Valuing the Environmental Impacts of Electricity Generation: A Critical Survey

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Valuing the Environmental Impacts of Electricity Generation: A Critical Survey

Thomas Sundqvist and Patrik Söderholm

Abstract

This article provides a critical survey of a large number of studies carried out during the 1980s and 1990s that have focused on valuing the external, primarily environmental, costs associated with electricity generation. It discusses a number of conceptual, policy-related and, in some cases, unresolved questions in the economic valuation of these types of impacts. These include: (a) the definition of externalities; (b) the choices of scope, relevant parameter input assumptions, and methodology; (c) the role of 'green' consumer demand in replacing environmental cost assessments; and (d) the behavioural assumptions underlying environmental impact valuation. By analysing these issues we gain an increased understanding of the reasons for the wide disparity in external cost estimates reported in previous studies. The article also concludes that in cases where the results of electricity externality studies are utilised as a basis for policy purposes, a conflict between the economic efficiency criterion, its theoretical foundations and other – not necessarily less legitimate – goals of policy may exist.

Keywords: Electricity Generation, Environmental Impacts, Externalities, Valuation, Survey.

1. Introduction

Electric power production plays a vital role in modern societies, but it also gives rise to negative impacts on the environment such as the pollution of air, water and soil. A large number of regulations and economic incentives exist worldwide to promote the introduction of more environmentally benign power generation technologies. However, when implementing such policy efforts two questions normally arise: (a) what technologies should be considered environmentally benign; and (b) how does one find a proper balance between the benefits of electricity production and the costs of environmental degradation? During the last decades policy makers have shown an increased interest in the general recommendations found in the economics literature. According to this strand of research the answers to the above questions lie in applying economic non-market valuation techniques to the specific environmental (and non-environmental) impacts that can be labelled externalities. Formally an externality, i.e., an external cost or benefit, is defined as an unpriced and uncompensated side effect of one agent's actions (in our case electric utilities) that directly affects the welfare of another agent (Baumol and Oates 1988). Since these effects are not reflected in market prices, there exists a need to assist market processes by assigning them monetary values and in this way integrate them into private and public decision-making.

In the early 1980s studies that explicitly attempted to assess and value environmental impacts in the power sector began to emerge (e.g., Schuman and Cavanagh 1982). During the 1990s there was a surge in the number of externality valuation analyses conducted, in large part due to increased attention from policy makers in Europe, with the ExternE-project (EC 1995; 1999), and in the USA (e.g., Rowe et al. 1995; ORNL and RfF 1994–1998). The results and the methods of many of these studies have been utilised as inputs in important modelling work and have served as vehicles in developing additional methodological work in the environment and energy field (Krewitt 2002). For instance, past studies on how different environmental regulation schemes affect national energy systems have made use of external cost adders.[1] Still, so far the results from previous studies have only to a limited extent significantly affected actual policy decisions. Some authors argue that this is because electricity environmental impact studies may have raised more questions than they have answered, and that there exist important limits to their usefulness in deriving policy-oriented recommendations (e.g., Stirling 1997).

The main purpose of this article is to provide a critical survey of previous external cost studies in the power sector, and with this survey as a basis discuss a number of conceptual and still unresolved issues in the economic valuation of electricity related environmental impacts. A number of important issues are identified and discussed but overall we focus especially on two interrelated questions: (a) the wide disparity in external damage costs reported in previous studies, and the extent to which this represents a problem; and (b) the usefulness of previous valuation efforts for policy purposes. One of the main theses of the article is that in cases where the results from valuation studies are used for policy purposes, there may exist a conflict between the theoretical foundations of environmental impact valuation, the related choice of economic efficiency as a policy goal, and the more pragmatic (but not necessarily less legitimate) policy goals pursued in practice. Also, the question of whether the wide disparity of estimates represents a problem cannot be answered unless the circumstances under which the results are to be used for policy purposes are clarified.

Previous studies have also critically surveyed past research on electricity externalities. See, in particular, OTA (1994), Kühn (1996; 1998), Lee (1997), Ottinger (1997), Stirling (1997; 1998), Schleisner (2000), and Krewitt (2002). In contrast to these earlier survey studies, which typically focus on the procedure of generating impact estimates, we focus on broader theoretical issues and especially on the use of these estimates in policy making. Moreover, while earlier surveys focus on a few selected studies we consider the results, methods, and scope of about forty different externality studies. This enables us to draw more general conclusions about the usefulness and the limits of the work conducted in this field.

Before proceeding an important semantic issue should be clarified. In this article we focus primarily, but not solely, on the valuation of

1 See, for instance, Bigano et al. (2000) (for Belgium), and Vennemo and Halseth (2001) (for Norway).

environmental externalities in the power sector. An *environmental impact*, whether negative or positive, is not necessarily an *environmental externality* (see section 2) in the sense that it is not reflected in market prices. Still, the environmental impacts referred to in this article are mostly environmental externalities as well, and the terms are therefore largely used interchangeably. Still, in many instances we simply revert to the more general term *externality*, which embraces both environmental and non-environmental externalities.

In section 2 the theoretical and practical issues related to externality assessment and environmental valuation are briefly introduced. Section 3 presents an overview of a large number of previously conducted electricity externality studies. In section 4, we identify and analyse six fundamental and policy relevant issues raised by the empirical attempts at valuing electricity externalities. The article ends with a summary of the main findings in section 5.

2. The Valuation of Externalities in Theory and in Practice

Externalities occur as a result of both consumption and production activities and they are the causes of market failures, something that in turn leads to a resource allocation that is non-optimal from society's point of view. Hence, theoretically an externality causes a type of situation in which the First Theorem of Welfare Economics fails to apply, and markets fail at accomplishing Pareto efficiency.[2] Specifically, in the case of a negative externality, there exists a difference between the private and the social costs of an activity. The private costs facing a producer measure the best alternative uses of resources available as reflected by the market prices of the specific resources used by the producer. The social costs of production, however, equal private costs plus external costs (Figure 1), and measure the best alternative use of resources available to society as a whole. Since there is a lack of market for the external impact, a profit-maximising producer has no incentive to integrate this effect into the decision-making process. Thus private costs are lower than the social costs. The difference between private and external costs is, however, not 'fixed'. If the external costs can be 'internalised' (i.e., made private), decision-makers will have an incentive to undertake actions that help mitigate, for instance, the negative environmental impacts arising from electricity generation.

In his seminal work Coase (1960) demonstrates that bargaining between the polluter and affected agents can, under certain circumstances (such as low transaction costs and full information), internalise externalities and achieve an efficient outcome. However, in most cases, due to the large number of parties involved, some kind of government intervention is called for. One way of correcting the inefficiency of an external cost is the use of so-called

2 It should be noted that the market failure requirement ensures that the focus is on externalities that directly affect economic efficiency, i.e., technological (rather than pecuniary) externalities (Baumol and Oates, 1988).

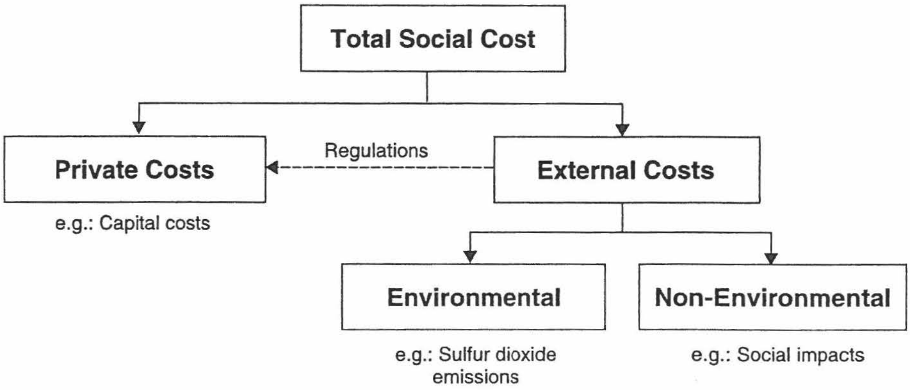

Figure 1: Total Costs to Society of a Productive Activity

Source: IEA (1995).

Pigovian taxes as originally suggested by Pigou (1924). This implies setting a tax equal to the value of the marginal external cost (at the optimum level of the activity in question) so that the private decision maker is provided with an incentive to economise not only on the traditional input factors but also on unpriced goods and services such as those provided by the natural environment. However, this solution to the externality problem requires that the tax authority is able to identify the external cost function. How do we go about assessing the size of this function, and hence the value of the damage caused by a negative externality (or the benefits incurred by a positive one)? The theoretical bases of such valuation exercises and the practical approaches used to empirically elicit these values are discussed below.

Externality Valuation in Economic Theory

The theoretical basis of the economic valuation of externalities is outlined in the welfare economics literature. This strand of research recognises that the economic value of a resource or service is ultimately a function of individual preferences, and the tool for analysing welfare changes is therefore utility theory. Our focus is on the economic valuation of an environmental good, but the general concepts are applicable to the valuation of all non-market goods. Following Perman et al. (1999), consider an individual that derives utility (U) from two goods, Q and Y. Q represents an environmental 'good' that the individual consumes and Y all other consumption possibilities available to the individual. Changes in the level of Q can refer to quantity changes or quality changes depending upon the type of environmental service involved. Assume that Q is a public good that is non-exclusive and non-divisible, so that the individual cannot adjust his or her consumption level. Now consider a project (e.g., a policy change) that, *ceteris paribus*, causes the environmental quality to increase (or improve) from Q' to Q' (see Figure 2). The project causes a positive change in the utility (or welfare) for the individual (represented by the move from indifference curve U_0 to indifference curve U_1). Given the presence of the project the individual is thus made better off. However, since utility is not directly observable and since environmental 'goods' are not part of market transactions we need to find alternative ways of assessing the value of this welfare change. Theoretically two standard monetary measures of quality based welfare changes are the compensating and the equivalent surplus.

To find the first of these measures, compensating surplus (*CS*), we start by noting that an increase in *Q*, everything else held constant, is equivalent to a reduction in the price of *Q*. And since the slope of the budget line is given by the relative price, the budget line (representing the individual's consumption possibilities) will change from a–b to a–c. In order to identify *CS* in Figure 2a we, hypothetically, constrain the individual at the pre-change environmental quality level (*Q'*) and utility level (U_0) by taking away just enough of the individual's income so that he or she can just afford to consume at the pre-change level (represented by the 'dotted' budget line d–e). *CS* is then a–d or the amount of money, that if foregone by the individual with the policy change, would result in him or her experiencing the pre-change level of utility or, in other words, the maximum willingness to pay (*WTP*) for the environmental improvement.

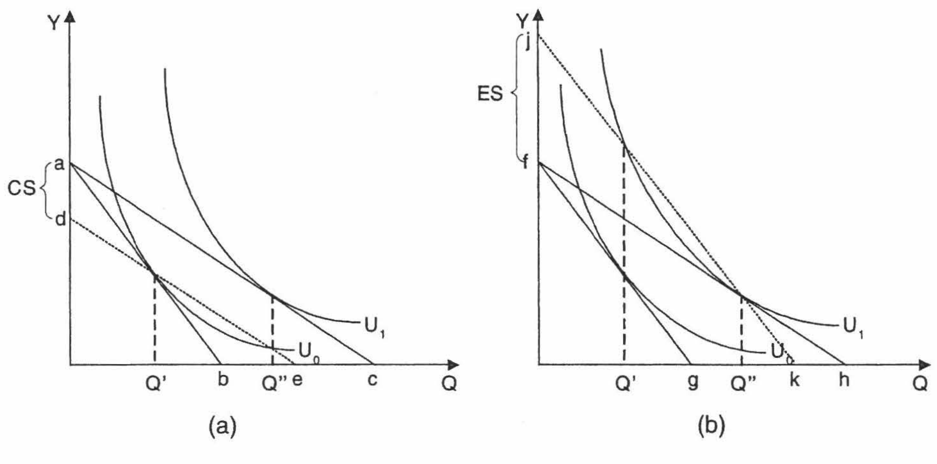

Figure 2: Compensating Surplus (CS) and Equivalent Surplus (ES)

The derivation of the equivalent surplus (*ES*) measure is presented in Figure 2b. *ES* is given by j–f and it is the amount that, at the original prices, would, if paid to the individual, result in him or her experiencing the same level of utility as the environmental improvement would have done, given that the environmental improvement, hypothetically, does not take place. Here *ES* thus equals the minimum willingness to accept (*WTA*) compensation for the environmental improvement not occurring. The interpretations of the *CS* and *ES* measures are reversed in the case of environmental quality deteriorations; *CS* is then equal to the minimum *WTA* and *ES* would be the maximum *WTP*.

In empirical studies it is generally the case that *WTA* measures tend to be substantially higher than *WTP* measures for the same change (e.g., Kahnemann et al. 1990). Thus, the choice of *WTP* or *WTA* as a measure of economic value may significantly affect the size of the resulting valuation estimate. Even though there exist theoretical reasons for this difference (e.g., Hanemann 1991), *WTP* is generally being advocated as the most appropriate measure of changes in welfare, primarily since *WTA* is not constrained by income and therefore creates an incentive problem (e.g., Arrow et al. 1993).

In sum, the economic valuation of many environmental (and non-environmental) impacts, builds on the assumption that people seek to satisfy their preferences, which are exogenously determined, complete, continuous, and ethically unchallengeable (subjective). The environment is essentially treated as any other private commodity, and people are willing to consider tradeoffs in relation to the quantity or quality of environmental 'goods'. According to the welfare economics literature the appropriate role of policy in the field of energy externalities would be to aggregate the monetary estimates of individual preferences and weigh them against other (more tangible) economic benefits and costs. Thus, the economics of non-market valuation builds on: (a) clear but also relatively restrictive behavioural assumptions (i.e., utility maximisation); (b) a sense of society as the sum of the preferences of its individual members; and (c) a view of the task of public policy involving the internalisation of external impacts and with utilitarianism as the ethical principle guiding social choice.

Externality Valuation in Practice

In practice there are two basic methodological approaches used for the valuation of externality impacts in the energy sector: the abatement cost approach and the damage cost approach.

The *abatement cost approach* uses the costs of controlling or mitigating damage or the costs of meeting legislated regulations as an implicit value of the damage avoided. The rationale behind this approach is that legislatures are assumed to have considered the willingness of the public to pay for alleviation of the damage in setting the standard, thus providing a revealed preference damage estimate no less reliable than the more direct valuation methods (see below). Pearce et al. (1992) stress that one of the serious caveats with the approach is that it relies on the rather strong assumption that these same decision makers make optimal decisions, i.e., they know the true abatement and damage costs. Figure 3 illustrates this problem. It displays the marginal abatement cost curve (*MAC*) and the marginal damage cost curve (*MDC*) resulting from some emissions (*E*). Thus, increased abatement is equivalent to lowered emissions (i.e., damage). Given that the curve *MDC* shows the true disutility of the damage done by the emissions, and if decision makers set a maximum standard of emissions at E_3, the abatement cost will underestimate the true damage cost, while if only emissions up to E_1 are

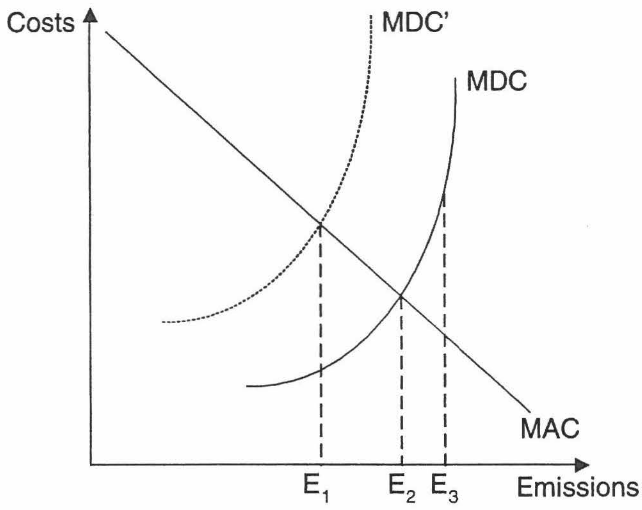

Figure 3: Marginal Abatement and Damage Costs

permitted the abatement cost will provide an overestimation. Only at E_2 marginal abatement costs correctly measure marginal damage costs. A necessary condition for social optimality, as Joskow (1992) notes, is that the abatement costs used are derived from the pollution control strategy that provides the least cost of control. If not, the estimates cannot adequately reflect damage costs.

Another limitation of the abatement cost approach, as noted by Bernow and Marron (1990), is that society's preferences change over time as information, analysis, values and policies change. Hence, past revealed preferences might bear little relation to actual impacts today and their current value to society. For instance, the implicit value of CO_2 emissions indicated by a revealed preference analysis would in many cases be very low since there still exist relatively few regulations targeted towards this problem.[3] This built-in 'tautology' of the approach means that estimates need to be constantly revised as regulations and policies change. More importantly perhaps, since policy is (per definition) optimal the abatement cost analysis provides no room for relevant policy implications, and one must therefore question why the analysis is needed in the first place.

The *damage cost approach* is aimed at measuring the net economic damage arising from negative externalities by focusing more or less directly

3 One alternative that is often advocated (see, for instance, Ottinger 1997), is to use control costs for existing (but not necessarily required) technologies (e.g., carbon sequestration in the case of CO_2 emissions). However, these estimates may not bear any relation to people's preferences towards the environment. The relevant policy question is whether people value the environment high enough so that the use of these control methods can be motivated, and this question cannot be answered by equalling held values with the control costs.

on explicitly expressed preferences. This approach can be subdivided into two main categories: top-down and bottom-up. Top-down approaches make use of highly aggregated data to estimate the environmental costs of, say, a particular pollutant. Top-down studies are typically carried out at the national or the regional level, using estimates of total quantities of pollutants and estimates of total damage caused by the pollutants. Specifically some estimate of national damage is divided by total pollutant depositions to obtain a measure of physical damage per unit of pollutant (Figure 4). These physical damages are then attributed to power plants and converted to damage costs using available monetary estimates on the damages arising from the pollutants under study. The main critique against the top-down approach is that it 'generically' cannot take into account the site specificity of many types of impacts, nor the different stages of the fuel cycle. Another argument that has been raised against the approach is that it is derivative since it depends mostly on previous estimates and approximations (Clarke 1996).

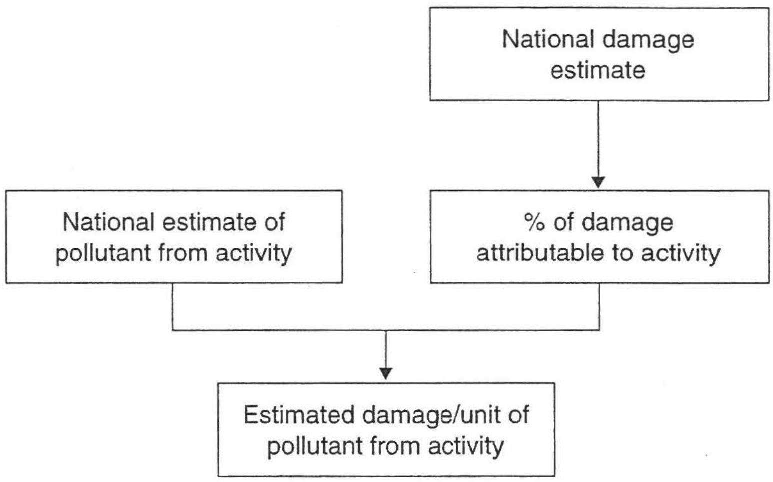

Figure 4: The Top-Down Approach

Source: EC (1995).

In the bottom-up approach environmental damages from a single source are typically traced, quantified and monetised through damage functions/ impact pathways (see Figure 5). This method makes use of technology-specific data, combined with dispersion models, information on receptors, and dose-response functions to calculate the impacts of specific externalities. The bottom-up approach has been criticised since applications of the method have unveiled a tendency for only a subset of impacts to be included in assessments, focusing on areas where data is readily available and where, thus, impact pathways can easily be established. Consequently bottom-up studies tend, it is argued, to leave out potentially important impacts where data are not readily available (Clarke 1996). Also, Bernow et al. (1993)

caution that the bottom-up approach relies on models that may not adequately account for complexities in 'the real world', especially noting that there may be synergy effects between pollutants and environmental stresses, and that there may be problems in establishing the timing of effects (i.e., between exposure and impact). The argument is hence that bottom-up approaches may not be sufficiently transparent. Still, this is the approach that, due to its focus on explicit estimates of economic welfare (rather than implicit such as in the abatement cost approach), appears to be most in line with economic theory. As is evident by the methodological choices of recent externality studies it is also the most preferred approach to the empirical assessment of externalities in the electricity sector (see section 3).

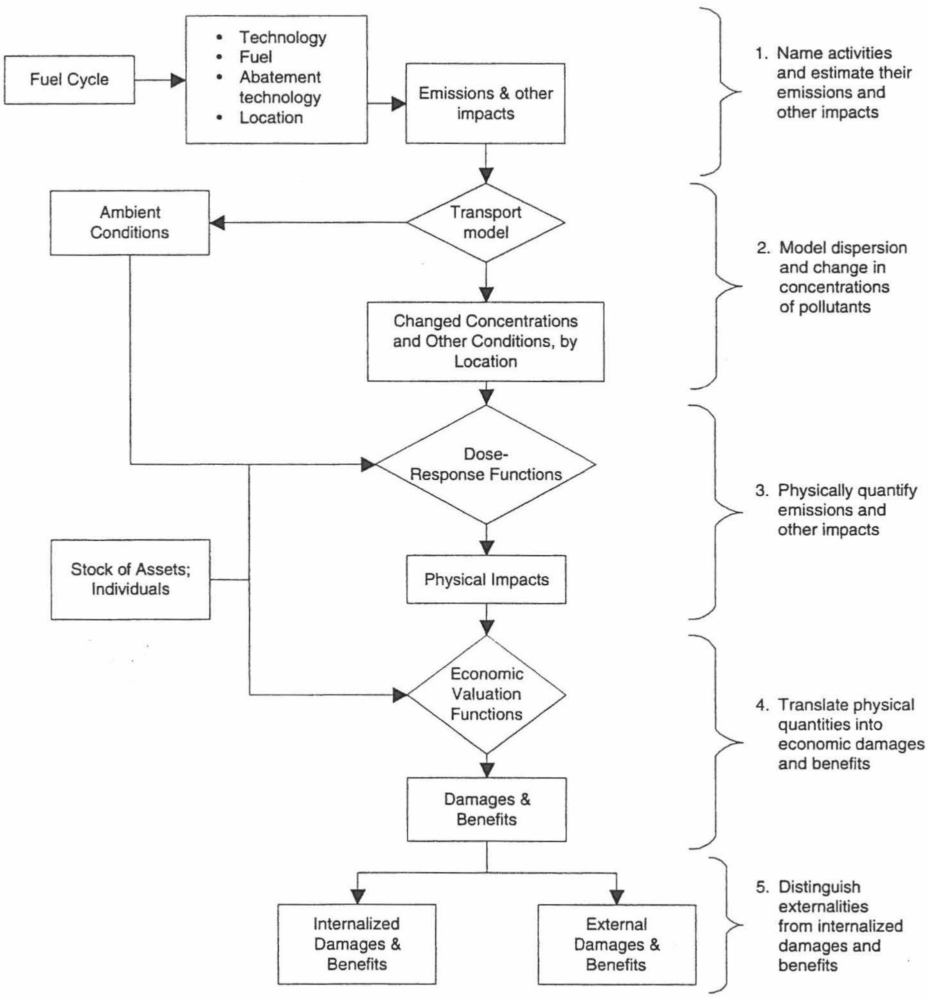

Figure 5: The Impact Pathway (Bottom-Up Approach)

Source: ORNL and RfF (1994).

There are several ways of addressing the problem of placing a monetary value on externalities in general and environmental impacts in particular. The first two approaches discussed above (abatement cost and top-down damage cost) directly give a monetary estimate of the damage associated with an environmental impact. The third approach, bottom-up damage cost, however, needs to translate the identified and physically quantified impacts into monetary terms. Generally it can be said that whenever market prices can be used as a basis for valuation, they should be used. However, since externalities by definition are external to markets, most impacts from externalities are not reflected in existing prices. Consequently, any attempt to monetise an environmental impact using bottom-up damage costing needs to rely on impact valuation methods. These methods can be sub-divided into *direct* and *indirect* methods. Figure 6 illustrates the various methods available for monetising environmental impacts.[4]

Even if no information is available from existing markets, it may be possible to derive values using *direct methods* that simulate a market. These

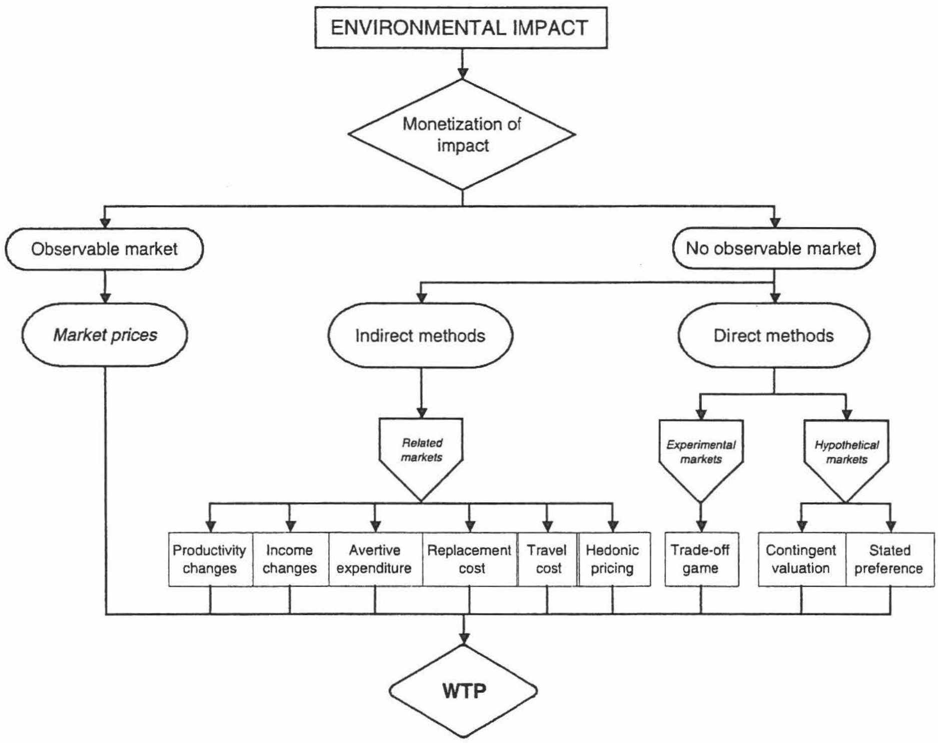

Figure 6: Overview of Impact Valuation Methods

4 There exists an extensive literature on different environmental valuation methods, and to review this in detail here would be beyond the scope of this paper. For an excellent overview, however, see Garrod and Willis (1999).

methods are direct in the sense that they are based on direct questions about – or are designed to directly elicit – *WTP*. An important advantage of the direct methods is that they can assess total economic values, i.e., use as well as non-use values (such as existence values). Well-known direct valuation methods include contingent valuation and stated preference (e.g., choice experiment).

None of the *indirect methods* can assess non-use values of the environment; they are based on the actual (rather than the hypothetical) behaviour of individuals. Either the environmental values show up as changes in costs or revenues on observable markets or in markets closely related to the resource that is affected by the environmental impact. The damage is thus valued indirectly using a relationship between the environmental impact and some good that is traded in a market. Examples of indirect valuation methods are hedonic pricing, travel costs, and replacement costs.

There are also methods that do not easily fit into the categories discussed above but that may nevertheless prove useful. The first of these, so-called benefit transfers, does not involve any valuation in itself. Benefit transfers instead make use of the results of previous studies that have derived monetary estimates for the environmental impact in question. That is, a study may utilise the results from another valuation study and adjust them for use in the present context. Economic values may also be assessed through opportunity costs, i.e., the net benefit of an environmental service. For example, a hydroelectric development of a river affects the recreational possibilities in the river. The opportunity cost of the development is then the forgone net benefits of the affected recreational activities in the river.

To sum up, it is clear that there exists an abundance of methods and techniques to approach the problem of monetising external costs. These methods may, however, as illustrated in Table 1, only be useful under specific circumstances and for specific impacts. As a result one single method may

Table 1: Relevance of Methods to Value Specific Effects

	Resource Degradation	*Pollution*	*Recreation*	*Natural Amenity*	*Work Environment*	*Non-use Benefits*
Indirect Methods:						
Productivity changes	!!	!		!	!!	
Income changes		!!			!!	
Avertive expenditure	!	!!	!!	!		
Replacement cost	!			!		!
Travel cost			!!	!!		
Hedonic pricing	!!	!!	!	!	!	
Direct Methods:						
Trade-off game	!		!	!		!
Contingent valuation	!		!	!		!!
Stated Preference	?	?	?	?	?	?

!! = Highly relevant, ! = Relevant, ? = Possibly relevant

Source: Adapted from Binning et al. (1996).

not permit all of the impacts to be addressed, and this necessitates the use of several methods in the assessment. What complicates things further is that the types of externalities that arise from various forms of electricity production also differ. Thus, since the types of externalities differ among fuels, different methods may have to be utilised in the monetisation of impacts for the variety of fuels. This is especially a problem if different methods tend to yield different results, thus producing environmental impact estimates that are incomparable with those of other methods. If this is the case, it may be hard to draw reliable conclusions about the ranking of different fuel sources in terms of external costs.

3. A Brief Overview of Previous Electricity Externality Studies

A considerable number of externality studies were carried out during the 1980s and 1990s. The focus in this survey is on studies, whose aim has been to assess the total external costs (and in some cases benefits) per kWh of different electric power technologies. Some studies were therefore deemed to be irrelevant for the present purpose since they only covered one specific impact, e.g., Fankhauser (1993) which only assesses global warming impacts, and some were identified but could simply not be obtained (e.g., BPA 1986). Table 2 provides an overview of about forty externality studies covered in the analysis.[5] An inspection of the different externality assessments laid out in Table 2 reveals several conceptual issues of importance, out of which five will be stressed here.

First, most of the fuel sources available for power generation have been addressed in previous valuation efforts, including coal, oil, natural gas, nuclear, hydro, wind, solar, biomass, and in a few cases lignite, waste incineration, geothermal, peat and orimulsion. However, most studies focus on the traditional fuels, such as coal and nuclear. There is thus a tendency to focus on existing technologies rather than on the technologies generally expected to play a significant role in the future (i.e., wind, biomass and so on.). In many cases this is understandable given that empirical data clearly are more available for existing (rather than emerging) technologies. Nevertheless, an important goal of externality valuation in the power sector has been to 'level the playing field' in the selection between traditional and new generating technologies, and this would probably require a stronger focus also on promising but not yet commercialised technologies.

Second, a majority of the studies have been carried out for the developed world (mostly for Western Europe and the USA). Thus, only in some rare cases the focus has been on developing countries where the need for additional power capacity is by far the greatest (e.g., IEA 1998). There are also reasons

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5 All monetary estimates presented in this article have been converted into US Dollars (1998) using mean exchange rates and the US Consumer Price Index. This process has not always been straightforward since the base years used in the studies are not always explicitly stated. Whenever this problem arose, the year of publication was used as a proxy for conversion and this may have lead to somewhat biased estimates.

Table 2: Overview of Externality Studies

Study	*Country*	*Fuel*	*Externality Estimate (US cents/kWh 1998)*	*Method*
Schuman & Cavanagh (1982)	US	Coal	0.06-44.07	Abatement cost
		Nuclear	0.11-64.45	
		Solar	0-0.25	
		Wind	0-0.25	
Hohmeyer (1988)	Germany	Fossil fuels	2.37-6.53	Damage cost (top-down)
		Nuclear	7.17-14.89	
		Wind	0.18-0.36	
		Solar	0.68-1.03	
Chernick & Caverhill (1989)	US	Coal	4.37-7.74	Abatement cost
		Oil	4.87-7.86	
		Gas	1.75-2.62	
Bernow & Marron (1990); Bernow et al. (1991)	US	Coal	5.57-12.45	Abatement cost
		Oil	4.40-12.89	
		Gas	2.10-7.98	
Hall (1990)	US	Nuclear	2.37-3.37	Abatement cost
Friedrich & Kallenbach (1991); Friedrich & Voss (1993)	Germany	Coal	0.36-0.86	Damage cost (bottom-up)
		Nuclear	0.03-0.56	
		Wind	0.02-0.33	
		Solar	0.05-1.11	
Ottinger et al. (1991)	US	Coal	3.62-8.86	Damage cost (bottom-up)
		Oil	3.87-10.36	
		Gas	1.00-1.62	
		Nuclear	3.81	
		Hydro	1.43-1.62	
		Wind	0-0.12	
		Solar	0-0.50	
		Biomass	0-0.87	
		Waste	5	
Putta (1991)	US	Coal	1.75	Abatement cost
Hohmeyer (1992)	Germany	Fossil fuels	11.12	Damage cost (top-down)
		Nuclear	7.01-48.86	
		Wind	0.12-0.24	
		Solar	0.54-0.76	
Pearce et al. (1992)	UK	Coal	2.67-14.43	Damage cost (top-down)
		Oil	13.14	
		Gas	1.05	
		Nuclear	0.81	
		Hydro	0.09	
		Wind	0.09	
		Solar	0.15	
Carlsen et al. (1993)	Norway	Hydro	2.68-26.26	Abatement cost
Cifuentes & Lave (1993); Parfomak (1997)	US	Coal	2.17-20.67	Abatement cost
		Gas	0.03-0.04	
ORNL & RfF (1994-1998)	US	Coal	0.11-0.48	Damage cost (bottom-up)
		Oil	0.04-0.32	
		Gas	0.01-0.03	
		Nuclear	0.02-0.12	
		Hydro	0.02	
RER (1994)	US	Oil	0.03-5.81	Damage cost (bottom-up)
		Gas	0.003-0.48	
EC (1995)	Germany	Coal	2.39	Damage cost (bottom-up)
		Oil	3	
		Lignite	1.37	

Table 2: continued

Study	*Country*	*Fuel*	*Externality Estimate (US cents/kWh 1998)*	*Method*
—	France	Nuclear	0.0003-0.01	Damage cost (bottom-up)
—	Norway	Hydro	0.32	Damage cost (bottom-up)
—	UK	Coal Gas Wind	0.98 0.1 0.11-0.32	Damage cost (bottom-up)
Pearce (1995)	UK	Coal Gas Nuclear	3.02 0.49 0.07-0.55	Damage cost (top-down)
Rowe et al. (1995)	US	Coal Oil Gas Nuclear Wind	0.31 0.73 0.22 0.01 0.001	Damage cost (bottom-up)
van Horen (1996)	South Africa	Coal Nuclear	0.90-5.01 1.34-4.54	Damage cost (bottom-up)
Bhattacharyya (1997)	India	Coal	1.36	Damage cost (bottom-up)
Ott (1997)	Switzerland	Oil Gas Nuclear Hydro	12.97-20.57 8.85-13.22 0.62-1.50 0.25-1.50	Damage cost (top-down)
Faaij et al. (1998)	Netherlands	Coal	3.98	Damage cost (top-down)
—	Netherlands	Coal Biomass	3.84 8.1	Damage cost (bottom-up)
EC (1999)	Austria	Gas Hydro Biomass	0.88 0.02 1.54-7.56	Damage cost (bottom-up)
—	Belgium	Coal Gas Nuclear	3.22-67.72 0.67-9.73 0.02-0.79	Damage cost (bottom-up)
—	Denmark	Gas Wind Biomass	0.99-11.19 0.08-0.51 2.34-12.55	Damage cost (bottom-up)
—	Finland	Coal Biomass Peat	1.07-18.15 0.83-2.00 0.69-1.69	Damage cost (bottom-up)
—	France	Coal Oil Gas Biomass Waste	9.61-29.45 11.79-39.93 2.70-7.68 0.82-2.51 22.17-68.73	Damage cost (bottom-up)
—	Greece	Oil Gas Hydro Wind Biomass Lignite	2.07-19.89 0.57-4.97 0.71 0.31-0.80 0.14-3.43 3.67-36.54	Damage cost (bottom-up)

Table 2: continued

Study	*Country*	*Fuel*	*Externality Estimate (US cents/kWh 1998)*	*Method*
—	Germany	Coal	2.38-23.67	Damage cost (bottom-up)
		Oil	5.30-35.16	
		Gas	0.83-9.55	
		Nuclear	0.08-1.45	
		Wind	0.05-0.31	
		Solar	0.08-1.69	
		Biomass	3.78-13.19	
		Lignite	2.83-56.57	
—	Ireland	Coal	6.16-31.90	Damage cost (bottom-up)
		Peat	4.62-5.32	
—	Italy	Oil	3.24-24.52	Damage cost (bottom-up)
		Gas	1.21-11.78	
		Hydro	0.47	
		Waste	—	
—	Netherlands	Coal	1.68-24.48	Damage cost (bottom-up)
		Gas	0.43-9.65	
		Nuclear	1.03	
		Biomass	0.49-2.86	
—	Norway	Gas	0.26-8.04	Damage cost (bottom-up)
		Hydro	0.32	
		Wind	0.07-0.35	
		Biomass	0.33	
—	Portugal	Coal	3.69-30.22	Damage cost (bottom-up)
		Gas	0.28-8.74	
		Hydro	0.03-0.07	
		Biomass	1.53-8.52	
—	Spain	Coal	4.64-32.60	Damage cost (bottom-up)
		Gas	7.13-9.53	
		Wind	0.24-0.34	
		Biomass	2.41-22.09	
		Waste	3.58-26.19	
—	Sweden	Coal	0.84-16.93	Damage cost (bottom-up)
		Hydro	7.83-18.54	
		Biomass	0.35-0.60	
—	UK	Coal	4.06-33.01	Damage cost (bottom-up)
		Oil	3.22-22.10	
		Gas	0.73-10.21	
		Wind	0.17-0.34	
		Biomass	0.72-3.22	
		Orimulsion	2.94-24.20	
Hirschberg & Jakob (1999)	Switzerland	Coal	4.54-23.16	Damage cost (bottom-up)
		Oil	5.13-26.09	
		Gas	1.17-8.06	
		Nuclear	0.29-1.90	
		Hydro	0-1.76	
		Wind	0.15-0.88	
		Solar	0.15-2.20	
		Biomass	3.67-8.50	
Maddison (1999)	UK/Germany	Coal	0.31/0.71	Damage cost (bottom-up)
		Oil	0.78	
		Gas	0.13	
		Lignite	0.73	

to believe that valuation estimates should differ substantially between developing and developed countries. In the developing countries incomes are lower, and the environmental effects of power production may be fundamentally different. An important example of the latter is the environmental externalities stemming from hydropower development. For instance, hydroelectric development in a temperate climate may give rise to global warming impacts due to mouldering of vegetation left in the reservoir, while hydroelectric development in colder climates will not (e.g., Moreira and Poole 1993). This raises serious concerns about transferring environmental values from studies conducted in, say, Western Europe, for use in a developing country context.

Third, examining the methodologies utilised over time reveals that the bottom-up damage cost approach seems to have become the dominant paradigm, while the abatement cost and top-down approaches were predominantly used in the 1980s and early 1990s (Figure 7). An important reason for this development is that the national implementation phase of the ExternE project (EC 1999), relies solely on damage cost bottom-up models, and these studies together represent a large share of the total number of projects conducted during the latter part of the 1990s. This also indicates, however, that the bottom-up model has been accepted as the most appropriate method with which to assess power generation externalities. The ExternE project has largely served as a vehicle in the methodological development of externality valuation. The scientific quality of the ExternE work as well as the methodologies used has been well accepted at the international level, and

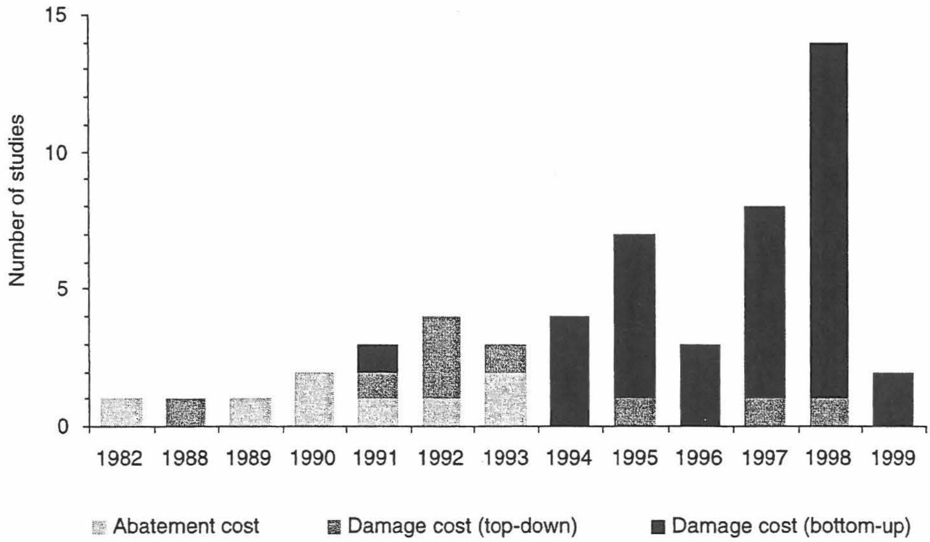

Figure 7: Methodological Choice Over Time

Source: Sundqvist (2000).

many followers rely heavily on the numbers and the methods presented (Krewitt 2002). However, this development raises the question of whether the choice of methodological approach (between abatement costs and damage costs) matters for the results. In section 4 we revert to this question in more detail and suggest that this choice very well may matter, which in turn raises important concerns about the reliability of external cost valuation exercises in the power-generating sector.

Fourth, as can be seen in Figure 8 the disparity of external cost estimates is considerable when compared across different studies (note the use of logarithmic scale). Figure 8 is based on the results from 63 externality studies[6]

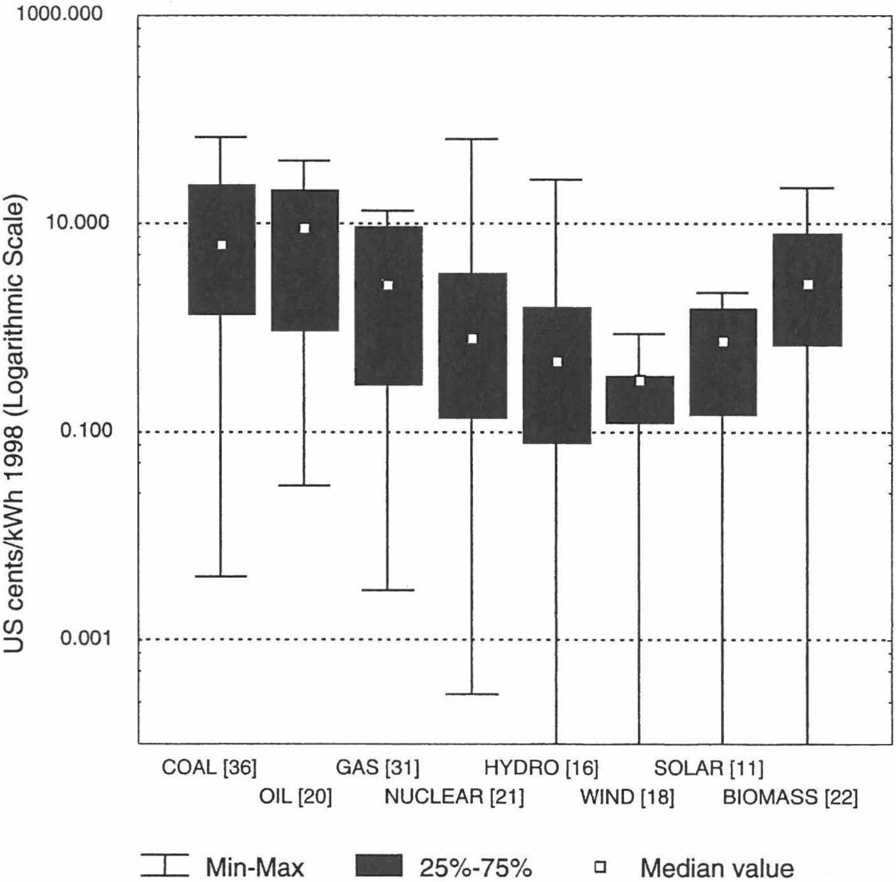

Figure 8: Range of External Cost Estimates

Sources: Table 2 and Sundqvist (2000).

6 These include those outlined in Table 2 and a number of additional studies that are not presented here in detail (see, however, Sundqvist 2000). Most of the latter observations build on secondary sources in which the details (i.e., methodology, scope etc.) of the studies are not reported.

and the numbers in 'square-brackets' show the total number of observations for each fuel source. The ranges also intertwine making the ranking of various fuels with respect to externality impacts a difficult task. Still, some tentative conclusions can be drawn. For instance, the results suggest that fossil fuel-fired power, in particular coal and oil, gives rise to the highest external costs, while some of the renewable energy sources, solar, wind and also hydropower, tend to have the lowest.

It is, however, also of interest to note that biomass-based electric power appears to incur substantially higher external costs than the other renewable energy alternatives.[7] This notion, if valid, questions some of the recent policy initiatives that attempt at encouraging the use of renewable energy *per se*, i.e., without distinguishing between the different renewables, through green certificates and competitive bidding systems.

For a specific fuel source the difference between low and high values is substantial and this is also true if one looks at single studies; the ranges reported can often vary from a tiny fraction of electricity market prices and the private costs of producing power to a number that is way above private cost levels. Looking at, for example, coal and oil the range of results produced by recent studies is from 0.004 to roughly 68 US cents per kWh for coal and from 0.03 to almost 40 US cents per kWh for oil (Table 3). In comparison, the projected lifetime generation costs for the cheapest new power plants (coal and natural gas) normally range between 2.5 and 7 US cents per kWh depending on country and site (IEA/NEA 1998). The reported discrepancies in results for similar fuels raise some concerns about the validity and reliability of the conducted valuation studies. Still, it must be made clear that there is no reason to question the general notion that to some extent the numbers *should* differ due to, for instance: (a) the use of different technologies (e.g., implying separate emission factors); (b) the characteristics of the specific site under consideration (e.g., population density, income, transport distances and

Table 3: Descriptive Statistics of Previous Externality Studies

(US Cents/kWh)	*Coal*	*Oil*	*Gas*	*Nuclear*	*Hydro*	*Wind*	*Solar*	*Biomass*
Min	0.004	0.03	0.003	0.0003	0	0	0	0
Max	67.72	39.93	13.22	64.45	26.26	0.88	2.20	22.09
Difference	16930%	1331%	441%	214833%	—	—	—	—
Mean	14.01	12.32	4.61	7.12	3.36	0.31	0.84	4.95
Median	6.38	9.11	2.62	0.81	0.32	0.32	0.76	2.68
Std. Dev.	15.99	12.45	4.58	16.96	7.59	0.24	0.74	5.57
N	36	20	31	21	16	18	11	22

Source: See Table 2.

7 Sundqvist (2002) shows that these tentative conclusions remain after having accounted for methodological choice, income, and whether the entire fuel cycle (rather than only the generation stage) has been evaluated.

so on.); and (c) differences in scope (e.g., a fraction of all externalities may be included, and/or the entire fuel cycle rather than only the generation stage has been evaluated). Overall, however, the question of whether the large ranges in estimates are motivated or not, is difficult to determine, especially since there exists no objective truth with which to confront the empirical estimates.

Fifth and finally, Table 2 and Figure 8 do not display the different types of externalities covered, but a closer examination of this also reveals important disparities among studies. For example, Table 4 lists eight studies that have assessed the impacts of hydropower. It is apparent that the types and the classification of impacts differ among studies (Sundqvist 2000), e.g., some of the hydropower studies have left out the 'typical' recreational impacts.

Table 4: Impacts Monetised in Eight Hydropower Studies

Study	*Impacts*	*Study*	*Impacts*
Ottinger et al. (1991)	Forest Wildlife Recreation Fur trapping	Martins et al. in EC (1999)	Health Agriculture Crops
Pearce et al. (1992)	Health Global warming	Nilsson & Gullberg in EC (1999)	Ecological Social
Carlsen et al. (1993)	Regional economic Nature conservation Forest Recreation Fish Reindeer herding	Diakoulaki et al. in EC (1999)	Health Forest Agriculture Noise Water Biodiversity Employment
EC (1995)	Health Forest Agriculture Water supply Recreation Cultural sites Ecosystems Employment Ferry traffic Local income	ORNL & RfF (1994)	Recreation Employment

There are also important differences among the various studies with respect to the number of stages of the entire fuel cycle assessed. For instance, all the hydropower studies assess solely the construction and generation stage. For coal, on the other hand, a large part focuses on several stages of the fuel cycle (Table 5). This also raises the question of what are the relevant scope and the appropriate externality classifications to use in these types of studies. Krewitt (2002) concludes in his evaluation of the ExternE project that it has

Table 5: Fuel Cycle Stages Monetised in Eight Coal Studies

Study	*Stages*
Schuman & Cavanagh (1982)	Generation
Chernick & Caverhill (1989)	Generation
Ottinger et al. (1991)	Generation
Pearce et al. (1992)	Generation
ORNL & RfF (1994)	Extraction Transport Generation
EC (1995)	Construction Mining Fuel processing Transport Generation Decommissioning
Krewitt et al. in EC (1999)	Extraction Transport Generation
Linares et al. in EC (1999)	Construction Extraction Transport Cleaning Generation Waste disposal

provided some partial answers to this question but that many important issues remain unsolved.

To sum up, this section has provided a rough and aggregate overview of previous attempts to place value on the external costs of electricity. In section 4 we take a closer look at the studies under review and identify some conceptual issues that need to be addressed before the usefulness and the reliability of externality studies can be assessed.

4. Fundamental Questions about the Valuation of Electricity Externalities

As was noted in the introduction to this article most of the previous surveys of electricity externality studies focus on selected technical and methodological issues that need to be resolved before the valuation exercises can provide reliable estimates of energy externalities. For instance, the appropriate ways of finding reliable estimates of CO_2 and mortality impacts have been discussed intensively (e.g., Freeman 1996; Krewitt 2002). In this section, however, a number of more fundamental issues concerning non-market valuation of energy externalities are discussed. In particular we discuss the role of externality valuation in policy-making, and in particular the issue of whether the specific theoretical foundations of economic valuation methods represent a problem when these methods are used in practice.

What Constitutes an Externality?

As was noted above, the welfare economics literature provides a relatively straightforward definition of the concept of *externality* (e.g., Baumol and Oates 1988). However, in practice the choice of 'relevant' externalities tends to differ between different valuation studies. In other words, some studies seem to differ considerably in their definition of what constitutes an externality. Two important examples will help to illustrate this point.

First, there is some disagreement on whether the consumption of non-renewable natural resources, such as fossil fuels and uranium, leads to external costs. Hohmeyer (1988) adds a resource depletion charge and an external cost to public investment in R&D in his study. According to his lower estimate, these two components together account for more than 80 per cent of the external costs of the nuclear fuel cycle. The classification of natural resource depletion as an externality is, however, questionable. Hohmeyer (as well as others) relies on the concept of 'backstop-technology' in the development of external costs for depletion impacts. This concept is based on the notion that the price for a given non-renewable resource will increase over time as the resource becomes scarcer in line with the so-called Hotelling rule (Hotelling 1931), but only up to the point where a substitute (backstop) technology becomes more attractive (e.g., the use of renewable resources). However, historical data indicate that the real prices for non-renewables have, due to technological developments, material substitution and exploration, fallen over time, something which is in direct contrast to the path predicted by the Hotelling rule (e.g., Radetzki 2002). Thus, for most natural resources the empirical data suggest decreasing (rather than increasing) scarcity and that the backstop-technology is not likely to ever become economically viable. Furthermore, in discussing the taxation of non-renewable resources as a way of internalising externalities, Mäler (1997) concludes that most tax regimes will only have distorting effects on the use of non-renewable resources, and that markets will often do at least as good a job of solving resource scarcity problems if left to their own.

Second, the inclusion of employment benefits (following a power generation investment) as an external benefit also strains the definition of what an externality is. In a paper on non-environmental externalities Bohi (1993:14) concludes:

> [...] the existence of a breakdown in the local labor market is required to establish the existence of an externality, where for some reason unemployed labor will not migrate to other areas to gain employment, and will remain unemployed unless there is an increase in local job opportunities.

Consequently, for employment effects to be considered 'external' the local and regional labour market must function poorly (i.e., market failures must be present) and workers must be immobile. Other authors, however, make a strong case for treating employment impacts as external benefits (e.g., ORNL and RfF 1994). For instance, to the extent that people obtain disutility (e.g., less self-confidence) from unemployment as such, this notion is valid.

However, in such a case the employment benefits should be compared across all fuel sources, and not only for renewable energy projects as is often the case.

A related question to that of externality definitions is whether one should credit the avoided external costs from replacing existing power generation as major benefits of 'new' investments in, say, wind, solar or biomass (see, for instance, Hohmeyer 1988). These avoided costs do not per se constitute externalities. Including these 'avoided' externalities of fossil fuels, as Lee (1997) notes, also gives rise to double counting of externalities for these fuel sources (i.e., an external cost for fossil fuels and an external benefit for the renewables). In addition, some studies include subsidies to the power generation sector as a negative externality (e.g., van Horen (1996) in the case of nuclear in South Africa). However, subsidies as such do not constitute externalities.

In sum, it must be noted that some studies – of which two have just been mentioned – attempt to accomplish something more than a valuation of the externalities *per se*. If the specific aim of a study, such as Hohmeyer's (1988), is to evaluate the benefits and costs of replacing existing power sources with new ones, it will be correct to include the avoided costs from replacing existing power sources. Similarly, van Horen's study (*de facto*) represents a first step towards assessing the total (internal and external) costs of power production (by assuming that the existing subsidies to nuclear power are not motivated from an economic efficiency point of view). However, both of the above analyses constitute broader research undertakings, and they must therefore not be confused with pure externality valuation studies.

For economists the existence of externalities motivates regulatory action and the use of taxes and subsidies to improve efficiency. However, often lay people and politicians hold the view that regulations and other policy measures should be used to promote the 'good' and discourage the 'bad' in a much broader sense. This suggests that economists have to be careful in explaining what the results stemming from externality appraisals show and what they do not show. The above also raises the more philosophical question of what should be the ultimate end of policy: economic efficiency based on individual preferences or other value judgments formed in public deliberations (or any mix of these policy principles)? If the latter path is chosen, externality valuation may still provide an important input into the policy decision process but the notion of some 'total' (or 'true') cost of electricity production appear less valid. We will revert to this question below.

What is the Relevant Scope of the Analysis?

Even though all the external costs and benefits of a given power generation source have been identified, there remains the issue of choosing the appropriate level of scope of the valuation effort. At least two choices have to be made. Which of the externalities are important enough to include in the valuation, and should the study address externalities across the entire fuel cycle (including downstream and upstream impacts) or only focus on the generation stage? We have seen (in section 3) that the studies conducted in the past have made different choices in these respects, and this is likely to explain some of the disparity in the estimates presented. For instance, by using a statistical analysis Sundqvist (2002) shows that the expected externality estimates of the studies focusing on the entire fuel cycle are higher, *ceteris paribus*, than those presented in generation-only studies. Sundqvist also tested for the impact of including or excluding CO_2-estimates; the outcome of this exercise indicated that whether CO_2 was assessed or not significantly affects externality estimates. Results such as these beg the question as to what is the relevant scope of an externality investigation. Let us first look at how this question has been dealt with in previous studies.

As a part of the process of deciding what externalities to include, many of the studies reviewed in this article (e.g., Rowe et al. 1995; ORNL and RfF 1994–1998; EC 1995; 1999) begin with a comprehensive screening of the relevant economic and scientific literature to determine which pathways and end points are likely to be important in the analysis. Thus, some of the externalities are *de facto* assigned a zero value on the basis of their presumed insignificance, while others are included in the analysis since the initial screening could not rule out the possibility of significant effects. It should be clear that this approach may lead to total externality values that are significantly downward biased.[8] What is perhaps of more importance, however, is those cases where externalities are left out of the analysis because there is insufficient scientific information to establish defensible monetary damage values. The most important example is global warming and CO_2 emissions (e.g., Freeman 1996). In the first phase of the ExternE project it was noted that the environmental damage cost estimates for greenhouse gas emissions presented in the literature spanned a range of several orders of magnitude (EC 1995), and the main report concluded that: '[...] all attempts to value these impacts require important normative judgements, and therefore the potential for synthesis or consensus is remote'.

After additional research efforts within the ExternE project (EC 1999) these general conclusions largely appear to be still valid (Krewitt 2002). It is, however, generally agreed that the external costs of CO_2 emissions are substantial and may therefore constitute a large share of the total value (Freeman and Rowe 1995). The environmental economist faces a dilemma here; is it better to leave out potentially important external damages from the valuation and present biased estimates or should one make use of rough proxy estimates (e.g., mitigation costs) so as to provide (or at least approach) some kind of 'full cost' estimate? The ExternE study (EC 1999) in the end chose the latter path and recommended the use of 'minimum, 'central' and 'maximum' estimates (see below).

This raises a number of important issues, though. The choice of what externalities to include in the assessment cannot be done entirely objectively

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8 Even if the value of each externality is deemed to be low, the total value of a large number of 'insignificant' values could be substantial. In some studies up to about 100 effects are classified as negligible (Ottinger 1997).

but is largely a matter of judgment. The judgment that has to be made is essentially whether, for instance, an environmental impact under consideration is 'mature' enough to 'undergo' economic valuation. This is however not only a question of whether the scientific knowledge is more or less established; it also involves the issue of whether the public is sufficiently informed and, hence, able to form an opinion of their own about the issue at hand. Again, economic valuation is ultimately about measuring people's given preferences towards goods, but if no relevant preference structure for a particular good, such as global warming, exists, the valuation effort may become arbitrary. We have also noted above that the use of abatement cost estimates (i.e., regulatory revealed preferences) provides poor substitutes since such estimates rely on the notion that all relevant preferences have already been perfectly integrated into policy decisions.

This questions ExternE's choice to present rough estimates of the global warming impacts. Moreover, global warming also entails important ethical and justice-related issues such as the question of whether human lives lost in developing countries should count less than corresponding lives in the developed world,[9] or to what extent impacts affecting future generations should be discounted or not (e.g., Azar and Sterner 1996). In such a case, therefore, the initial challenge of policy may not lie in 'measuring' and aggregating individual preferences but in specifying the conditions for public discourse over common ways of understanding what the pertinent issues are about (see also below).[10]

The above also suggests that any notion of 'full, 'total' or 'true' cost of electricity has to be understood as at best hypothetical. The fact that externality studies leave out potentially important externalities may not be a problem *per se*. On the contrary, non-market valuation builds on specific basic assumptions, primarily about the behaviour and preferences of the public, and if these are not well articulated it may be better to refrain from monetary valuation. These concerns are likely to be particularly valid for 'goods' with which people have relatively little past experience, and which are 'complex' and involve far-reaching and unknown consequences (Vatn and Bromley 1994). Global warming as well as the risk profile of nuclear accidents and radioactive waste provide two such examples.

Finally, the choice of what parts of the fuel cycle to focus on also complicates the assessment of the 'full' cost of electricity. Where should one draw the appropriate analytical boundaries, and thus the limits of the external cost assessment? This choice is perhaps somewhat more technical than that about what externalities to include, but it is no less important. It is also

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9 This is normally an outcome of the fact that global warming impacts are valued at national or regional prices.

10 One may of course argue that even if people have less developed preferences towards global warming as such they may still be able to express their willingness to pay to avoid the *consequences* of global warming (which may be easier to comprehend than its causes). Still, this notion disregards the fact that people's preferences towards these consequences and their view on the ethical issues involved are likely to be dependent on the causes of these same effects.

complicated by the fact that the values of many upstream and downstream impacts tend to be highly site-specific and depend, for instance, on the mode of transportation and the location of the place where the fuel is extracted (Freeman 1996). One problem is how one is to choose to define the 'entire' fuel supply chain. For instance, if we include the externalities incurred by transporting fuels from the mine or well to the power station in the analysis, should we then not also include the corresponding transport-related externalities involved in bringing building materials and workers to the power plant construction site? The choice of a relevant level of analysis is not easy, and this also suggests that the notion of any 'total' or 'true' cost of power generation is flawed. Ideally this choice should be guided by the need for relevant policy information.

What are the Relevant Parameter Input Assumptions?

There are essentially two different categories of parameter input assumptions made in electricity externality studies; technical assumptions (e.g., energy efficiency, dose-response functions, emission factors), and economic assumptions (i.e., monetary values, discount rates). Previous survey work in the field has spent a lot of time on these issues (e.g., Lee 1997; Schleisner 2000). However, while past discussions have normally focused on what are the 'best' estimates (assumptions) to make, we will focus in more detail on the role of the above assumptions in explaining the wide disparity of externality estimates (Figure 8) and for providing relevant policy implications.

Table 6 shows the assumed emission levels in tons per kWh for some air pollutants in some selected externality studies. These differ significantly across studies.[11] The reported emission levels, as Lee (1997) also notes, are greater in the earlier studies (late 1980s and early 1990s) which also produced higher estimated damages than the studies from the mid 1990s. One important reason for this is that two of the early studies include global warming. The ExternE national implementation studies (EC 1999) also report global warming impacts and thus present estimates that are generally above the levels of the earlier studies. However, the monetary values incurred for the CO_2 impacts span over a wide range. On average the values range from a low of 0.55 to a high of 12.48 US cents per kWh, with discounted mid-point estimates of 1.73 (3 per cent discount rate) and 4.32 (1 per cent discount rate) US cents per kWh, thus significantly affecting the totals (see Kühn (1998) for more on this issue). Overall this shows the importance of input assumptions and hence of the input-data used in the studies.

Another example concerns the assumptions made about the monetary

11 The assumptions about emission intensities are often based on actual observations that in turn are affected by existing legislation. However, the 'level of legislation' differs considerably among countries. As a result, there is a considerable potential for error here; in the countries with strict regulations a considerable part of damages may already be internalised, while in countries with more lax regulations only a fraction of the total damages may be internalised. Thus, the comparison of studies across countries and hence among varying levels of regulations may be rather complex.

Table 6: Assumed Emission Levels for Coal: Power Generation Stage

Pollutant (tons/kWh)	*Hohmeyer (1988)*	*Ottinger et al. (1991)*	*Pearce et al. (1992)*[a]	*ORNL & RfF (1994-1998)*[b]	*EC (1995)*[c]	*EC (1999)*[d]
SO_x	8.33	9	1.32 & 15.4	1.74 & 0.81	1.21 & 0.88	1.18 & 1.36
NO_x	3.84	3.04	2.98 & 5.84	2.9 & 2.2	2.43 & 0.88	1.70 & 2.22
CO_2	excluded	1050	1200 & 1420	excluded	excluded	1015 & 900
External Cost (US cents/kWh)	2.37–6.53	3.62–8.86	2.67–14.43	0.11–0.48	0.98–2.39	4.64–32.22

Notes;
a: estimates for new and old plant
b: estimates for US South West and US South East
c: estimates for West Burton (UK) and Lauffen (Germany)
d: estimates for Spain and France

values used to value mortality impacts. Many previous studies use the value of a statistical life (VOSL) and the assumptions made concerning this value tend to differ. For example, the ExternE core study (EC 1995) uses a VOSL-value of 2.6 million USD for Europe while van Horen (1996) relies on a value range of 2.9–5.6 million USD for (the poorer country) South Africa.

In the national implementation part of the ExternE project (EC 1999) the decision was made to introduce an alternative measure on which to base the valuation of mortality impacts due to air pollution. This is the so-called Years of Life Lost (YOLL) approach, which essentially assigns a *WTP* to the risk of reducing life expectancy rather than to the risk of death (ibid.). The YOLL-values attributed to the mortality impacts are, as is evident from Figure 9, reduced by up to two orders of magnitude as compared to the values based on the VOSL-method (see also Kühn 1998). In addition, the core project (EC 1995) that relied on the VOSL-approach did not include values for chronic mortality impacts due to air pollution, something that the national implementation studies do.

Overall, the above implies very different messages to the policy makers about mortality impacts depending on method used and the scope of the investigation. Schleisner (2000), who compares the ExternE core project (EC 1995) and the Rowe et al. (1995) study, supports the view that the assumptions underlying the valuation of human health and mortality impacts as well as dose-response functions are major drivers of external cost estimates. Clearly this also provides a major explanation as to why the total externality estimates from different studies often differ much (even for the same fuel source).

So far in this sub-section we have noted that the input assumptions made in externality assessments play a significant role in affecting the overall external damage estimates, and thus in guiding policy. To some extent this is of course how it should be; different sites and different technologies (depending, for instance, on vintage) incur various emission impacts and

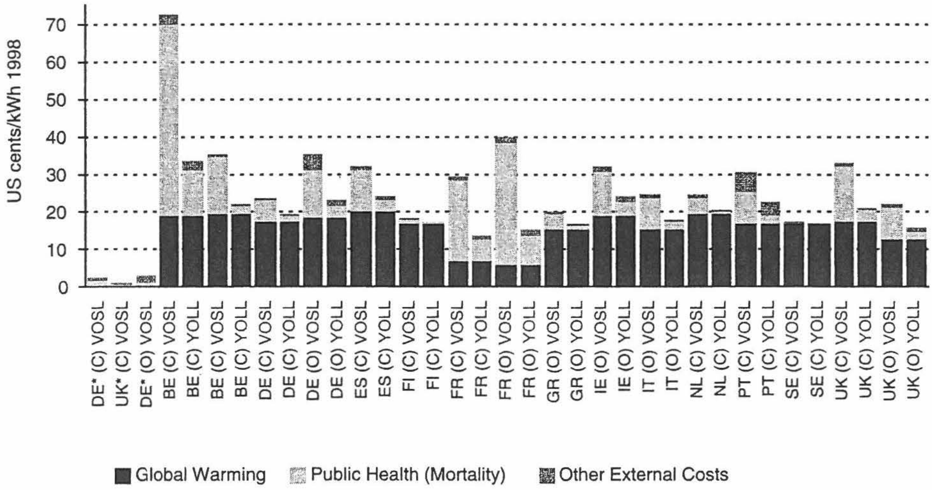

Figure 9: External Cost Estimates in the ExternE Core and National Implementation Projects: Coal (C) and Oil (O) Fuel Cycle

Sources: EC (1995; 1999).

hence damages.[12] Still, the large sensitivity in results due to parameter input assumptions also creates problems for policy makers. *First*, policy makers often wish to gain some notion as to which are the most important environmental damages for each power generation source. This enables them to target policy measures, such as subsidies, R&D support, and regulations, towards the most important impacts. However, previous research efforts have only provided some limited guidance on this particular point. As has also been noted by OTA (1994), in many studies a single category of damages seems to dominate the total external cost estimates. For example, in Hohmeyer's (1988) study and the initial ExternE coal study (EC 1995) human-health related impacts dominate the aggregate damage (75 per cent and 76–95 per cent of the total, respectively), but for van Horen (1996) CO_2 impacts constitute the majority of estimated damages from the coal fuel cycle (80–90 percent). *Second*, in deregulated electricity markets one of the most important uses of external cost estimates is as inputs in the development of environmental taxes and tradable permit schemes (Freeman 1996). For this reason, though, policy makers would need relatively 'safe bets' about the general impacts involved, but so far previous studies have provided only

12 This puts in doubt those studies that rely heavily on so-called benefit transfers, and thus draw on the original research of others without making necessary modifications. For instance, in valuing the external damages caused by air pollutants from coal generation in South Africa van Horen (1996) employs the dose-response relationships developed within the New York State Environmental Externality Costing Study (Rowe et al., 1995), but is unable to fully adjust the model to South African conditions.

wide ranges of estimates. Again, the impact of global warming is a good illustration of this.

This discussion raises a fundamental issue in non-market valuation that is rarely touched upon in the electricity externality debate. Most environmental economists would agree that environmental valuation requires that a relevant 'project' has been defined, and that involves the choice between two or more relevant alternatives (Brännlund and Kriström 1998). In the case of electricity externalities these 'projects' are normally the investments in different power plants (few companies or governments undertake investments in entire fuel cycles). However, in externality studies these investment projects are often hypothetical, i.e., they do not represent an existing (real) situation, and valuation estimates are therefore transferred from other studies. The problem with this is that according to the literature on non-market valuation (and indeed that on market valuation as well) economic values are context dependent and project-specific (e.g., Garrod and Willis 1999). In other words, it may not make much sense to talk about a universal *WTP* for avoiding one ton of SO_2 being emitted, even though that is just what many policy makers would like to know about as they often prefer more or less harmonised standards and taxes across different regions.

Does the Choice of Methodological Approach Matter?

As has been noted above, there is no reason to question the fact that external cost estimates vary considerably across different studies if there exist rational reasons for this discrepancy. One important example, though, where one would be concerned about the reported disparity is if the choice of methodology has a significant impact on the results. In section 2 we emphasised that there is a vast number of different valuation methods (Figure 6), but in this section we will focus solely on the broader approaches to externality valuation: abatement costs, top-down damage costs, and bottom-up damage costs. Let us first consider what lessons can be drawn from previous studies about the difference between abatement costs and (bottom-up) damage costs

Few studies analyse this difference, but Table 7 presents some empirical evidence of environmental cost estimates conducted by the California Energy Commission for various air pollutants and districts in California (CEC 1993). This study applies both abatement cost and damage cost methods on the same cases making direct comparison possible. Even if the estimates vary as a function of district-specific conditions, it is clear that (except for PM in some of the districts) the abatement cost method tends to generate significantly higher environmental cost estimates than those developed using the damage cost approach.

Joskow (1992) provides one important explanation for this phenomenon. Abatement costs will (theoretically) be representative of damage costs if and only if they are derived from the *least cost* strategy, but normally most studies employ the most commonly used (or mandated) abatement technology when assessing pollution abatement costs. For example, in deriving the

Table 7: Value of Air Emission Reductions in California

District: (USD/Pound)	*South Coast*		*Ventura County*		*Bay Area*		*San Diego*	
Method:	DC	*AC*	DC	*AC*	DC	*AC*	DC	*AC*
SO_x	4.88	*13.02*	0.99	*4.08*	2.28	*5.85*	1.76	*2.37*
NO_x	9.52	*17.4*	1.08	*10.85*	4.83	*6.84*	3.66	*12.03*
CO	0.00	*6.12*	0.00	*I*	0.00	*1.45*	0.00	*0.72*
ROG	4.55	*12.43*	0.18	*13.88*	0.07	*6.71*	0.07	*11.51*
PM	31.32	*3.75*	16.05	*1.18*	15.78	*1.71*	9.35	*0.66*

District: (USD/Pound)	*San Joaquin Valley*		*Sacramento Valley*		*North Coast*	
Method:	DC	*AC*	DC	*AC*	DC	*AC*
SO_x	0.99	*11.71*	0.99	*5.39*	0.99	*1.98*
NO_x	4.26	*5.98*	4.01	*6.01*	0.53	*3.96*
CO	0.00	*2.10*	0.00	*3.30*	0.00	*I*
ROG	2.45	*5.98*	2.72	*6.01*	0.31	*2.31*
PM	2.47	*3.42*	1.44	*1.85*	0.37	*0.59*

Where: DC: Damage Cost, AC: Abatement Cost, I: Internalized, SO_x: Sulfur Oxide, NO_x: Nitrogen Oxide, CO: Carbon Monoxide, ROG: Reactive Organic Gases, and PM: Particulate Matter.

Source: CEC (1993).

external damages from SO_2 emissions in the USA Bernow et al. (1991) make use of the costs for installing scrubbing equipment. At the time when this study was conducted this indicated a cost per ton of SO_2 of about USD 1500–2000, and this estimate corresponded fairly well to the projected prices of future SO_2 emission allowances in the tradable permit system soon to be implemented in the USA. However, the actual prices of SO_2 allowances for most of the period 1992–1997 varied between USD 100 and USD 200 per ton (Schmalensee et al. 1998), indicating that the compliance costs have been much lower than originally expected. As noted by Smith et al. (1998:23):

> [E]stimates in the range of [USD] 1000 per ton or more have always been for the marginal costs, i.e., costs associated with the most difficult-to-control sources. That narrow focus overlooks the flexibility made possible through emissions trading.

In practice, many of the US coal-fired plants chose to rely on low-sulphur coal in their production rather than to invest in scrubbers. Technical progress in the abatement technology field also contributed to lower sulphur prices. Thus, the failure of previous studies to identify the least cost abatement technologies or strategies tends to lead to an exaggeration of the damage costs involved. Even more importantly, with technical change and increased flexibility the abatement cost approach indicates a decreasing valuation of the environment even though, for all practical reasons, the opposite is true.

By using the results from over forty electricity externality studies across eight fuel sources Sundqvist (2002) provides some econometric evidence in support of the conclusion that methodological choice matters for the results. He reports that the probability of obtaining a 'low' externality value is, *ceteris paribus*, generally lower when the abatement cost approach or the top-down damage cost approach are used while the opposite is true for the bottom-up damage cost approach. The reason why the top-down approach also tends to produce relatively high external damage is that there may arise practical problems in attributing the 'exact' damage to each individual source, which may force researchers to rationalise and use standardised rules for the attribution-process. These rules may fail to ascribe the aggregate damage to each and every individual source, especially smaller sources, thus producing estimates for specific power plants that are positively biased since these plants, normally, are easily identifiable as well as significant sources of pollution.

The fundamental question raised by these results is whether the three broad approaches to externality valuation are at all comparable. So far we have suggested that the differences reported may be primarily due to purely practical or technical reasons such as identifying the least cost control strategy and/or addressing all relevant end points. If these issues can be resolved, thus, the methods would (at least theoretically) generate similar results. In the remainder of this article, however, we suggest that this is not necessarily the case. The reason for this has to do with the fact that people tend to have more than one preference ordering, something which is in contrast to the standard behavioural assumptions made in the economics literature (e.g., Nyborg 2000). Before proceeding, however, we approach the question of whether establishing a market for 'green' electricity can make externality studies redundant for policy purposes.

Can Consumer Demand for Green Electricity Make Environmental Valuation Redundant?

The assessment of environmental externalities in the power sector is motivated by perceived market failures, i.e., the socially optimal level of 'green' power is arguably higher than the level chosen by private investors. However, if consumers are willing to pay a premium for 'green power' and act accordingly in the electricity market, such 'green' preferences would, some proponents argue (e.g., US-DOE 1999; Global Green USA 2002; SNF 2002; Vattenfall 2002), induce the industry to approach the optimal level of environmentally benign power sources. Ideally externality assessments could become redundant and there would be no need for additional regulatory measures to correct for market failures.

However, there are several problems with this approach (Brennan 2001). *First*, a higher demand for 'green' power may be interpreted as a change in preferences in favour of 'green' power sources, but since economists evaluate policy efficiency based on exogenously given preferences the very idea of preference change questions the foundations of economic policy analysis. In practice, of course, it is difficult to distinguish between activities that change people's preferences and those that change behaviour by altering the available information. Brennan (2001:7) points out that:

> In the case of green power, the blurry distinction would be between activities that increase one's underlying preference for environmental protection and those that give consumers information to act on environmental preferences they already have.

Second, even if we would be able to distinguish between changes in behaviour due to (a) preference change and (b) new information, none of these alternatives tend to support the conclusion that 'green' demand can replace externality assessments. If consumers increase their demand for 'green' power due to changing preferences the electricity production will still involve a market failure; the increase in 'green' power demand simply implies that the optimal level of 'green power' has increased but there will still exist a difference between the optimal and the actual level of 'green' power capacity. Thus, the need for standard regulatory measures and thus for externality assessments remains. Even the very idea of green preferences as policy substitutes creates a problem as it begs the question of how one would define the no-policy alternative (i.e., the one corresponding to the free market solution in the welfare economics literature). Brennan (2001:16–17) concludes:

> The lesson from the above is that adding preference change to the policymakers' toolkit creates a huge range of ambiguities for the economist's appraisal of policy effectiveness. This should not be surprising; to expect otherwise would be to expect that preferences could be both policy instrument and policy criterion [see section 2], i.e., both the means and the end of policy.

If demand for 'green' power in the past has been suppressed due to information failures we are essentially dealing with two types of market failures: an environmental externality and incomplete information. If information becomes (in any sense) complete, principally the environmental externality problem would still be there even though the total environmental impacts may be less severe. Also in this case 'green' power demand would not be able to replace externality assessments.

Still, 'green' electricity demand could make sense from a regulatory point of view if the policy goals are defined, not by the economic efficiency criterion outlined above, but instead are based on deliberations about the public good in which preferences are formed rather than considered as given (or, less favourably, by politicians' self-interests). However, even though one accepts the view that green preferences can serve as a substitute for taxes, regulations and so on, there would still be a need for environmental valuation exercises. For instance, companies who wish to market their electricity as 'green' need to understand how people perceive and value different aspects of their power generation portfolio. Valuation studies would provide important implications in that they indicate the willingness to pay for 'green' electricity in general, and the extent to which households and/or other customers are willing to pay more for certain characteristics of the 'green' power sources than for others.[13]

Moreover, even though one accepts the notion that 'green' preferences can serve as a legitimate policy instrument one may doubt whether they are sufficiently developed to make a real difference. According to the economics literature public goods, i.e., goods characterised by non-rivalry and non-excludability in consumption, are normally underprovided in the market place (e.g., Varian 1992). Since many environmental goods are essentially public in this sense, 'green' power markets may not promote enough of the environmental benefits embodied in the cleaner power technologies. This conclusion, however, builds on the assumption that people's preferences are entirely based upon utility maximising behaviour and that they are unlikely to express altruistic concerns. In the next section we discuss the possibility that people may have multiple preference orderings (and hence additional ethical codes), and the consequences for the social choice between power generation sources.

Preferences Matter but which Preferences?

Critics of environmental valuation based on economic methods normally stress that the methods used rely on overly restrictive assumptions, something which implies that they often produce poor descriptions of the environmental values people actually hold as well as of the process of preference formation (e.g., Spash 1997; 2000). Specifically, externality valuations rely on the notion that individuals aim at maximising personal utility and that they possess well articulated, ethically unchallengable, and exogenous preferences for any environmental goods. However, environmental values often have a broad ethical content, and since ethics are a matter for argument environmental valuation should, it is argued, be endogenous to the political process and rely on social agreements (e.g., Jacobs 1997). In other words, the initial challenge of environmental policy may not lie in 'discovering' private preferences but in specifying the conditions for public discourse over what is worth valuing and for what reason.

Sagoff (1988) claims that individuals essentially have at least two different preference orderings. Their *private* preferences reflect only their own well-being as consumers of private goods, while the *public* preferences reflect moral values about what persons, as citizens, believe is right and just for society as a whole. In their roles as citizens, people may express a rights-based (or a deontological) belief system, which denies the principle of utilitarianism (and tradeoffs) and instead recognises the priority of the right over the good (Spash 1997). Such a moral position is likely to be particularly prevalent in the case of environmental goods. The environment is often the subject of ethical concern, and it involves many cross-cutting dimensions

13 Such studies would imply a greater reliance on choice experiment applications, since they encourage people to consider different attributes of a good (e.g., green hydropower) rather than changes in the good as a whole (e.g., Hanley et al. 1998).

which cannot be causally separated. In this way the 'market analogy' and the 'commodity fiction' of environmental valuation may break down (Vatn 2000).

There exists empirical evidence that people express public, rather than private, preferences when considering complex environmental issues and when confronted with *WTP* questions in contingent valuation surveys (e.g., Russell et al. 2001). It is also the case that the consumer-citizen distinction provides one explanation to some puzzling phenomena frequently observed in CVM studies. For example, as noted above, a large number of studies have found very large discrepancies between *WTP* and *WTA* measures for environmental goods, the latter often being several times larger than the former. This may be explained by the notion that in the *WTA* case, if the environmental good is morally considerable, then the acceptable level of compensation will be extremely high or even undefined.

Söderholm and Sundqvist (2000a; 2000b) show that the distinction between private and public preferences is highly relevant when considering the environmental costs of power generation. Most importantly, many power generation externalities are either 'new' (e.g., the risk profiles of nuclear power) or 'complex' (e.g., ecosystem changes due to hydropower developments). In addition, most power generation fuel cycles involve significant impacts on the health and deaths of humans. These impacts raise a moral dilemma; to what extent should we treat humans as means to an end (utility) or as ends in themselves? Such a question should ideally be resolved within the realms of public discourse. The social choice problem with respect to many energy and environmental issues is thus, first of all, about advancing common ways of understanding what the pertinent issues are about. This implies that environmental research in the social science field must increasingly address the instruments and content of political and moral debate and not simply the technicalities of established valuation methods. In other words, the process may count every bit as much as the outcome (Sagoff 1998).

The above discussion also adds a new perspective to the observed differences in reported valuation estimates between the abatement cost approach and the damage cost approach. The two methods involve different ethical bases. For example, the damage estimates developed within the ExternE-project are considered *ex ante*, i.e., the damages themselves determine whether one power source is 'better' than another, while the estimates derived using the abatement cost approach are *ex post*, i.e., the price is an outcome of a political process and does not play a direct role in the decision. The damages developed in 'advance' (as in the ExternE-project) may therefore not be directly comparable to 'implicit' estimates that are based on the cost of abating as revealed by decision makers since they reflect different reasoning processes. Policy makers may in their formulation of regulations very well rely on other ethical foundations than economic welfare theory.

5. A Summary of the Main Findings

This article has analysed past research efforts on valuing the externalities, and primarily the environmental costs, arising from power generation. In doing this we have raised a set of conceptual and, to some extent, unresolved issues, but in general we have focused especially on two interrelated questions: (a) the wide disparity in external costs reported in previous studies, and the extent to which this represents a problem; and (b) the usefulness of previous valuation efforts for policy purposes.

A number of plausible reasons for the reported wide disparity of estimates for specific fuels were identified : e.g., differences in scope, the use of varying technical and economic assumptions, and the choice of methodology (i.e., the abatement cost versus the damage cost approach). The use of different scopes and input parameter assumptions (and thus distinctly different reported external cost estimates) is often justified. For instance, if people are unlikely to possess developed preference structures for certain environmental impacts, either because they have little past experience or the impact involves far-reaching and unknown consequences, it may be better to refrain from monetary valuation. Moreover, since different sites and different technologies incur various damages (even for the same fuel) the use of different parameter assumptions regarding, say, emission intensities and economic values is motivated. Economic valuations are, and should be, context-dependent and project-specific. This suggests that any notion of some 'total' cost of power generation appears invalid.

However, the above may create problems for the use of externality estimates for policy purposes. People in general and indeed policy makers often 'have the expectation that external costs are as simple to understand as price tags in a store,' (Krewitt 2002:847), but in practice the empirical estimates of external costs have provided few general guidelines on how to allocate public funds (i.e., subsidies, R&D support) between different power sources. In addition, due to the context-dependent and site-specific characteristics of economic valuation estimates it may be impossible (or at least very impractical) to implement uniform taxes based on external cost estimates. Nevertheless, this does not imply that the valuation efforts have been in vain. Previous studies have taught us a lot about the environmental impacts of power generation (in particular health effects), and even if much of this knowledge cannot be transferred directly into a tax or a regulation it should be able to impact upon the focus of the political debate and ultimately on policy decisions.

Finally, the usefulness of previous economic valuation efforts for policy purposes is also complicated by the fact that according to the welfare economics literature, valuation builds on: (a) relatively restrictive behavioural assumptions: and (b) the idea that the ethical principle guiding social choice is economic efficiency. However, if these assumptions are relaxed it may have profound consequences for the use of environmental valuation studies. Since people are likely to express public rather than private (i.e., utility maximising) preferences towards some external impacts, the social choice between different power sources must increasingly be made within the realms of public discourse where additional ethical principles may play a role. This also implies that there may exist a fundamental ethical difference between the abatement cost approach, in which externality estimates are revealed from the results of the political decision process, and the damage cost approach in which the same estimates are drawn directly from people's expressed preferences (as indicated in, for instance, contingent valuation studies).

In addition, the view that economic efficiency is the ultimate goal of policy is not likely to be shared by all lay people and politicians. This means that, in contrast to many economists, they are likely to be more indulged to promote: (a) a much broader definition of externalities than that available in the literature; and (b) the use of green power markets as a substitute for external cost assessment and implementation. Overall, this suggests that in addition to further methodological work, there is a need to direct future research efforts also to the apparent incompatibility between the intended use of external cost assessments, its theoretical foundations and its practical use in shaping policy decisions.

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