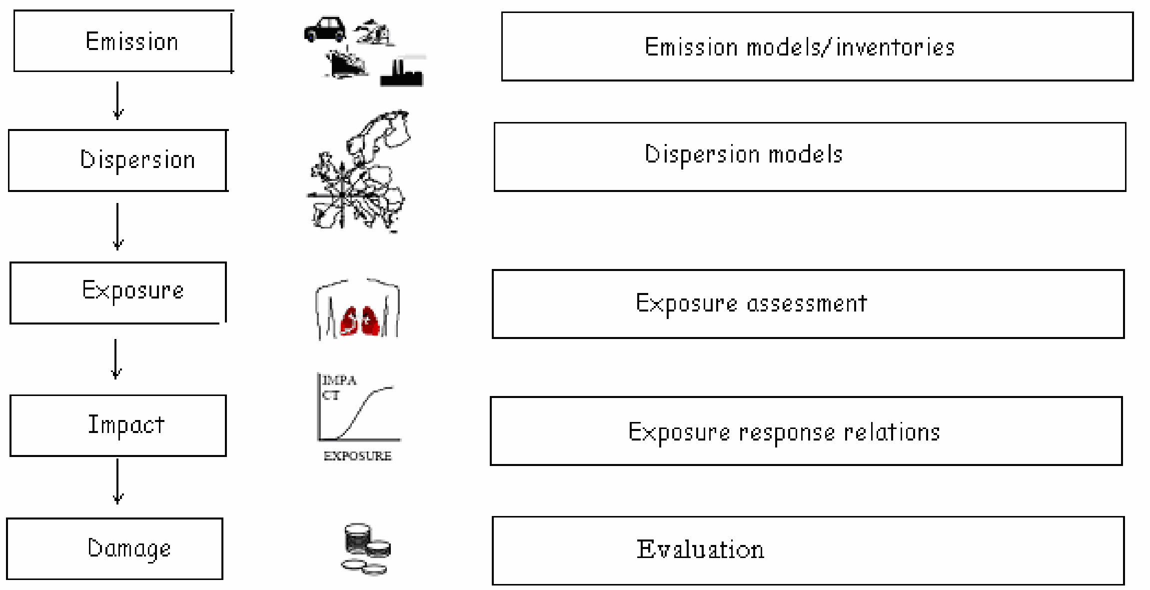
MOC= MSC+ MUC

3 The impact pathway of the emissions

To illustrate the concept of external costs, we first look at emissions to air from traffic. Figure 2 depicts the impact pathway of emissions where the starting point is the source of emissions. The different points leading to the evaluation of emission's impacts are the following:[1]

1. Emissions: The determination of emission factors of road transport is often the product of national, EU and international research.
2. Dispersion: The pollutants dispersed to the atmosphere from the transport sector are modelled using dispersion models e.g. EMEP (Co-operative Programme for Monitoring and Evaluation of the Long-range Transmission of Air pollutants in Europe)
3. Exposure: the impacts of transport externalities on health and the environment are location specific and based on traffic condition. Hence the exposure assessment relates to population and the ecosystem being exposed to the externalities.
4. Impact: The exposure response relations are based on epidemiological studies
5. Evaluation of impacts on both the humans and the ecosystem is based on valuation studies in order to monetise the external effects.

Figure 2: Impact pathway of emissions



In the case of emissions and the transport sector table 1 shows example of emissions factors used in the Swedish ExternE.

[1] Source: adapted from http://www.its.leeds.ac.uk/projects/spectrum/downloads/D6.pdf

Table 1: Emissions factors Swedish ExternE. (g/vehicle km)

	Private car gasoline	Private car Diesel	Lorry (>3,5 ton)	Bus	Motorcycle
CO	24.5	0.8	2.4	3.2	18.6
NO_X	0.91	1.24	9.7	10.4	0.08
NMVOC	2	0.21	1.53	1.57	5.82
CH_4	0.31	0.01	0.1	0.14	0.19
Exhaust particles	0.02	0.18	0.53	0.47	0.12
N_2O	0.043	0.023	0.03	0.03	0.002
NH_3	0.055	0.001	0.003	0.003	0.002
CO_2	359	266	1067	994	102
SO_2	0.009	0.004	0.018	0.017	0.003
Benzene	0.115	0.004	0.001	0.001	0.35
1-3 Butadiene	0.021	0.002	0.051	0.052	0.071
B(a)P	0.00000035	0.0000017	0.0000009	0.0000009	0.00000048

Source: Johansson et al (2003)

When it comes to exposion at the local level the estimations in the case of road transport are made based on particles PM_{10}, SO_2, Bensen, 1-3 Butadien and BaP. The population weighted exposure is calculated using the formula:

$$C_{pop-weighet} = \frac{\sum_{all\ grid} C_{Gridcell} Pop_{gridcell}}{Total\ population}$$

Where C= content and Pop = population.

4 Monetary valuation of different effects

The monetary valuation of effects being the result of negative externalities e.g. air pollution is not a straightforward procedure since many of the effects have no market value. In general we often talk about the total value of something. As shown in Figure 3, this total value is often composed of both use and non-use values. The use value is the value derived from actual use of a good or service. The non-use values, also referred to as “passive use” values, are values that are not associated with actual use, or even the option to use a goods or service.

The use value includes direct, non-direct and option values. The direct use value is the value attributed to direct utilisation of ecosystem services. Non-direct-use values or "functional" values relate to the ecological functions performed for example by forests, such as the protection of soils and the regulation of watersheds. Option value is the value that people place on having the option to enjoy something in the future, although they may not currently use it. On the other hand, the non-use values include both bequest and existence value. Bequest value is the value that people place on knowing that future generations will have the option to enjoy something. Existence value is the value that people place on simply knowing that something exists, even if they will never see it or use it. In order to assess these values, environmental economics uses several methods.

These methods may be based on:

- Stated preferences involving studies including questionnaires asking respondents for their willingness to pay such as in the case of contingent valuation and stated preference methods.

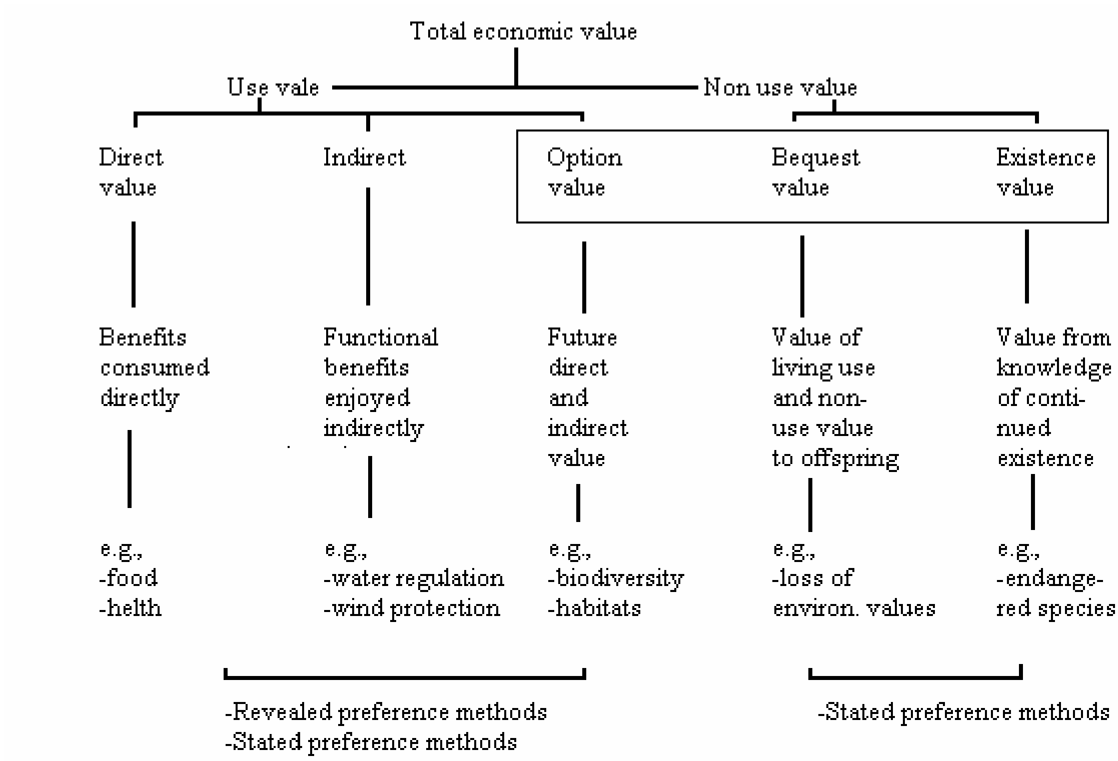
$$WTP = f(Y_i, E_i, A_i, Q_i) \qquad i = respondent$$

WTP may be function of income Y, education E, age A and environmental quality. Reduction of emissions for instance may lead to higher WTP.

Other methods are based on revealed preferences that are often based on consumers´ or producers' behaviour or actions such as:

- the hedonic price method is used to estimate the value of environmental effects on properties such as the effect of noise or air pollution on house prices;
- The production function method is used to estimate the value of the environmental effects on production such as the effect of ground-level ozone on the production of wheat or timber.
- Travel cost is used to estimate ecosystem values such as the value of a nature reserve or a historical site.

Figure 3: The total economic value



Source: Belhaj (2007)

In the case of health effects, other methods than stated or revealed preference ones can be used to estimate the impact of negative externalities. These methods may be HALY, DALY or QALY. The HALY is a Health Adjusted Life Year i.e. a generic term that includes the two most popular measures, the QALY or Quality Adjusted Life Year and the DALY or Disability Adjusted Life Year. The QALY is simpler. A value of quality of life is assigned from 0 (dead) to 1 (perfect health). The DALY is different in that the reference states are 0 for perfect health and 1 for dead, and it is estimated for particular diseases, instead of as a health state (for more details related to these methods see ENGRI (2004)). Further, other valuations of health effects are based on the value of statistical life or (VSL) or the value of lost years (VOLY) where the relation between the two is as follows:

The willingness to pay for Δs (the change in the risk to die) leads to the value of statistical life such as:

$$VSL = \sum_i WTP_i \;/\; \Delta s N$$

where N is the population at risk. Within the ExternE (Externalities of Energy - a research project of the European Commission) for instance, VSL is supposed to be equal to the discounted value of $VOLY$. Hence

$$VSL = VOLY \sum_{i=a}^{T} \left(aP_i \;/\; (1+r)^{i-a} \right)$$

where aPi is the conditional probability to live until year i for a person at age a. T is the maximum expected life length and r is the discount rate.

However, based on data scarcity to assign a value to a DALY a second best approach is proposed and used by Pearce et al. (2004). This approach is also in line with the CAFE CBA (2005). This second best approach combines VSL and DALY. This is because VSL is firmly grounded in economic theory being based on WTP.

In Pearce et al. (2004) an economic value is placed on a DALY in two different ways: The first approach looks at health expenditure. They assume that this expenditure is spent on avoiding and treating the causes of DALYs, they compute health expenditure per DALY. This approach, however, may lead to underestimation of damage costs

The second approach proceeds in the same way but notes that the 'value' of a DALY is greater than the healthcare costs incurred and must include the willingness-to-pay (WTP) of individuals to avoid the health states in question. They follow a procedure adopted in a World Bank study of pollution control and assign a WTP value to a DALY based on an 'anchor' estimate of the value of a statistical life' (VSL) and an implied value of a 'life year' (VOLY).

5 Estimation of external costs

The evaluation of avoided mortality from air pollution is a complex issue. From the EC DG Environment workshop (2000), the €$_{2000}$ 1 million is estimated to the value of preventing a statistical transport fatality of €$_{2000}$ 1,4 million (traffic accident with deadly outcome) adjusted to the fact that the persons dying from air pollution are mostly elderly (European Commission DG Environment, (2000)). The lower and upper estimates of the value of avoided fatality are €$_{2000}$ 650 000 and €$_{2000}$ 2 500 000. Table 2 brings together some values for morbidity.

Table 2: Monetary values for morbidity (€)

Impact	Value
Chronic bronchitis	168 840
Respiratory Hospital admission	4400
Congestive heart failure	3360
Chronic cough in children	240
Restricted activity day	120
Asthma attack	85
Cough	42
Minor restricted activity day	42
Symptom day	42
Bronchodilator usage	40
Lower respiratory symptom	8

Estimation of user costs

In the case of MUC values are very scarce. However, EPS model (Steen (2000) that is based on 'willingness to pay' uses rapeseed oil as a reference material to value fossil oil as a resource being a surrogate value of the MUC. This value is estimated to 0.507 €/kg (1999 prices).

6 Congestion

Traffic congestion costs consist of incremental delay, driver stress, vehicle costs, crash risk and pollution resulting from interference between vehicles in the traffic stream, particularly as a roadway system approaches its capacity. Each vehicle on a congested road system both imposes and bears congestion cost.[2] Various methods are used to quantify congestion costs. The most appropriate approach for many applications, although difficult to perform, is to calculate the marginal delay caused by an additional vehicle entering the traffic stream, taking into account the speed-flow relationship of each road segment. Another approach is to determine the user fee needed to reduce demand to design capacity, which reflects travellers' willingness-to-pay for road use. A third approach is to calculate unit costs of current expenditures on congestion reduction projects. In theory these three methods should produce similar cost values, assuming that roadway

[2] http://www.vtpi.org/tca/tca0505.pdf

capacity is expanded based on vehicle delay costs as reflected in vehicle users' willingness to pay, but in practice they often provide different results. Table 3 shows examples of congestion costs.

Table 3: Marginal external costs (€/carKm)

		Congestion
Urban	Peak	0.80
	Off-peak	0.25
Non-Urban	Peak	0.17
	Off-peak	0.53

Source: De Nocker et al. (2006).Note: Motorcycle = car/2; Bus = car x 2.

7 Accident externality pricing[3]

The goal of accident externality pricing is to make the driver internalise the external accident cost. There are two categories of accident costs that can be distinguished, the internal and the external costs. Internal accident costs are the costs that the driver takes into account. External accident costs are the cost the driver does not take into account. These costs concern the others who are affected by the accident as well as the costs to society. Therefore, the total marginal accident cost is the extra cost imposed by a user on all users (including him/her) and the general public due to his travel decision. If the number of accidents does not change due to a trip decision, there is no need to charge the user. However, evidence shows that if the traffic volume rises the number of accidents rises as well. Hence, there is a marginal effect.

The marginal cost with respect to the traffic volume Q for mode of transport j can be written as:

$$MC_j = \sum_i \sum_s \frac{\partial A_{ji}^s}{\partial Q_j}(a^s + b^s + c^s)$$

Where A_{ji} is the number of accident victims between mode *j* and *i*. Four types of accidents can be distinguished: fatal accidents, accidents with serious injuries, accidents with light injuries and accidents with only material damage. s =1…4 denote these different types. For each accident type the total accident cost per accident can be viewed as consisting of the willingness to pay for reducing the accident risk to zero, both on part of the motorists themselves (a^s), and on the part of their dependants, relatives and friends (b^s) as well as the direct accident costs borne by the rest of society (c^s).

8 External cost of road noise

Road noise may have effects on health in different ways:

[3] http://www.its.leeds.ac.uk/projects/spectrum/downloads/D6.pdf

-general nuisance
-difficulties to hear
-sleep disturbance

In order to estimate the external costs of noise both stated preferences based on contingent valuation studies as well as revealed preferences based mainly on hedonic price methods have been used. Table 4 shows the results of different studies

Table 4: Studies on road noise

Source	Country	NSDI hedonist % of property price /dB
Vainio (1995) Haolomo (1992)	Finland	0.36 0.98
Weinberger et al (1991)	Germany	0.5- 1.3
Colins et al (1994) Bateman et al (1999)	UK UK	0.65 - 1.28 0.20
Soguel (1994) Pommerehne (1988) Iten et al (1990)	Switzerland	0.91 1.26 0.9
Saelensminde et al (1994)	Norway	0.24 - 0.54
Lambert (1992)	France	1.0

Source: Howarth et al (2001). Note: NSDI=Noise sensitivity depreciation index i.e. % depreciation of house price of each decibel above the baseline level (50db)

Furthermore, based on a meta-analysis including several contingent valuation studies a functional form has been developed to estimate the value of decreased noise (Bertrand (1997)):

$$MWTP = e^{2.348 + 0.00000509Y + 0.0497\,n}$$

Where the marginal willingness to pay is a function of income *Y* and noise level *n*.

9 Corrosion

The total damage cost on material caused by acidification includes, besides the material's sensitivity of pollution levels, the quantity of different materials and the pollution levels such as (Andersson (1994)):

$$K_{i,j,m} = A_{i,j,m} * C_m\left(SO_{2i,j}, H_i^+\right)$$

where, K = Annual damage cost caused by acidification (€/year), A = Quantity of material (m^2), C = Annual damage cost caused by acidification per m^2 (€/m^2 year) as a function of SO_2 concentration and deposition of H^+. m = material, i = county and j = urban or rural area.
The annual damage cost parameter used in equation (above) is approximated by the increased costs caused by excess maintenance of the materials i.e.,

$$C_{tot,m} = \frac{p_m}{L_m}$$

where, P_m = Excess maintenance / replacement cost of material m (€/m^2) and L_m= Lifetime of material m as a function of the concentration of SO_2 and the deposition of H^+. The total cost estimates of corrosion are finally separated into the cost caused by dry deposition (SO_2 concentration) and the cost caused by wet deposition (H^+ deposition). The damage cost calculations caused by acidification in Sweden is based on Andersson (1994). The study focuses on the effect on buildings and estimates the changes of damage cost.

Table 5: damage cost in Swedish counties (million SEK)

County specific damage costs from air pollution.	EMEP grid cells in county	Annual SO_2 damage cost (SEK/$\mu gSO_2/m^3$)	H^+ damage cost (SEK/meq/m^2)
Stockholm county	5.5	21.2	5.0
Uppsala county	3	5.7	1.4
Södermanland county	2.5	5.8	1.4
Östergötland county	5	9.1	2.2
Örebro county	3.75	6.7	1.6
Västmanland county	2.75	5.4	1.3
Blekinge county	1.75	4.3	1
Skåne county	7.5	23.5	5.6
Värmland county	11.25	7.9	2.3
Dalarna county	16.75	10	2.4
Gävleborg county	9	8.2	2
Västernorrland county	10.75	8.7	2
Jämtland county	26	6.4	1.6
Västerbotten county	26.25	9.2	2.2
Norrbotten county	48.75	8.9	2.1
Jönköping county	4.25	9.1	2.2
Kronoberg county	3.5	6.5	1.6
Kalmar county	8.25	6.8	2
Gotland county	5	2.3	0.6
Halland county	2.5	7.5	1.8
Västra Götaland county	13.25	35.2	8.5

Source: Andersson (1994)

Another difference is that the concentration and deposition cost estimates are not divided into urban and rural areas. The assumption is that there are a limited number of SO_2 sources in the urban areas and that the concentration and depositions values for rural areas received from EMEP are a good approximation.

10 Ecosystem

Whilst some consensus has been reached with regards to estimating health effects, consensus on the external costs related to ecosystems is not reached depending on several factors. To start with is it the ecosystem service and not the effect on the ecosystem which is of interest? Other uncertainties are related to the definition of ecosystem services which may vary depending on locations and preferences as well as the establishment (or abolition) of critical loads.

For the estimation of ecosystem external costs both stated and revealed preferences may be used. For the revealed preferences several methods have been developed (see above) to monetize the external effects. However, depending on the pollutant stated preference studies may be scare such as in the case of heavy metals. The scarcity of the studies that may be used through benefit transfer may be based on the low awareness of the general public of the effects. Therefore alternative methods may be used to estimate proxies for the damage cost. There are especially two methods that may be used:

The first method is called the standard price method (Vermoote and De Nocker (2003) and the second one is called the Ecotax method (Johansson (1999) and Finnveden et al (2006)).

10.1 The standard price approach

This method estimates the revealed preferences of policy makers. It calculates the benefits of emission reduction – as perceived by policy makers - based on the abatement costs to reach a well-defined emission reduction target (Vermoote and De Nocker (2003). These costs are a proxy for the benefits that policy makers attribute to these reductions, as we assume that policy makers act as rational decision makers who carefully balance (their perception of) abatement costs of emission reductions with (their perception of) the benefits of these emissions. However, as the standard price approach is based on the current preferences of policy makers, as reflected in air quality policies, it cannot be used for cost-benefit analysis or policy advices related to these emission reduction policies. Nevertheless, this second-best method gives useful data for comparison of energy technology and fuels because it gives us 'shadow prices' for a non-market scarcity, protected ecosystems from acidification and eutrophication (ibid).

10.2 The ecotax method

The Ecotax method also called the valuation weighting method is a monetarisation method based on a tax system. The method relies on two basic assumptions. The first is that the members of parliament represent the will of the people, and the second is that the environmental tax system represents the priorities of the parliament.

This Ecotax method proceeds in the following way to estimate damage costs of certain emission. In cases where the tax or fee used is not on the reference substance of the characterization method (as is the case for many substances), the calculations are made according to the principle that a contribution to an impact category can be considered equally harmful independent of what caused it. The value of one extraction or emission may be translated into another extraction or emission contributing to the same impact category, by means of characterisation factors (Johansson (1999)).

For example, an emission of 1 kg of methane is, according to IPCC (1995), equivalent to 56 kg of carbon-dioxide over a 20 year time frame. Emissions of carbon-dioxide has the value 0.041 €/kg, and thus, the emission of methane receives the value (56 kg/kg * 0.041 €/kg) = 2.30 €/kg. The performance of such calculations results in weighting factors that include both the characterisation and the valuation sub-steps of impact assessment.

Hence, the functional form for the estimations is as follows:

$$X_x = (A_a / B_b) * Y_x$$

(X_x) represents the value of substance x, which is the one-step weighting factor searched for, (A_a) is the valuation weighting factor for substance *a* and (B_a) is the characterization factor for substance *a* and (Y_x) is the characterization factor for substance x, given by the characterization method.

Based on the Ecotax method as well as the Swedish taxes and fees the following one step weighting factors are estimated for:

- Aquatic ecotoxicity based on aquatic ecotoxicity potentials (AEPs) for emissions to water from Jolliet and Crettaz (1997);

- Aquatic ecotoxicity for metals released to soil and air;
- Terrestrial ecotoxicity for metals released to soil and air; and
- Global warming

Table 6: One-step weighting factors for global warming, based on the global warming potentials (GWPs) for different time frames from IPCC (1995) and on the tax on the carbon dioxide content of fossil fuels.

Trace gas	GWP	20 years One-step Weighting factor (€/kg) Based on 0.76 SEK/kg CO2	GWP	50 years One-step Weighting factor (€/kg) Based on 0.76 SEK/kg CO2	GWP	500 years One-step Weighting factor (€/kg) Based on 0.76 SEK/kg CO2
Carbon dioxide	1	0.08	1	0.08	1	0.08

10.3 Comparison of marginal costs

Table 7 shows marginal costs that are based on different methods i.e. EPS 2000 (Environmental Priority Strategies (Steen (2007)), the impact pathway method (Holland et al (no date).The Stern (2007) value is based on consequence analysis of climate change. The Ecotax method is based on Johansson (1999).

Table 8: Marginal external costs of emissions (€/kg)

	EPS 2000	Impact pathway[4]		Stern	Ecotax
		Regional	Urban		
CO_2	0.108	0.0024		0.12	0.08
NO_x	2.13	2.6	2.6		
SO_2	3.27	1.7	6*		
CO	0.331				
VOC		0.7	0.7		
NMVOC	2.14				
$PM_{2,5}$		1.7	33*		
PM_{10}	36				
Fossil oil reserves	0.507				

* city of 100 000 people, ½ mio x 5, 1 mio x 7,5

11 Concluding remarks

To apply the Polluters Pays Principle is not an easy task based on the fact that several methods may be used to estimate external costs related to health, the ecosystem as well as the built environment. The sources of uncertainties are related to both health effects as well as ecosystem impacts where some next best methods are used to estimate the external effects. Furthermore, depending not only on the methods but also on the applied approach the difference between the estimated external costs is shown to be significant.

[4] The recommanded marginal costs for CO_2 by ExternE are depicted in table below

	Minimum	Low	Central estimate	High	Maximum
CO_2	0.0001	0.0014	0.0024	0.0041	0.0164

The estimateds in this table are lower than the values that are used in a range of other recent studies (Mayeres et al (2001). The results presented here should therefore not be taken as final estimates. The impacts covered by the models used are only a fraction (of unknown size) of all climate change impacts. Particularly, large-scale disruptions, such as a breakdown of North Atlantic Deep-Water formation or a collapse of the West-Antarctic Ice Sheet or impacts in the 22nd century, are excluded from the analysis. The methodologies to estimate climate change impacts in a different future remain weak.

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