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METHODS

Pricing environmental externalities in the power sector: ethical limits and implications for social choice

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Abstract

During the last decade, a series of valuation studies have made attempts at estimating the external environmental costs of various power generation sources. The purposes of this paper are: (a) to explore some of the ethical limits of the economic valuation of environmental impacts; and (b) to analyze what the implications are of these limits for the social choice between different electric power sources. Environmental valuation based on welfare economic theory builds on restrictive behavioral foundations and can only partly model moral values, although such values are an essential part of people's preference towards the environment. In addition, public preferences are seldom exogenously given as is commonly assumed in economic theory, but are instead formed in public discourse. For this reason, the range of electricity externalities where economic valuation (and thus cost–benefit analysis) should be applied is likely to be narrower than often assumed. After analyzing the scope, methodology and the results of the so-called ExternE project, the paper concludes that many power generation externalities are either inherently 'new' or inherently 'complex'. In these cases, the initial challenge lies not in 'discovering' private preferences, but in specifying the conditions for public discourse over common ways of understanding what the pertinent issues are about. This implies that research on the environmental externalities of power generation must, in addition to refining the theory and the applications of existing non-market valuation techniques, also address the instruments and content of political and moral debate.

Keywords: Power generation; Ethics; Neoclassical welfare economic theory; Environmental valuation; Externalities

1. Introduction

One of the key elements of energy and environmental policies in the western world is to 'get prices right' and to ensure that environmental externalities are accounted for in market mechanisms. Policy makers and economists have particularly targeted the environmental damages arising from power generation. The reasons for focusing especially on the power-generating sector are two-fold. First, power generation generally provides much more flexibility in terms of fuel choices than is the case for other energy sectors (e.g. transport) and the various technologies have significantly different environmental impacts. Second, power

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plants are concentrated in relatively few and thus easily identifiable facilities.

A series of valuation studies have made attempts at estimating the environmental costs of various power-generating technologies. Most of these studies were commissioned by governmental authorities, such as the European Commission, the US Department of Energy and the UK Department of Trade and Industry (Ottinger et al., 1990; Pearce et al., 1992; Rowe et al., 1995; European Commission, 1995a, 1999). Stirling (1998) (p. 268) concludes in his review and methodological critique of some of the most important external cost studies that:

> [. . .], there is little doubt that neoclassical environmental valuation techniques are the approach to environmental appraisal currently preferred by the official bodies responsible for the formulation, implementation, and international coordination of environmental regulation in the electricity supply sector.

In other words, the theoretical support for externality valuation exercises is drawn from the neoclassical welfare economics literature. Within this strand of research, there are a number of valuation methods in use (e.g. abatement cost, contingent valuation, hedonic pricing, etc.), but ultimately they all aim at discovering people's preferences expressed as willingness to pay (WTP) for environmental goods and services (see Sections 2 and 3). The valuation and internalization of externalities is generally deemed necessary for assisting market processes and for making efficient social choices.[1] The implications for energy policy of these external cost assessments are thus essential. For example, in order to improve efficiency in the selection of new power generation sources damage estimates can be used to determine 'adders' to the private production costs (Eyre, 1997). In addition, external cost estimates can be used to evaluate existing pollution taxes and/or tradable permit systems, or help in designing new ones. Taxes and subsidies that reflect the external costs or benefits will then ensure that profit-maximizing firms select the mix of goods and production technologies that best satisfy environmental and economic goals.

However, a number of researchers in the social science field have questioned the use of non-market valuation techniques as the basis for integrating public input into the environmental policy process (e.g. Sagoff, 1988; Spash, 1997). It is argued that these methods rely on overly restrictive assumptions and ethical principles, implying that they often produce poor descriptions of the environmental values people hold and therefore serve as inadequate inputs to policy decisions. So far, though, the validity of these concerns in the empirical context of power generation externalities is only poorly understood (Stirling, 1997).

The purposes of this paper are thus to: (a) explore some of the ethical limits of environmental valuation methods within the welfare economics paradigm; and (b) discuss what the implications of these limits are for the social choice between power-generation technologies. The main thesis of the paper is that the scope of electricity externalities where environmental valuation can be applied from an ethical point of view is probably narrower than commonly assumed. Specifically, many environmental impacts in the power generation sector involve moral concerns for which private preferences are not always readily available, but rather must be formed in public discourse. For this reason, economic valuation provides an insufficient (but not necessarily unnecessary or illegitimate) basis for social choice. Also, since various power sources give rise to different types of externalities—some likely to be less amenable to social cost pricing than others—the choice between different technologies becomes

[1] According to the Coase (1960) theorem, bargaining between the polluter and the affected agent(s) can, under certain circumstances (such as low transaction costs), internalize externalities and achieve an efficient market outcome. However, in most cases, due to the large number of parties involved, such bargaining will be too complex and expensive and government intervention is therefore called for.

more complex than is implied by the welfare economics literature.

Before proceeding it is important to note that one of the most important ethical principles in welfare economics is that 'only' human (subjective) preferences should count; all values in this case are thus anthropocentric in the sense that they lack existence apart from the human valuer. This is the approach taken in this paper. Thus, the possible existence of 'strong' intrinsic values (e.g. Rolston, 1982), implying that the environment has an 'objective' value that is independent of human existence, is brought up neither in economic theory nor in this paper.[2] Our main argument, however, is that in contrast to welfare economics, which assumes a single preference ordering for each individual, there are strong reasons to believe that people possess two or more preference orderings, using different ones in different instances. This implies that the usefulness of economics in making rational choices over limited resources ought to be complemented by other forms of social agreements about what should be the important criteria in energy and environmental policy.

The paper proceeds as follows. In Section 2, we briefly review the methods used to assess the external costs of electricity generation and present some of the results obtained in previous studies. Section 3 discusses the ethical foundations and the limits of environmental valuation techniques as well as alternative philosophical approaches to human preferences and social choice. Section 4 analyzes these ethical limitations in the empirical context of the power generation externalities examined in the European Commission's so-called ExternE project. Finally, Section 5 provides some concluding comments and remarks.

2. The valuation of power generation externalities: methods and results

An externality is an unpriced benefit or cost directly bestowed or imposed upon one agent by the actions of another agent. Externalities cause market failures in the sense that there will exist a difference between the private and the social (private plus external) costs and benefits of an action and the free market's allocation of resources will, as a result, be non-optimal from society's point of view (Varian, 1992). Most electricity externality studies assess the negative externalities (external costs), most importantly the environmental damages, for selected power generation sources. In these cases, the private costs of power production is thus deemed to be lower than the social costs and electricity markets will tend to clear at a price level below the marginal social cost. The social choice between different power generation technologies will be inefficient and biased towards energy sources with low private production costs, but not necessarily low social costs.

Even though externalities are not reflected in market transactions, they do have a direct impact on people's welfare and thus on economic value. The economic valuation of externalities and thus of many environmental impacts, builds on the assumption that people seek to satisfy their preferences, i.e. maximize utility or welfare. The change in the level of individual welfare resulting from a given environmental change is typically measured as the amount of income necessary to maintain a constant level of utility before, and after, the change. In this way, one can elicit welfare changes in monetary terms through willingness-to-pay (or willingness-to-accept) measures (see also Section 3). Externality valuation is thus ultimately concerned with applying different empirical methods to identify these measures. There are two broad methodological approaches employed in

[2] However, we still consider what may be referred to as 'weak' intrinsic values, in the sense that they are non-instrumental (rather than objective) and refer to a situation in which humans consider that something has a value in itself irrespective of whether it has value in attaining something else of value (i.e. they are non-instrumental values). See Stenmark (2002) for a discussion of the distinction between 'weak' and 'strong' intrinsic values.

practice to assess the value of electricity externalities: (a) the abatement cost approach and (b) the damage cost approach.[3]

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 having considered the willingness of the public to pay for alleviation of the damage and the relevant abatement costs in setting the standard,[4] thus providing a revealed preference damage estimate not necessarily less reliable than the more explicit valuation methods (see below). An example of a study that utilizes the abatement cost methodology is Bernow and Marron (1990).

The damage cost approach, on the other hand, aims at providing an explicit (rather than an implicit) measure of the economic damages arising from a negative externality. Damage costing can be either *top-down* or *bottom-up*. Top-down approaches make use of highly aggregated data to estimate the external costs of, say, particular pollutants. Researchers adopting the top-down approach normally start at the national or the regional level, using estimates of total quantities of a specific pollutant. These physical damages are attributed to power plants and converted to damage costs using available monetary estimates (e.g. US$ per SO_2 emitted) on the damages arising from the pollutants under study (e.g. Hohmeyer, 1988). In the bottom-up approach, damages from a single source are typically traced, quantified and monetized through damage functions/impact pathways (e.g. European Commission, 1995a). This approach makes use of technology-specific data, combined with dispersion models, information on receptors and dose–response functions to physically quantify the impacts of specific externalities. These physical impacts then need to be converted to damage costs either by using available information or through original valuation studies.

There exist several ways of monetizing these externalities. The first two approaches discussed above—abatement cost and top-down damage cost—directly provide a monetary estimate of the damages associated with the externalities. However, in the third approach—bottom-up damage cost—one needs to translate the identified and physically quantified impacts into monetary terms. Generally, whenever market prices can be used as a basis for valuation, they are used. However, since externalities by definition are external to markets, impacts from externalities are not reflected in market prices. Consequently, any attempt to monetize an externality when making use of the bottom-up damage cost approach need to rely on non-market valuation methods. These methods can in turn be subdivided into (a) direct methods and (b) indirect methods.[5]

The direct methods attempt to create a hypothetical market for the environmental good. These methods are *direct* in the sense that they are based on direct questions to households about willingness to pay. The direct methods possess the advantage that they can assess total economic values, i.e. the use as well as the non-use values (i.e. existence values) associated with the good. Well-known techniques sorting under this approach include contingent valuation and choice experiments. The *indirect* methods take their basis in the actual (rather than the hypothetical) behavior of individuals. Either the welfare effects in terms of willingness to pay show up as changes in costs or revenues in observable markets or in markets closely related to the resource that are affected by the externality. The damage is thus indirectly valued using an existing relation between the externality and some good that is traded in a market. Examples of indirect methods are hedonic pricing and travel costs.

[3] See Sundqvist and Söderholm (2002) for a critical survey of a large number of economic studies focusing on the valuation of environmental externalities in the power generation sector.

[4] Specifically, the public decision makers are assumed to choose the level of abatement at which the marginal damage curve and the marginal abatement cost curve intersects.

[5] 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).

In recent years, policy makers and researchers have given increasing attention to the assessment of external costs in the electricity sector. Several major studies have addressed the issue and examples included in the ExternE-project in Europe (European Commission, 1995a) and in the US, the New York State Environmental Externality Cost Study (Rowe et al., 1995). As noted above, welfare economic theory directs us on how to value externalities and previous electricity externality studies have relied heavily on the methods outlined above. According to the welfare economic theory, the choice of method should not affect results of the externality assessments significantly, i.e. it should not matter for the outcome whether people's willingness to pay has been 'filtered' through the political process or if it has been elicited directly in, for instance, contingent valuation surveys. Still, this presumption builds on the rather strong assumption that politicians make optimal decisions, i.e. they know the true (marginal) abatement and (marginal) damage costs and they aim at maximizing social welfare. In addition, as noted by Joskow (1992), abatement costs will only be representative of damage cost if they are derived from the pollution control strategy that gives the least cost of control.

For the studies that have been completed, the externality estimates produced for each electricity source range from very high effects to more or less insignificant effects. Fig. 1 displays the external cost estimates from 63 different studies carried out during the 1980s and 1990s. For example, looking at coal, the range of external cost estimates is from 0.03 to <1000 US cents per kWh. Similar ambiguities exist for the other electricity sources.

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 etc.); and (c) differences in scope (e.g. only a fraction of all externalities may be included, the entire fuel cycle rather than only the generation stage has been evaluated etc.). Still, by employing statistical analysis and 132 observations of external cost estimates for a set of different fuels, Sundqvist (2002) shows that one additional and more troubling reason for this disparity is also the choice of externality assessment approach. Most notably, the probability of obtaining a low externality cost value is, *ceteris paribus*, lower when the abatement cost or top-down damage cost approaches are used while the opposite is true for the bottom-up damage cost approach. One reason for the difference in results between the abatement cost approach and bottom-up damage costs is that many analysts tend to base their calculations on existing regulations (rather than the least-cost regulation) when estimating the abatement cost (e.g. Joskow, 1992).[6] However, the analysis in this paper also adds a new perspective to the observed differences in reported externality estimates between the abatement cost approach and the damage cost approach, i.e. between implicit and explicit valuation. Policy makers are in their formulation of regulations likely to base their decisions also on additional ethical foundations and the implicit values reported in abatement cost studies may thus reflect a different reasoning process than that outlined in the welfare economics literature.

Fig. 1 also displays that the ranges intertwine across fuels 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.

[6] 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 rationalize and use standardized 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 larger power plants that are positively biased since these latter plants, normally, are easily identifiable as well as significant sources of pollution.

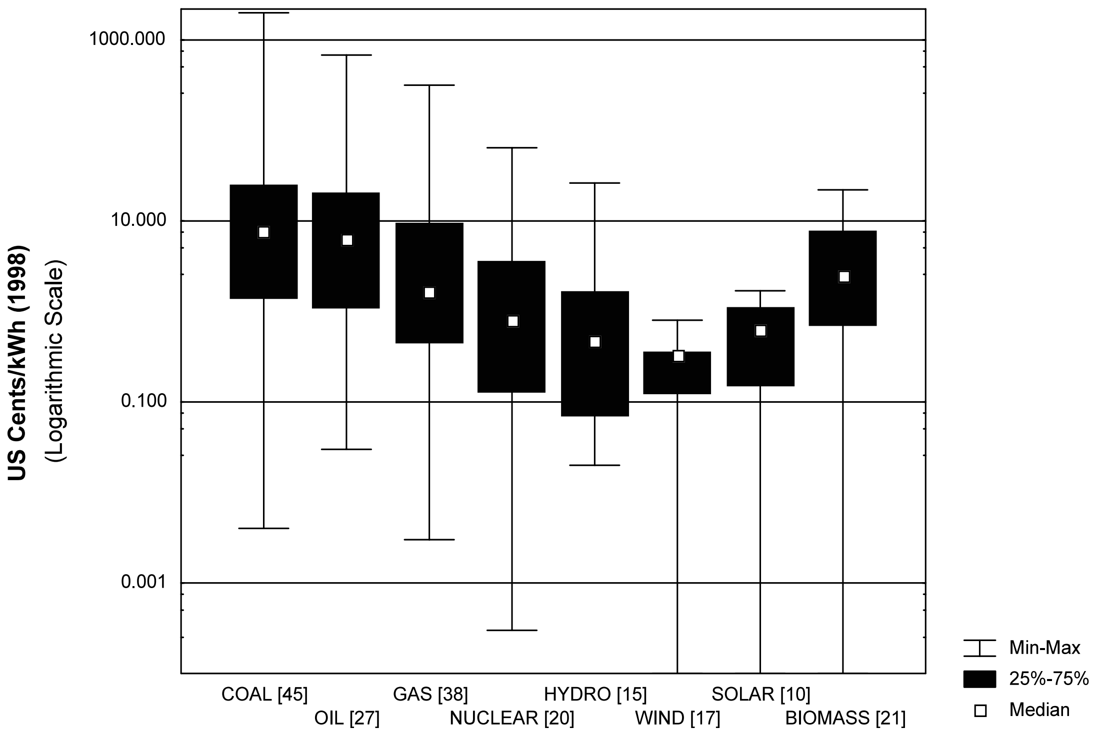

Fig. 1. Range of external cost estimates in power generation. Sources: Sundqvist (2002) and Sundqvist and Söderholm (2002).

According to Stirling (1997) (p. 531), "[t]his ambiguity in the comparison of different options is a serious defect in technique which aspires to present a robust and systematic representation of environmental performance." He argues that one of the most important defects of these studies is that they fail to address the multi-dimensional nature of power generation externalities. The different dimensions relate to, for example, the distribution of effects in terms of space, time and people, the particular forms they take (e.g. in terms of severity, reversibility etc.) and the degree of autonomy of those affected (Ibid.). Thus, according to Stirling, most of the existing valuation studies are still 'immature' and very preliminary; more realism in the treatment of the multi-dimensional nature of the external effects is therefore needed.

While previous critics, such as Stirling, address many of the practical and the methodological problems associated with assessing the externalities arising from power generation, the analysis in this paper is of a more fundamental nature. We argue that the behavioral and ethical foundations of environmental valuation, as applied to the valuation of external effects, are likely to be too restrictive for serving as the sole basis for social choice.

3. Ethical limits of welfare economics and the implications for social choice

Since the basic thesis of this paper is that the economic valuation of environmental externalities relies on specific behavioral assumptions and ethical foundations, it is useful to briefly review these before discussing alternative ethical bases for social choice and their consequences.

In the welfare economics discipline, human beings are treated as autonomous individuals who seek to satisfy their private preferences, which are complete, ethically unchallengeable (i.e. subjective) and exogenously determined. This implies that individuals have given preferences ('indifference maps') for public goods and are willing to consider tradeoffs in relation to the quantity or quality of these goods (Pearman et al., 1999). The objective of the analysis is to elicit from each individual his/her personal valuation of given environmental 'goods', measured in willingness to pay (WTP) terms. For example, within this theoretical framework each individual i's welfare is often expressed as:

$$U_i = (X,\ Z), \tag{1}$$

where U is the utility of individual i, X is a vector of the quantity private goods and Z represents the quantity of the public environmental good (e.g. air quality). The maximum WTP of individual i for increased provision of the public good is given by the solution to:

$$U_i(X^0,\ Z^0) = U_i(X^0 - \mathrm{WTP},\ Z^1), \tag{2}$$

which is equivalent to the compensating variation associated with the move from Z^0 to Z^1 at the initial level of private-good consumption level, X^0. Thus, if individuals are utility maximizers, welfare may be interpreted as units of measure of the maximum WTP for a given outcome (or reversing the property rights aspect, as a measure of the compensation an individual would require giving up some existing good, i.e. the minimum willingness to accept, WTA).[7]

Generally the welfare economics literature suggests that these welfare measures should be aggregated into the overall preference (utility) of society. The policy that maximizes total preference satisfaction needs to be chosen. The fundamental philosophical positions guiding social choice are thus that the net utility (benefits over costs) from the *consequences* of an action determines whether that action is right or wrong and a sense of society as the sum of the preferences (utilities) of its individual members. It should be noted, however, that this choice of ethical principle for *social* choices does not follow logically from the fact that utility maximization is assumed to constitute the behavioral foundation for *individual* choices. However, in practice they are likely to be closely related. The use of WTP as a welfare measure builds entirely on the assumption of utility maximizing behavior and there would probably be few reasons to estimate WTP as such if these estimates are not intended to form part of, say, social cost–benefit analyses that in turn are important (but not necessarily the only) input into the political decision process. Thus, as many philosophers point out, the development of societal ethical guidelines is largely an empirical question about individual's behavior and values.

While the standard environmental valuation techniques build on the assumption of utility maximizing behavior, the environmental stance of individuals is in many cases likely based on a deontological or rights-based approach to decision-making (e.g. Brennan, 1995). In this context, decisions are made based on whether the act itself is right or wrong regardless of its consequences, i.e. this approach recognizes the priority of the right over the good. For example, people may believe that aspects of the environment, such as wildlife threatened by a hydropower development project, have an absolute right to protection. They are thus willing to defend the existence or the well being of the environment apart from any instrumental value it provides. This is in line with the Sen (1977) distinction between *sympathy*, where concern for others forms one part of the utility function and *commitment*, where acts of altruism are chosen, even though they may result in lower utility for the individual. In other words, deontology denies the rationality attributed to making tradeoffs, whatever the commodity and therefore

[7] In Eq. (2) the unit of WTP is the quantity of private goods. However, by employing so-called indirect utility functions one can express WTP as a money metric measure. See, for instance, Freeman (1993). The choice between WTP and WTA as a welfare measure depends on the assumed property rights situation. For instance, in the case where the individual can be assumed to be the property right owner it can (theoretically) be valid to ask for the WTA in the case of a deterioration of the resource. Otherwise the individual may find the question (scenario) illegitimate and may choose to refuse to respond or can provide a protest bid.

suggests the existence of so-called lexicographic preferences. In this case, the axiom of continuity is violated, and the utility function in Eq. (1) is indefinable for an individual.[8] Thus, the indifference curves collapse to single points, denying the principle of substitution.

Spash and Hanley (1995) present empirical support for the existence of a deontological ethics, and conclude that standard valuation methods that elicit bids for biodiversity preservation fail as measures of welfare changes due to the existence of lexicographic preferences. Stevens et al. (1991) performed a contingent valuation study of species preservation in New England. A majority of the respondents (79%) agreed with the statement that: "all species of wildlife have a right to live independent of any benefit or harm to people." Still, when confronted with the WTP question, most of the respondents refused to pay. In other words, they were reluctant to choose between something of instrumental value (private goods) and a true moral position and in this way they applied a decision-making process inconsistent with the welfare economics paradigm.[9]

The motivation for the existence of a rights-based ethics, however, need not rely solely on empirical evidence. It is equally important to recognize that utilitarianism (and consequentialism) will not in itself be a sufficient moral theory for social choice. Since we cannot evaluate the net utility of an infinite number of alternatives, pure utilitarianism becomes a tautology. Some options simply have to be ruled out and this selection cannot be justified in utilitarian terms; instead we need to choose among options that we regard as morally or politically worth considering.

This does not imply that we should abandon the utilitarian approach to social choice. It merely points to the simple fact that people may approach the same issue in different ways, i.e. with different ethical standpoints. Environmental values often have a broad ethical content and since ethics are a matter for discussion[w1], environmental valuation ought to be endogenous to the political process and ultimately rely on social agreements. In other words, "the collective choice problem is, first of all, about advancing common ways of understanding what the pertinent issues are about. Only then can we develop a basis for collective choice predicated upon the elicitation of individual choice," (Vatn and Bromley, 1994, p. 142). Reasoned political argument among citizens does not exclude utilitarian (or indeed any other) belief systems but contextualizes them and helps us reflect upon our own arguments. We may not agree on the importance of different fundamental moral values but may still be able to come to a consensus on how to deal with moral aspects of practical issues. This consensus on the principles for social choice may (or is even likely to) involve a reliance on social-cost benefit analyses in some—but as indicated above not in all—instances.

This line of reasoning mirrors the work of Sagoff (1988, 1998). He suggests that individuals have two distinct roles; they act both as consumers with private preferences and as citizens with public preferences. Private preferences reflect what the individual thinks is good from a pure utility maximizing perspective, e.g. he or she prefers Coke to Pepsi. Public preferences, in contrast, state what a person believes is best or right for the community as a whole, e.g. 'society should not legalize drugs'. For instance, some people may regard environmental pollution as something inherently wrong, and what Sagoff rejects is the view of such moral objections as constituting just another kind of external cost that can and should enter a cost–benefit analysis.

Although the distinction between private and public preferences often is hard to operationalize, the consequences of not understanding the difference can lead to results that we would normally like to avoid. For example, economists usually argue that for the purpose of cost–benefit analyses it does not matter *why* people value environmental goods. As such, economists assume that all preferences are private and they grant equal credibility to every motive that underlies these preferences. To base social choice on this approach, Sagoff argues, is the equivalent of trying to decide whether a person on trial is guilty by

[8] The seminal work in this area is Georgescu-Roegen (1936).

[9] See also Common et al. (1997), who survey the empirical evidence on this issue and Russell et al. (2001).

discovering, before any evidence has been heard, what the preferences of the jury are in this regard and then calculating the net benefits of the two possible verdicts. It thus involves "an underlying confusion between preferences that may be priced and values that are to be heard, considered, criticized, and understood" (Sagoff, 1988, p. 95).[10]

This suggests therefore that, apart from simply 'speaking out' their given private preferences, individuals engage in a social process in which they form a collective understanding as citizens about what is appropriate, right or good, and in this way construct a *basis* for social choice. In other words, public preferences are endogenous rather than exogenous. For this reason, public values are also context relative, i.e. they are determined by social processes that play important roles in internalizing norms and beliefs about what is right and wrong.[11] Preferences are also likely to change over time due to the influence of education and cultural variations (Norton et al., 1998).

Private preferences towards private goods may of course also be endogenous and thus change over time, but normally this does not call for broader public deliberations about fundamental values. The social learning process however does become particularly important when individuals are confronted with public goods that: (a) they have little past experience of (i.e. preferences normally do not exist until we find a need to build them); (b) involve ethical dilemmas; or (c) have very complex characteristics. This is often the case when environmental goods are involved. The myriad of different classes of environmental effects, the many cross-cutting dimensions of these effects and the different risk characteristics involved cannot be casually separated in many cases. In addition, the conventional way of learning about the attributes of a good—learning by doing—becomes difficult and indeed often risky. It is one thing to choose between Pepsi and Coke, but another to choose between the preservation of an entire ecosystem and the development of a hydropower plant.

In summary, in this section we suggest that environmental goods and services embody characteristics that present serious ethical complications when social choices are to be made on the basis of recommendations derived from standard environmental valuation techniques. Preferences toward public goods are often endogenous to the political process and there is thus an important distinction between private and public preferences. The latter includes not only utility maximizing motives, but also other ethical positions, such as a deontological approach to decision making. In many cases, therefore, the initial challenge lies not in 'discovering' private preferences, but in specifying the conditions for public discourse over what is worth valuing and for what reason.[12] This becomes particularly important for many environmental goods, which are often both 'new' (e.g. global warming) and 'complex' (e.g. ecosystems).

4. The ExternE study as a basis for social choice in the power generation sector

In this section we discuss the relevance of the above theoretical discussion for social choice in the empirical context of power generation externalities

[10] Still, one important limitation of Sagoff's analysis is that even though he stresses the importance of public participation and public discourse for environmental issues, he does not attempt at characterizing this public sphere in a theoretically compelling way. See, however, Fiske (1991, 1992) for an interesting and systematic account of social interaction in which market pricing is only one of four relational models.

[11] This so-called deliberative approach to environmental valuation also lends support from the normative political theory of deliberative democracy, which recognizes that it is no less rational to focus on the procedure of the political decision-making process than on its outcome. See, for instance, Jacobs (1997) and Sagoff (1998) for reviews of this literature.

[12] See also the seminal work by Kapp (1978) who concludes: "Indeed the really important problems of economics are questions of collective decision-making which cannot be dealt with in terms of calculus deductively derived from a formal concept of individual rationality under hypothetically assumed and transparent conditions" (p. 288). Thus, for Kapp environmental policy was a question of political economy rather than a technical issue to be decided by cost–benefit analysis. Of course, in practice valuation based on cost–benefit analysis may not necessarily differ much from that provided by public deliberations. See Page (1992) for some empirical evidence on this latter point.

as addressed in the so-called ExternE project (European Commission, 1995a, 1999). This project aimed at evaluating the external costs of the different power generation fuel cycles in the EU. The results and methods of the studies have been utilized as inputs in important modeling work and have served as vehicles in developing additional methodological work in the environment and energy field .[13] As the ExternE project represents one of the most ambitious and internationally recognized attempts at coming up with 'true' external cost estimates for the different power technologies (Krewitt, 2002), it serves well as a case study of the ethical limits of environmental valuation in the power sector. Tables 1 and 2 present the different power generation externalities quantified and priced within the ExternE core project (European Commission, 1995a).[14]

All studies that form part of the project primarily use the bottom-up damage cost approach. The analyses begin by identifying the range of the burdens and impacts that result from the different fuel chains. Only impacts deemed to have 'significant' effects are included in the final assessment. These are quantified and monetized based on WTP measures, using methodologies appropriate for each specific externality. This implies that the ExternE project is not at all entirely comprehensive in its assessment of environmental externalities (e.g. it omits ozone impacts from gas-fired power generation).[15]

When inspecting Tables 1 and 2 we first note that most of the fuel cycles involve significant impacts on the health and deaths of humans ('public and occupational health'). In the ExternE project considerable attention was put on evaluating these impacts and much was learnt, especially about the importance of fine particles emissions for public health (Krewitt, 2002). In the core project, the value of a statistical life was used to calculate the external costs of mortality[16] and chronic and acute morbidity effects from air emissions were monetized using previous estimates of WTP to avoid different symptoms. However, according to a deontological ethics, human beings are moral ends in themselves and an infinite amount would be required to compensate for the death of a human being. This comes into direct conflict with the ethical basis of the ExternE project, which (implicitly) aims at maximizing society's total utility.

This does not imply that we should spend the entire public budget on saving lives and preventing morbidity impacts; it simply points to the fact that such impacts involve a moral dilemma. To what extent should we treat humans as means to an end (utility) or as ends in themselves? This question cannot be resolved with the help of cost–benefit analyses, but rather within the realms of public discourse.[17] It is not enough in this instance to make the remark that we do already reveal our preferences against health and death risks by our daily risk-taking behavior. "Precisely because we fail, [...], to give life-saving the value in everyday personal decisions [...], we may wish our social decisions to provide us the occasion to display the reverence for life that we espouse but do not always show," (Kelman, 1981, p. 38). This suggests also that, in contrast to the postulations of welfare economic theory, in social choices involving less than perfect information about risks it may be sensible to make a distinction between

[13] See, for instance, Bigano et al. (2000) and Vennemo and Halseth (2001).

[14] In 1999, the ExternE core project was followed up by the so-called national implementation projects (European Commission, 1999), whose aim has been to develop an EU-wide set of external cost data for the different fuel cycles and countries, utilizing the methodology developed within the core project.

[15] In addition, the focus is on environmental externalities, and externalities attributable to, for instance, fuel supply security are beyond the scope of the analysis. See, however, Bohi and Toman (1996) for an overview of the existence of energy security externalities.

[16] In the national implementation part of the ExternE project 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.

[17] Of course, public deliberations do not guarantee wise or viable decisions. Still, for the resolving of moral issues they should provide an appropriate (if not entirely sufficient) starting point.

Table 1
Externalities priced within the ExternE core project: coal, oil and gas

Externality	Coal	Oil	Gas
Public health	PM, ozone, and accidents: Mortality, morbidity, and transport impacts	PM and ozone: Mortality, morbidity, and transport impacts	PM: Mortality, morbidity, and transport impacts
Occupational health	Diseases from mining and accidents during mining, transport, construction, and dismantling	Accidents: death and injury impacts	Accidents: death and injury impacts
Agriculture	Sulfur, acidification, and ozone: crop and soil impacts	Sulfur, acidification, and ozone: crop and soil impacts	
Forests	Sulfur, acidification, and ozone damages	Sulfur, acidification, and ozone damages	
Marine	Acidification impacts	Accidents with oil tankers	Fishery: extraction impacts
Materials	Sulfur and acidification damages on surfaces	Sulfur and acidification damages on surfaces	Sulfur and acidification damages on surfaces
Amenity	Noise: operational road and rail traffic impacts		Noise: operational impacts
Global warming	CO2, CH4 and N2O damages	CO2, CH4 and N2O damages	CO2, CH4, and N2O damages
Total estimate (US cents/kWh)	2.8–4.1*	2.7–2.9*	1.7*

Source: European Commission (1995a).
* The global warming impacts constitute roughly half of the reported external cost estimates for coal-, oil- and gas-fired power. In the ExternE core project, the global warming estimates were drawn from Cline (1992).

Table 2
Externalities priced within the ExternE core project: nuclear, hydro and wind

Externality	Nuclear	Hydro	Wind
Public health	Radiation and non-radiation: mortality and transport impacts from operations and accidents		Accidents: travel to and from work
Occupational health	Radiation and non-radiation: mortality and transport impacts from operations and accidents	Accidents during construction and operation	Accidents during manufacturing, construction, and operation of turbine
Agriculture		Loss of grazing land	Acidification: damage on crops
Forests		Forest production loss due to flooding and land use	Acidification damages
Marine		Water supply and ferry traffic	Acidification damages
Materials			Acidification damages
Amenity		Visual amenity loss	Noise and visual amenity loss: operational impacts
Global warming			CO_2, CH_4, and N_2O damages
Recreation		Fishing and hunting	
Cultural objects		Objects of cultural and archeological interest	
Biodiversity		Terrestrial and aquatic ecosystems	
Total estimate (US cents/kWh)	0.0003–0.01	0.3	0.1–0.3

Source: European Commission (1995a).

preferences, in terms of individual choices made, and welfare, which is a broader measure of well-being (Johansson-Stenman, 2002).

Since different power-generation sources differ in terms of their relative impact on mortality and morbidity, the above concerns may have a direct impact on the actual choice between fuels. For example, the risks presented by nuclear power are generally more dominated by disease impacts than those of, say, gas and hydropower. In addition, the aggregation of effects of different severity (e.g. morbidity versus mortality) into a single monetary value also raises the ethical question of how society should weigh the importance of each of these impacts.

From an ethical point of view mortality and morbidity impacts are likely to differ from those externalities affecting *materials*, such as corrosion caused by acidic deposition. In the latter case, the implicit trade-off is between higher electricity production and less material damages. Parts of the natural environment (including humans) are (for all practical reasons) never at stake here and for this reason, private preferences may well serve as an appropriate basis for social choice.

Another ambiguity in how to deal with novel social choice problems when one considers the fundamental differences in the nature of risk between the different electricity alternatives. With nuclear power, an option with very low probabilities of very large negative impacts were introduced in the electric power arena. This is in contrast to fossil-fueled power generation, which gives rise to continuous but also comparably modest impacts. The Krewitt (2002) (p. 844) review of the ExternE project concludes that:

> The instruments for the assessment of consequences from beyond design accidents in nuclear power plant are well established, and the message from the use of such models is rather clear and non-ambiguous: the impacts from a single event can be very large, resulting in up to several ten thousand cases of fatal cancers, and in monetary terms they could amount to billions of Euro. Normalized to the probability of the event, and to the electricity generation over the power plant's lifetime, the expected value of risk (i.e., the probability times consequences) is low, a fact which is even robust against uncertainties in the accident probability.

Many experts claim that laypeople in general tend to overestimate the very low probabilities of nuclear accidents, but people are often unimpressed by arguments stating that the *expected* damages of nuclear are lower than those of other alternatives. An extended research tradition (e.g. Slovic, 1987) attempts to explain such behavior. In particular, it is noted that the public finds it especially hard to accept risks that are hard to identify because they arise from novel circumstances or technologies or have a catastrophic potential and may constitute a threat to future generations. Laypersons also rank as serious, risks that are involuntary, uncontrollable or having an uncertain and inequitable distribution of consequences, and for many power generation possesses a large number of these risk profiles (Ibid).

There is thus a large degree of 'catastrophe aversion' among the public. This is far from an indication of 'irrational' behavior; instead, it expresses that the willingness to accept a certain risk is related to the capacity to deal with the consequences should they arise.[18] For example, nuclear waste management risks are essentially irreversible after the plant has been commissioned, while the visual amenity and noise impacts from wind power are more or less reversible since the plant can be removed. Such differences are likely to affect the public preferences toward power-generating technologies. In sum, most people are not willing to engage in a trade-off discussion

[18] Of course, a neoclassical counter-argument would be that catastrophe aversion simply reflects the fact that the insurance market is insufficient and unable to correctly pool risks in the case of a catastrophic incident (e.g. Radetzki and Radetzki, 2000). For this reason, the government has to cover these additional risks and provide a de facto subsidy to the nuclear industry. Nevertheless, we argue that even in the presence of perfectly functioning insurance markets the moral dilemma would still be there, and it is unlikely that compensation for future accidents would make the perceived catastrophe aversion problem disappear.

regarding events that may lead to disastrous effects (for present or future generations) even though the probability of that disaster is extremely low. Thus, in such cases there simply exists no well-defined private utility function on which to base external cost estimates.

Furthermore, in trying to evaluate and assess the problems of nuclear waste management and radiation, individuals need to rely heavily on the statements of scientists. These can provide important information about the main physical relationships and may be able to present different scenarios and discuss the outcome of each. Still, the perceptions of what constitutes a significant risk are essentially socially constructed. The question of how risk should be evaluated therefore requires a broader public discussion in which, of course, scientists ought to take an active part.[19] This would enable the public to revise their perceptions of different risk profiles by considering the arguments of researchers as well as of other laypeople. In this way, any 'overestimations' of the risks involved could be removed, not by informing people about the 'true', 'objective' risks, but by encouraging them to reflect and engage in deliberations with others.

In the 1980s, many countries (e.g. Austria, Germany, Italy, Switzerland, etc.) put a moratorium on further nuclear expansion. These political outcomes do not only reflect the fact that nuclear energy was found economically inefficient. They also express ethical commitments towards future generations and unwillingness to accept the associated risks. Thus, nuclear energy is essentially a new 'good' with complex and far-reaching risk characteristics, for which most societies have not yet found an overall ethical position on which to base public and private decisions.[20]

This latter argument applies to the impacts of global warming (primarily caused by carbon dioxide emissions) as well. If it were not for reports from scientists, people in general would know nothing of their existence. The effects of global warming are inherently global, irreversible, long-term and asymmetrically distributed over time. This is in heavy contrast with other emissions from the power sector (e.g. sulfur dioxide), whose impacts are more tangible and directly connected to present human (dis)utility.

Again, society (and in this case countries) need to establish the conditions on which to base social choices in this matter. For example, an ethical position about the claims of future generations needs to be chosen. In this process, the need for *actual* compensation when there are damages to future generations,[21] as well as any inviolable rights of coming generations would have to be considered (Spash, 1993). Related to this, one has also to decide what is the relevant degree of risk acceptance, and within which limits of risk should pure cost–benefit be applied. A first attempt to agree upon a global climate policy was made by the rich market economies at the Kyoto conference in 1997. After making some necessary simplifying assumptions, Radetzki (2000) concludes that the *implicit* marginal price set on carbon dioxide emissions by the political process in Kyoto is somewhere in the range five to 25 times higher than the more explicit marginal damage cost estimates employed in the ExternE project (see Table 1).[22]

What do we make of this discrepancy? According to welfare economic theory, it suggests that the outcome of the Kyoto process was highly ineffi-

[19] The problem is complicated further by the fact that most scientists tend to transform genuine *uncertainty* into *risk*, where risk reflects a situation where the probabilities of different outcomes are known. In other words, they make an implicit assumption that their understanding of causal effects and overall system behavior (e.g. the nuclear power process) is more or less correct (Shackley and Wynne, 1996).

[20] In addition, the opposition towards nuclear has not only been directed towards environmental and risk-related issues. It has also been a struggle between the local and the national level of the political life, where local communities often see no benefits in nuclear development and resist to accept decisions exclusively taken at the national level.

[21] This differs from the ethical approach in welfare economics, which normally builds on a *potential* compensation criterion.

[22] This implicit price equals the carbon price, which would have to prevail in order to fulfill the emissions reductions agreed to at Kyoto. Krewitt (2002) also reports the existence of substantial differences between this implicit price and marginal damage costs for Europe.

cient, this since the constraints on carbon emissions agreed to are not motivated by generally accepted external cost calculations. However, if one accepts the ethical approach discussed in the present paper one should note that the 'Kyoto price' and the 'ExternE price' reflect different reasoning processes and are therefore not directly comparable. Within the ExternE project, hypothetical prices are established in *advance* as one of the raw materials for calculating the 'total' cost of energy. Thus, these prices together determine whether a specific energy source is better than another. The 'Kyoto price', on the other hand, did not play a causal role in the decision made at Kyoto but at most merely reflects the economic results of the political process. In this latter case, it is therefore the process that defines the legitimacy of choice, not the result. Accordingly, any inadequacies of the outcome arrived at under this process are essentially inadequacies of the process that produced them and cannot be attributed to the fact that the 'in effect' price put on carbon emissions is much higher than the 'true', or 'total' price presented in the ExternE project. As was suggested above (Section 2), this implies that there is a fundamental ethical difference between the abatement cost (regulatory revealed preference) approach and the damage cost approach.

In order to evaluate the legitimacy of the Kyoto process we need to know how ordinary citizens frame their discussions on global warming and develop preferences about climate policy. A major research project has investigated what dimensions of climate change are important to the European public (Kasemir et al., 2000). Focus groups, covering ≈ 600 people in seven densely populated areas in Western Europe were convened. The researchers conclude that the participants usually favored a two-stage policy process. First governments need to set limits—'tolerable windows'—on the behavior of firms and individuals, especially in terms of overall energy use. These limits reflected primarily ethical (and not economic) considerations expressed as safe minimum standards. In a second stage, however, cost considerations become highly important. Climate policy should find cost-efficient ways to stay within these 'windows'. Thus, the deliberations of the groups indicated clearly that value for money rather than monetary valuation—i.e. cost efficiency rather than cost–benefit analysis—appears to be the relevant issue for laypeople in Europe in attempting to reach a judgment on climate policy.

Finally, the ExternE project includes a contingent valuation (CVM) study of some of the impacts of hydropower development in Norway (European Commission, 1995b). These impacts comprise three basic damage components: losses of recreation, cultural objects and ecosystems/nature. The respondents were asked how much they were willing to pay to avoid the above impacts.[23] This sub-study, we argue, implicitly raises many of the ethical dilemmas posed in this paper.

First, the complexity of the three 'goods' differs much. Recreation is essentially a private good, and a hypothetical bid for, say, hunting or fishing permits may be as trustworthy as any market price. However, as soon as the valuation range is broadened to include entire ecosystems, the problem of what is actually valued—and for what reason—becomes apparent. In a CVM study, ecosystems are described in a manner that renders them commodity-like (with a use value and an existence value) and there may be little room for what we would normally claim is the most important aspect of an ecosystem—its functional aspects (e.g. its life-supporting mechanisms and the role of ecological diversity) (Vatn and Bromley, 1994). In addition, the site dependent impacts on local ecosystems may be hard to quantify.

Second, the complexity of ecosystems is also related to the moral philosophies held by individuals. If we believe that a particular ecosystem is essential for life to be worthwhile, there is an indirect moral commitment to the system itself. According to this view, there would be no substitute means for achieving human satisfaction,

[23] It is worth noting that the hypothetical price derived from this CVM study basically equals the total external hydropower cost of 0.3 U.S. cents per kWh reported in Table 2. The remaining external costs are, in other words, comparably small and range between 0.0004 and 0.001 U.S. cents per kWh (European Commission, 1995b).

and this invalidates a contingent pricing analysis.[24] A similar argument can be made for some cultural objects. People wish to see some pattern to their lives and they want their lives to be set in some larger context. In many instances, cultural phenomena provide exactly that desired context. This is in contrast to a pure recreation good; it can normally be replaced by something else with an equivalent value. For the above reasons, it is probably fair to conclude that the values derived from this study, although competently conducted, are likely to serve as an insufficient guide toward an informed choice between preservation and hydropower development.

5. Concluding remarks

The pricing of power-generation externalities, it is argued, is necessary for making consistent and meaningful comparisons between technologies. Tradeoffs (however unfair they may seem) must always be made, and it is best to make them explicit in a cost–benefit analysis. Our main argument in this paper, however, is that this argument is based on restrictive behavioral assumptions and ethical principles outlined in the welfare economics literature. We do not claim that one has to choose this philosophy or reject it; we simply point to the fact that choosing this particular perspective gives us only partial insight into many environmental issues. All policies that attempt to reflect human preferences have to be sensitive to the actual behavior of humans; they cannot simply assume that all humans possess a single well-defined utility function, which is 'employed' in all situations. Since environmental issues often have a broad ethical content and people tend to possess different preference structures, there are no simple answers to the question of how decision-makers should collect public preferences and integrate them into the environmental policy process. What is clear, however, is that any meaningful policy process should aim at incorporating these different modes of articulating preferences towards the environment.

In practice, most societies adopt a two-step approach to achieving environmental goals, and—as we have tried to show in this paper—probably for good reasons. Take the example of the US sulfur allowance system. First, the government sets limits on the behavior of firms and individuals. For example, the Environmental Protection Agency (EPA) sets a cap on overall sulfur dioxide emissions. Ideally, these limits (or minimum standards) reflect not only the social costs and benefits of the policy, but also society's attitude toward risk and its ethical commitments towards the rights of natural amenities and ecosystems. This first step therefore requires a broad political dialogue among citizens and experts in order to illuminate and address the dilemmas and the underlying value conflicts. In a second step, the EPA encourages electric utilities to buy and sell emission allowances. The utilities will do so only when benefits exceed costs. Thus, within the overall emission limit, pure cost–benefit principles are allowed to dominate choices. There is, in other words, a fundamental ethical difference between a tradable permit system (which to some extent represents the solution to a cost-effectiveness analysis), and a pure cost–benefit analysis (that forms the sole basis of the policy decision).

It may well be that the American government, in some sense, allows too much or too little sulfur emissions. Put differently, one may argue that the implicit price on sulfur emissions is too low or too high. However, it is hard to see in what way a cost–benefit analysis of the 'full' cost of electricity would help us resolve this. Environmental valuation based on the welfare economics theory is primarily a tool for aggregation of private preferences and not for public discussion. In a democratic society, however, the discussion itself is important, since ethical positions and public preferences tend to be endogenous to the political process. Our analysis of the ExternE study's evaluation of a number of electricity externalities shows that the understanding of people's preferences towards many environmental impacts in this

[24] This dilemma is probably best illustrated by the building of China's Three Gorges dam. It leads to the flooding of large tropical forests and to the displacement of millions of people (e.g. The Economist, 1999).

sector requires a stronger focus on the instruments and the content of political and moral debate. The ExternE project may very well have provided a nice starting point for such a discussion, but it will not be able to substitute for it. Any talk of the 'full' cost of electricity has thus to be understood as at best metaphorical.

We do not suggest in this paper that standard non-market valuation exercises are fundamentally flawed. Under some circumstances (e.g. private goods, few ethical conflicts, a lot of prior experience on the part of the valuer etc.), they provide very relevant and reliable information for policy makers. What we suggest, however, is that in other cases, e.g. for 'new', 'complex' goods, researchers need to take two issues more seriously than has been the case in the past: (a) the process of preference formation; and (b) the distinction between public and private preferences. Researchers must increasingly help people build preferences (rather than assume them as given).[25] In general, there is a need for combining analyzes based on intensive value structuring, involving small numbers of people in focus groups, with more extensive value information gathered via surveys from large numbers of people. Such studies may also involve monetary valuation (e.g. WTP elicitation), but should also include a strong focus on the ethical values held by the respondent.

Both public and private preferences are important for informed social choices. However, a common problem is that people often express public preferences in surveys designed to elicit private preferences. Put differently, people's view of the issues presented in the scenarios presented to them in CVM surveys is often not compatible with the theoretical framework used to interpret the responses. To some extent this is of course a practical problem, and one may, for instance, alter the scenario preceding the WTP question so as to only trigger private preferences (e.g. Russell et al., 2001). However, in order to trigger also the public preferences one would need to adopt a broader theoretical framework when analyzing people's responses/arguments in focus groups as well as in surveys. The usefulness of economics in making rational choices over limited resources is vital, but in the environment and energy field it must be complemented by other forms of social intelligence about what should be the important criteria in social choice.

[25] See also Johansson-Stenman (2002) and Gregory et al. (1993) for more on these issues.

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