Nuclear Development

Externalities and Energy Policy: The Life Cycle Analysis Approach

Workshop Proceedings
Paris, France
15-16 November 2001

NUCLEAR ENERGY AGENCY
ORGANISATION FOR ECONOMIC CO-OPERATION AND DEVELOPMENT

ORGANISATION DE COOPÉRATION ET DE DÉVELOPPEMENT ÉCONOMIQUES

En vertu de l'article 1er de la Convention signée le 14 décembre 1960, à Paris, et entrée en vigueur le 30 septembre 1961, l'Organisation de coopération et de développement économiques (OCDE) a pour objectif de promouvoir des politiques visant :

- à réaliser la plus forte expansion de l'économie et de l'emploi et une progression du niveau de vie dans les pays Membres, tout en maintenant la stabilité financière, et à contribuer ainsi au développement de l'économie mondiale ;
- à contribuer à une saine expansion économique dans les pays Membres, ainsi que les pays non membres, en voie de développement économique ;
- à contribuer à l'expansion du commerce mondial sur une base multilatérale et non discriminatoire conformément aux obligations internationales.

Les pays Membres originaires de l'OCDE sont : l'Allemagne, l'Autriche, la Belgique, le Canada, le Danemark, l'Espagne, les États-Unis, la France, la Grèce, l'Irlande, l'Islande, l'Italie, le Luxembourg, la Norvège, les Pays-Bas, le Portugal, le Royaume-Uni, la Suède, la Suisse et la Turquie. Les pays suivants sont ultérieurement devenus Membres par adhésion aux dates indiquées ci-après : le Japon (28 avril 1964), la Finlande (28 janvier 1969), l'Australie (7 juin 1971), la Nouvelle-Zélande (29 mai 1973), le Mexique (18 mai 1994), la République tchèque (21 décembre 1995), la Hongrie (7 mai 1996), la Pologne (22 novembre 1996), la Corée (12 décembre 1996) et la République slovaque (14 décembre 2000). La Commission des Communautés européennes participe aux travaux de l'OCDE (article 13 de la Convention de l'OCDE).

L'AGENCE DE L'OCDE POUR L'ÉNERGIE NUCLÉAIRE

L'Agence de l'OCDE pour l'énergie nucléaire (AEN) a été créée le 1er février 1958 sous le nom d'Agence européenne pour l'énergie nucléaire de l'OECE. Elle a pris sa dénomination actuelle le 20 avril 1972, lorsque le Japon est devenu son premier pays Membre de plein exercice non européen. L'Agence compte actuellement 27 pays Membres de l'OCDE : l'Allemagne, l'Australie, l'Autriche, la Belgique, le Canada, le Danemark, l'Espagne, les États-Unis, la Finlande, la France, la Grèce, la Hongrie, l'Irlande, l'Islande, l'Italie, le Japon, le Luxembourg, le Mexique, la Norvège, les Pays-Bas, le Portugal, la République de Corée, la République tchèque, le Royaume-Uni, la Suède, la Suisse et la Turquie. La Commission des Communautés européennes participe également à ses travaux.

La mission de l'AEN est :

- d'aider ses pays Membres à maintenir et à approfondir, par l'intermédiaire de la coopération internationale, les bases scientifiques, technologiques et juridiques indispensables à une utilisation sûre, respectueuse de l'environnement et économique de l'énergie nucléaire à des fins pacifiques ; et
- de fournir des évaluations faisant autorité et de dégager des convergences de vues sur des questions importantes qui serviront aux gouvernements à définir leur politique nucléaire, et contribueront aux analyses plus générales des politiques réalisées par l'OCDE concernant des aspects tels que l'énergie et le développement durable.

Les domaines de compétence de l'AEN comprennent la sûreté nucléaire et le régime des autorisations, la gestion des déchets radioactifs, la radioprotection, les sciences nucléaires, les aspects économiques et technologiques du cycle du combustible, le droit et la responsabilité nucléaires et l'information du public. La Banque de données de l'AEN procure aux pays participants des services scientifiques concernant les données nucléaires et les programmes de calcul.

Pour ces activités, ainsi que pour d'autres travaux connexes, l'AEN collabore étroitement avec l'Agence internationale de l'énergie atomique à Vienne, avec laquelle un Accord de coopération est en vigueur, ainsi qu'avec d'autres organisations internationales opérant dans le domaine de l'énergie nucléaire.

ORGANISATION FOR ECONOMIC CO-OPERATION AND DEVELOPMENT

Pursuant to Article 1 of the Convention signed in Paris on 14th December 1960, and which came into force on 30th September 1961, the Organisation for Economic Co-operation and Development (OECD) shall promote policies designed:

- to achieve the highest sustainable economic growth and employment and a rising standard of living in Member countries, while maintaining financial stability, and thus to contribute to the development of the world economy;
- to contribute to sound economic expansion in Member as well as non-member countries in the process of economic development; and
- to contribute to the expansion of world trade on a multilateral, non-discriminatory basis in accordance with international obligations.

The original Member countries of the OECD are Austria, Belgium, Canada, Denmark, France, Germany, Greece, Iceland, Ireland, Italy, Luxembourg, the Netherlands, Norway, Portugal, Spain, Sweden, Switzerland, Turkey, the United Kingdom and the United States. The following countries became Members subsequently through accession at the dates indicated hereafter: Japan (28th April 1964), Finland (28th January 1969), Australia (7th June 1971), New Zealand (29th May 1973), Mexico (18th May 1994), the Czech Republic (21st December 1995), Hungary (7th May 1996), Poland (22nd November 1996); Korea (12th December 1996) and the Slovak Republic (14th December 2000). The Commission of the European Communities takes part in the work of the OECD (Article 13 of the OECD Convention).

NUCLEAR ENERGY AGENCY

The OECD Nuclear Energy Agency (NEA) was established on 1st February 1958 under the name of the OEEC European Nuclear Energy Agency. It received its present designation on 20th April 1972, when Japan became its first non-European full Member. NEA membership today consists of 27 OECD Member countries: Australia, Austria, Belgium, Canada, Czech Republic, Denmark, Finland, France, Germany, Greece, Hungary, Iceland, Ireland, Italy, Japan, Luxembourg, Mexico, the Netherlands, Norway, Portugal, Republic of Korea, Spain, Sweden, Switzerland, Turkey, the United Kingdom and the United States. The Commission of the European Communities also takes part in the work of the Agency.

The mission of the NEA is:

- to assist its Member countries in maintaining and further developing, through international co-operation, the scientific, technological and legal bases required for a safe, environmentally friendly and economical use of nuclear energy for peaceful purposes, as well as
- to provide authoritative assessments and to forge common understandings on key issues, as input to government decisions on nuclear energy policy and to broader OECD policy analyses in areas such as energy and sustainable development.

Specific areas of competence of the NEA include safety and regulation of nuclear activities, radioactive waste management, radiological protection, nuclear science, economic and technical analyses of the nuclear fuel cycle, nuclear law and liability, and public information. The NEA Data Bank provides nuclear data and computer program services for participating countries.

In these and related tasks, the NEA works in close collaboration with the International Atomic Energy Agency in Vienna, with which it has a Co-operation Agreement, as well as with other international organisations in the nuclear field.

INTERNATIONAL ENERGY AGENCY
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The International Energy Agency (IEA) is an autonomous body which was established in November 1974 within the framework of the Organisation for Economic Co-operation and Development (OECD) to implement an international energy programme.

It carries out a comprehensive programme of energy co-operation among twenty six* of the OECD's thirty Member countries. The basic aims of the IEA are:

- To maintain and improve systems for coping with oil supply disruptions;
- To promote rational energy policies in a global context through co-operative relations with non-member countries, industry and international organisations;
- To operate a permanent information system on the international oil market;
- To improve the world's energy supply and demand structure by developing alternative energy sources and increasing the efficiency of energy use;
- To assist in the integration of environmental and energy policies.

** IEA Member countries: Australia, Austria, Belgium, Canada, the Czech Republic, Denmark, Finland, France, Germany, Greece, Hungary, Ireland, Italy, Japan, Korea, Luxembourg, the Netherlands, New Zealand, Norway, Portugal, Spain, Sweden, Switzerland, Turkey, the United Kingdom, the United States. The European Commission also takes part in the work of the IEA.*

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FOREWORD

Improving the sustainability of energy systems by incorporating energy externalities into related policy making represents a key challenge that has been studied by both the International Energy Agency (IEA) and the OECD Nuclear Energy Agency (NEA). The life cycle analysis (LCA) approach, which consists of an evaluation of the potential environmental impacts of a product through its life cycle "from cradle to grave", provides a conceptual framework for a detailed and comprehensive, comparative evaluation of energy supply options. LCA could help national energy policy making by pointing to opportunities to improve the sustainability of the full fuel cycle operations in OECD countries (for example, by improving the sustainability of mining practices) and providing quantitative input to the political debate on improving the sustainability of energy systems.

For these reasons, the IEA and the NEA jointly organised a workshop held in Paris on 15-16 November 2001 on the subject "Energy Policy and Externalities: The Life Cycle Analysis Approach". The workshop brought together internationally recognised experts in this field with energy policy makers to examine the state of the art in assessing and internalising external costs of energy in power generation and transportation using LCA, as well as the potential and limitations of the LCA method for its use in energy policy making.

These proceedings provide a record of the presentations made by the LCA experts along with a summary of the discussions held among experts, policy makers and members of the industry. The results of the expert analyses made available during the workshop suggest that when monetised, energy externalities can form a significant part of the total economic cost of certain sources of power generation or transportation fuels. While large uncertainties remain, and may prevent their direct application in policy making, the LCA results may at least provide a qualitative guide to policy makers on the relative environmental impacts of the different energy technologies and therefore on the implied benefits of greater utilisation of low-impact technologies. They may also serve as a basis for a constructive dialogue with key stakeholders on resource development and use, recognising that different stakeholders have different interests and responsibilities.

It is hoped that these proceedings of “Energy Policy and Externalities: The Life Cycle Analysis Approach” will play a similar role, helping to inform the debate among energy policy makers, the industry and other stakeholders on the challenges and opportunities of improving the sustainability of energy use.

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EXECUTIVE SUMMARY

Introduction

The International Energy Agency (IEA) and the Nuclear Energy Agency (NEA) organised jointly a Workshop entitled "Energy Policy and Externalities: The Life Cycle Analysis Approach" to discuss issues related to assessing external costs of energy using the life cycle analysis approach and investigate the relevance of this process for decision making.

The meeting, held on 15-16 November 2001 at the IEA Headquarters, was chaired by David Pearce and attended by some 75 experts from governmental bodies and industries. Bill Ramsay and Luis Echávarri opened the meeting by welcoming the participants.

David Pearce identified four issues in his opening keynote address: consistency between Life Cycle Analysis (LCA) and economic theory generally; uncertainties with respect to health-related externalities; uncertainties and relevance of discounting costs related to global warming; and the empirical underpinnings for "disaster aversion" in externality estimates.

He stressed the need to ensure economic consistency in assessing external costs noting that an environmental impact was only an externality if it was not fully compensated for. Occupational risks (which may already be accounted for in higher wage rates) and resource depletion (which might be reflected in market prices) are two areas where at least some of the costs have been internalised. The fact that most of the externality literature points to global warming and health effects as dominating energy externalities is important in the light of the controversy surrounding "dose-response" effects. While the usual academic conclusion that more research is needed always seems frustrating to policy makers, it should be recognised that we do not seem to know enough about the economic valuation of life risks to be confident about the kinds of adders being produced in externality studies. Likewise, estimates of global warming damage neglect adaptation responses and do not reflect adequately the allocation of costs and benefits both among rich and poor countries and in time. Also, we need to look far more rigorously at the way in which discounting should be integrated into damage estimates in particular for impacts/damage occurring in the long term.

"Disaster aversion" refers to an apparent higher economic value attached to a large number of deaths in a single accident compared to an equal number of deaths spread over in a large number of more frequent types of accident. The ExternE study and others have looked at this matter and attempted to develop aversion factors that would lead to higher risk estimates for nuclear power. However, there is little empirical literature to support this.

Finally, one can see the need for additional "meta-analysis" of the external cost literature in order to compare in a systematic way results from different studies and analyse differences in order to explain why damage estimates differ (assumptions, local conditions, technologies, methods, …).

Approaches & issues

The methodology and main conclusions of the ExternE project, carried out for the European Commission, were outlined by Ari Rabl, of the École des Mines, as one example of approach to external cost assessment. The methodology uses a bottom up approach to estimate the impacts of different emissions from different power generation and transportation fuel options through inventory of each emission, estimate of its dispersion, examination of the impact based on the dose response-relationship (impacts being measured essentially in terms of years of life lost) and the economic valuation of these impacts. The results generally show that the estimated external costs are much higher for fossil fuels (highest for coal and oil than gas), and much lower for nuclear and renewables (although photo-voltaic stands out as a renewable energy technology with relatively high impact). These results are subject to a large number of uncertainties and quantitative estimates are probably no more reliable than a factor of 3. Uncertainties arise not only from data limitations, but also from difficulties in quantifying certain impacts (ecosystem), assumptions about future management of waste and improvements in technology, and intergenerational considerations.

External costs of energy/electricity life cycles

The work on life cycle analysis presented by Louis Wibberley**,** BHP Sustainable Technology, covers a range of energy sources and technologies for steel and electricity production. A streamlined "cradle to gate" approach was taken, with a focus on energy/technology comparisons, detailed understanding of the production chain, and an attempt to look particularly for opportunities to improve environmental performance as measured by greenhouse gas emissions. The results show that the use of coal and natural gas for steel-making may have comparable emission impacts, if slag from the blast furnace is fully utilised as clinker for cement making, and off gases from the furnace are used to produce electricity. They also indicate potential gains from increased use of scrap inputs, and from using coal bed methane at the mine. Overall a reduction in greenhouse gas intensity of up to 50% could be obtained. In the power production sector, LCA shows the largest possibilities for improvement through use of more efficient technologies, use of biomass to displace coal and utilisation of fly ash in cement making. One interesting technological possibility is combining solar thermal technology with coal power generation, which improves net solar efficiency to 30-40% (compared to 13% for photo-voltaic). Estimated additional costs for large-scale use of solar thermal in an existing coal plant are about 0.04 cents US/kWh.

The results of "Well to Wheel Analysis" of transportation fuel alternatives undertaken for a consortium of oil companies (Shell, BP, Exxon Mobil) and GM by Argonne National Laboratory in the US were presented by Jean Cadu**,** Shell. The analysis evaluates separately the fuel cycle (well to tank) and the propulsion (and emission control) systems (tank to wheel). The analysis assessed 75 different fuel pathways (gasoline diesel, Fischer Tropsch diesel, ethanol, methanol, hydrogen, and natural gas) and 15 different propulsion systems (conventional, conventional hybrid, and fuel cell) on a single vehicle type (GM full-size pickup). The results show that in terms of primary energy consumption, petroleum hybrid engines or fuel cell engines are significantly better than most natural gas options and all renewable options. However, in terms of greenhouse gas emissions, fuel-cell vehicles offer a clear edge (although diesel-engine hybrid vehicles are close), but renewable fuel options using ethanol (from biomass fermentation) offer the lowest emissions. A similar study is now underway for Europe to reflect the differences in oil supply, refining system, and vehicle efficiency (an Opel Zafira is the vehicle model). The study will also examine a greater number of renewable fuel pathways.

A critique of the life cycle analysis approach, was presented by Edgar Furuholt, Statoil, following a brief introduction from Marc Darras, Gaz de France. He noted that production chains often produce multiple products, some for energy use, some for others and that allocation of the emissions was to some degree arbitrary and must be handled with care. Given the wide degree of variability of characteristics of oil and gas production, any emission estimate could be derived given the appropriate selection of wells, extraction processes etc. Similarly, impact assessments fail to take into account unknown health and environmental impacts of new chemicals. For example, in the case of the chemical MTBE, a life cycle assessment was performed but failed to foresee possible health and groundwater effects. The presentation argued that LCA has no objective scale, contains too many assumptions, and is too complex to provide transparent results. Therefore, it should not be used as the basis for comparing widely different generating options or as the basis for internalising external costs. On the other hand, it is valuable for systematic descriptions of resource use and environmental impact characteristics, and can be used more precisely when the production chains and technology options are all very similar, or in choosing amongst locations for the same technology option.

The work carried out by the "Centre d'étude sur l'évaluation de la protection dans la domaine nucléaire" (CEPN) within the ExternE study, was summarised by Caroline Schieber, CEPN, who presented its main results with emphasis on factors affecting the estimates of nuclear power external costs. Nuclear power impacts differ from the fossil-fuelled chain impacts in that they are principally the result of exposure to radioactivity from routine operation of fuel cycle facilities and power plants, and in the case of accidents. According to the ExternE study, based upon the French nuclear chain, most of the exposure results from the electricity generation and fuel reprocessing phases and takes place over a very long period. The external cost is largely attributed to impacts on workers while the cost of impacts on the public are rather small (about 0.00002 € per kWh). This figure is not greatly increased by accidents, using the "large consensus" assumption that such accidents would occur at a frequency of 1 per 100 000 reactor-years and that, in such an accident, 1% of the radioactive materials would be released to the environment. Even if a risk aversion effect is assumed, the figure for accidents would only be around 0.0001 € per kWh, still a small figure. Long-term impacts of mining waste are also quite low and their management according to current regulations greatly reduces releases as compared to past practices according to the data from UNSCEAR reports.

Externalities from hydropower projects were addressed by Frans Koch, IEA Implementing Agreement for Hydropower. He emphasised that the motto: "avoid (environmental externalities), mitigate (damages that can't be avoided), compensate (damages that can't be mitigated)", adopted within hydropower projects already actively contributes to reduce externalities. The Implementing Agreement surveyed a large number of energy LCA studies to better understand the relative position of hydropower. Emissions of greenhouse gases from hydropower dams are normally quite low, with few exceptions. The survey found that other positive benefits of hydropower dams such as irrigation or flood control are not normally taken into account by such studies. Energy security benefits are also not generally recognised, and would represent a useful extension of LCA. Also not all environmental impacts can be usefully internalised by an LCA (e.g. loss of visual amenity is not usually included in LCA), and LCA does not take into account cultural differences of the value of different amenities.

LCA analyses for photo-voltaic (PV) and wind were summarised by Paolo Frankl, Ecobilancio Italia. He noted that there were a large range of LCA values for PV in the literature. Estimates of LCA impacts, originally quite high although they included only PV modules, have fallen quite steeply despite including the balance of system materials in the assessment, because of the decreasing requirements for inputs into solar PV modules and the improved outputs. Whereas systems installed in the late 1990s could be expected to require eight years before their energy output exceeded the energy used to produce the panel, this figure could easily fall to 2.3 years based on latest technology available. There are further possibilities to reduce this through heat recovery. For amorphous silicon technologies, it is already less than one year. Emissions from current PV are only about a factor of 2.6 lower than emissions from the electricity displaced from the Italian grid, but can easily be expected to improve to a factor of 20 in the near future (and even higher factors may be possible with heat recovery). For wind, externalities are quite low, although the operation phase produces both noise and loss of visual amenity. Wind damage estimates are the lowest of all the ExternE fuel cycles studied. The experience with PV shows the need to look at LCA in a dynamic way, particularly with respect to new technologies. A new international research project, ECLIPSE, will look at the life cycle inventories for future power generation technologies, focussing on PV, wind, fuel cells, biomass and CHP technologies. Sensitivity analysis will look at the impact of rapid technological improvement and differences in local conditions. The output, by the end of 2003, will be a report and guidelines on how to assess these technologies in life cycle analysis work.

Comparative assessments in electricity and transportation

The results of LCA for power generation based on German conditions, presented by Alfred Voss, Institute of Energy Economics and the Rational Use of Energy, University of Stuttgart, show that coal (particularly lignite) power generation have the highest external costs in terms of years of life lost, followed by PV (because of its high energy intensity and low insolation in Germany) and natural gas with nuclear, wind and hydro giving the lowest results. In terms of costs, this implies coal/lignite have external costs around

3 € cents per kWh, gas and PV around 1 € cent, with nuclear wind and hydro about 0.1 € cents. If these external cost estimates are combined with direct costs, nuclear, which is already nearly competitive with coal and cheaper than natural gas, becomes the lowest cost option for power generation. There are however, many uncertainties in terms of data, choices of discount rate etc. LCA can provide valuable support to decision-makers with regard to technology evaluation, comparison of future energy supply options, cost benefit analysis of policy measures and extension of green-accounting frameworks. Since uncertainties are still so large, it is better to use LCA only where appropriate regionally differentiated pollutant-specific damage estimates for cost internalisation are available (e.g. for SO_x, NO_x and particulate matter). The risk of low probability accidents can be internalised into the monetary accounting system by introducing liability insurance obligations.

An analysis on upstream and life cycle emissions, including greenhouse impacts, in the transport sector was presented by Mark Delucchi, University of California Davis. The assessment of greenhouse gas impacts in the US shows that against a baseline gasoline vehicle, the impact of including the full fuel cycle generally reduces the relative advantages of alternative transportation fuels. While a switch to diesel is estimated to save 30% as compared to gasoline, the savings from natural gas/LPG are (around 20%), for ethanol from corn (8%) and for battery electricity vehicles using power from coal (6%) are much smaller. This is largely due to the use of LCA rather than end-use comparisons. However, the results also show that there would be large savings from the use of ethanol from fuel cells using methanol (39%) or natural gas (50%), while ethanol from wood in a conventional engine appears to have the greatest savings (63%). In external costs of motor vehicle use, analysis results were presented for both air pollution and energy security impacts (including SPR, military expenditures, macroeconomic costs and pecuniary costs) as well as water pollution, noise and congestion impacts. The results suggest that externalities amount to 1.2 US cents per mile travelled in gasoline powered vehicle. The most significant externality is related to air pollution. Costs associated with US defence, the SPR, and climate change are quite insignificant. The only other variable of significance is the impact on the economy, through the transfer of wealth outside the US (referred to as "pecuniary externality") and the oil price shock impacts on the economy. A comparison of external costs and subsidies for different transportation modes in the US (gas or electric cars, transit bus, light rail, heavy rail) showed that subsidies available to public transit system greatly outweigh the benefit in reduced externalities avoided. In the comparison of social costs of transportation alternatives, differences in external cost, while not trivial, are outweighed by the differences in direct costs or in subsidies.

Round table

A background paper prepared by the Secretariat was presented by Maria Rosa Virdis, IEA, to initiate the roundtable discussion. First, she remarked that previous speakers had stressed the limitations of LCA, particularly for purposes of policy making (internalisation of externalities, green taxation, standards) rather than its usefulness as a tool. She illustrated through a couple of examples how LCA could be used to correct existing policies or to set benchmarks for new policies. The first example relates to estimated external costs of diesel versus gasoline vehicle use. Based on results of the ExternE study, she suggested that health impacts (and damages) due to particulate emissions from diesel vehicles in a sample of European cities seem to be much higher than those caused by cars using gasoline and equipped with catalytic converters. However, the damages, while approximately covered by existing gasoline taxes are only fractionally covered in the case of diesel fuel (with prevailing tax burdens on diesel and gasoline in Europe). In fact, European countries on average tax diesel 20-40% less than gasoline, thus favouring the former notwithstanding its higher health costs. Several policy approaches were suggested to eliminate this distortion and to better internalise external costs. The second example considered the proposed EC policy to allow for a maximum subsidy of 5 € cents/kWh for electricity from renewable sources. This value was calculated on the basis of the external costs avoided by renewables, again based on ExternE results. Implications for the relative competitiveness of technologies and fuels were discussed, as well as some of the advantages and limitations of subsidy policies. While indicating some of the most obvious limitations of the LCA approach and possible areas of improvement, she recommended further development of the methodology and wider dissemination and use of its results as a tool to support policy making.

Following this presentation, five pannelists presented brief remarks on the usefulness of life cycle analysis for government policy making and business decision making.

Birgit Bodlund (Vattenfall) recounted the Swedish power industry experience with cradle to grave/gate life cycle analysis. She cited three values for LCA: informing political debate (the politicians were surprised by the results and asked why they had not been informed earlier); contributing to continuous improvement in design and operation of the power system; and certifying the environmental qualities of the power through Environmental Product Declaration (EPD 14025) in response to the requests of customers. She noted that the good marks nuclear gets through such analysis are not sufficient to answer criticism of nuclear power. The complexity of LCA means that its implications are more readily taken up by future decision-makers than by the current generation.

Marc Darras (Gaz de France) noted that decision-makers never limit themselves to a strict cost-benefit analysis. LCA helps identifying the various criteria, economic, social and environmental, that are involved in making decisions and the associated uncertainties. It thus can be a valuable tool in defining the context for complex decisions. However, such decisions ultimately require the actors concerned to rely on their own judgement for ensuring that appropriate weight is given to each criterion. LCA alone cannot set external cost because of the uncertainties in the data, the intergenerational dimension, and the exclusion of non-energy aspects. Decision-makers also need to keep in mind the geographic distribution of impacts, costs and benefits while LCA does not reflect it.

Ture Hammar (Danish Energy Agency) discussed how LCA evolved in Denmark from more integrated approaches to pollution reduction beginning with resource depletion and environmental impact assessment in the 1970s, to a concerted effort to develop “clean” energy technologies in the 1980s to a move to “internalising externalities” by eliminating subsidies for “dirty” technologies and introducing environmental taxes and subsidies for “clean” technologies. The present approach is also more comprehensive and integrates sustainability. While LCA can be a useful tool in thinking globally about energy matters in a sustainable development perspective, it cannot be the basis for political decisions. For example, the Danish perspective on the Swedish nuclear reactor near Copenhagen inevitably takes other factors into account. LCA needs to try and capture the dynamics of technology development. Another application of LCA is in support of “ecolabelling”, which aims at leading the energy consumers towards choosing environmentally sustainable production. The Danish electricity utilities have introduced “ecolabelling” for electricity and heat and published results comparing waste generation from different energy sources and found generally that CHP is less polluting than conventional fossil generation.

Ron Knapp (World Coal Institute) stated that the coal industry supports internalisation of all positive and negative externalities into energy costs but not focusing on a single externality. He commented that 7 of the 10 largest coal exporters and most coal importers are countries which do not have limits on their GHG emissions under the Kyoto Protocol. He commented that politicians are beginning to awaken to issues about energy security despite the recent fall in oil prices but doubted whether the market, with the “tunnel vision of bankers” was considering resource depletion in its decisions. He stressed that while coal improved its environmental performance dramatically over the past two generations, this change has come about through political will. Government policy should not close off technology options, but allow them to be improved e.g. CO_2 capture and storage technologies. When using LCA analysis, political boundaries and implications for different countries should be take into account. For example, two power stations in Tokyo, one coal-fired, one natural gas-fired, have about the same amount of emissions on a life cycle basis but gas has far lower emissions on an end-use basis and may thus be preferred over coal by the user country. Power companies cannot be expected to make choices on a life cycle basis unless there is an incentive from government policy makers for them to do so.

Marcella Pavan (Electricity and Gas Regulatory Authority of Italy) discussed the impact of market liberalisation on the role that regulators could play in dealing with externalities. The regulatory authority is principally concerned with promoting competition and ensuring high quality of service in electricity and gas. Among its tasks are to review contracts and satisfy itself regarding certain technical issues with respect to security of supply. In these conditions, it is difficult to see how much economic textbook principles on externalities can actually be applied in the regulatory context, given the different government institutions that are involved at different stages of the life cycle. Some Italian examples illustrate this point. First example, there are taxes on sulphur and nitrogen oxide emissions, but the tax levels (53 and 104.8 € per tonne, respectively) are less than 1% of the values of external costs found by ExternE. A second area is the use of externality adders in the valuation of DSM programmes. The law permits utilities to engage in energy conservation programmes and allows cost recovery to include a component that recognises reduced environmental damage. A third example is in the area of carbon taxes, which not only differ for different fuels but exclude/exempt autoproduction and ignore the effects of methane leakage of gas pipelines. She stated that the real problem with externality internalisation or life cycle analysis is the political will to use the results. This means more research needs to be done on country-specific internalisation tools to take into account the different policy contexts in each country. It must also take into account market liberalisation, through the use of market-based instruments such as tradable green or emissions certificates.

The debate between panellists and participants that followed the above presentations focused on the usefulness of LCA and its limitations. The discussions also identified some areas for future research. The role of taxes and subsidies was addressed to a certain extent, with emphasis on the challenge raised by uncertainties on external costs which limit their relevance in fixing taxes but are not an excuse for political inaction in front of polluting activities and potentially harmful subsidies.

LCA was found useful as a qualitative guide to policy-makers on the relative environmental impacts of different energy technologies and therefore on the implied benefits of greater utilisation of low-impact technologies. The role of LCA in communicating about impacts of energy and transport activities was underlined. For example, LCA results synthesising a wide range of data and analyses may facilitate information flows between government and industry policy makers and the public, and serve as a basis for a constructive dialogue with key stakeholders on resource development and use. LCA offers a framework to compare global impacts (i.e. from greenhouse gas emissions) and site specific impacts. The approach may be used for fine tuning existing technology choices, identifying opportunities for improvement in these technologies and guiding companies in choosing suppliers.

The limits to LCA applicability are due mainly to: uncertainties on results; incomplete scope (some impacts are not covered); static analysis (technology progress cannot be reflected); and site-specific character of the method. Uncertainties on the results raise a challenge for using the results in policy making. Since results are site-specific it is not easy to draw generic conclusions from LCA studies. The aggregated nature of LCA, encompassing an entire chain of activities taking place in various jurisdictions limits its relevance for policy making. Since the scope of LCA does not cover security of supply, ecosystem integrity, biodiversity, or social impacts, the approach is not comprehensive enough for measuring the sustainability of an energy system. Furthermore, LCA focuses on what is "in the light" (i.e. what can be analysed readily), but is not much help for criteria which cannot be readily quantified and differences in social value systems between countries are not reflected. Last but not least, technology developments, which may significantly change life cycle impacts, are not taken into account since the assessment is static and does not reflect dynamic system evolution.

The possible areas for future research identified include: assessment of externalities such as security and diversity of supply, as well as loss of forest cover; further investigations in the field of discount rates applicable in the very long term and value of statistical life; incorporation of dynamics, technology progress, in LCA; evaluation of energy policy measures with LCA; further effort to reduce uncertainties in ExternE; and establishment of a data base containing information on externality assessment and the way it is being used (possibly under the umbrella of the IEA and NEA).

Main findings from the workshop

David Pearce summarised the workshop through comments from an environmental economist's perspective. He suggested that the discussions should distinguish between damage done (impact) by a system (energy or transport in the present case) and a policy-relevant externality, i.e. cost associated with a damage (or a benefit) that is not internalised in the price paid by consumers. In the area of fuel taxes, although they are already very high, it is questionable whether environmental externalities are already incorporated. Similarly, the positive "employment externality" of energy production may be irrelevant as most economic theory would suggest that in near-full employment economies it does not exist. He questioned whether the incorporation of "accident aversion" as an external cost factor had an empirical basis. He wondered to what extent "September 11-type" events are already incorporated in the analyses of ExternE. He argued that while some questioned whether aesthetics or other qualitative externalities could be valued, there is research, some if it already incorporated in LCA, that does attempt to quantify such externalities through the willingness to pay to avoid them. However, biological diversity impacts are much harder to define since we do not know what they are, how to measure them, or how they are affected by different power generation or transportation fuel options. He stated that probably the most important issue in LCA is the question of time and discounting which is particularly critical in discussing the greenhouse gas emission problem since the damage caused by global warming will occur mainly in a rather long term future and will vary with time. He stressed the substantial interface between LCA and the economics of resource depletion and added that the question of whether current economic decisions reflect resource depletion is not yet answered. Finally, he reminded the audience that "politics will decide" how and to what extent externalities are ultimately incorporated into economic decisions and politicians are not making the best of all possible decisions in the best of all possible worlds.

Opening Session

ENERGY POLICY AND EXTERNALITIES: AN OVERVIEW ON EXTERNAL COST ISSUES

Workshop Chairman:
Professor David Pearce, University College London

ENERGY POLICY AND EXTERNALITIES: AN OVERVIEW

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1. The uses of externality adders

Substantial progress has been made in estimating the monetary value of the environmental impacts of different energy systems. Perhaps the best known study in Europe is that sponsored by the European Commission and known as the ExternE programme [10-15,16-19]. In the USA a comparable project is that jointly sponsored by the US Department of Energy and the European Commission [35, 36-37, 40, 42, 39, 38, 41]. There are many others. In each case what is sought is a monetary value of an environmental impact arising from a unit of energy, usually standardised as a kilowatt hour (kWh). These environmental impacts are usually termed "externalities". An externality exists if two conditions are met. First, some negative (or positive) impact is generated by an economic activity and imposed on third parties. Second, that impact must not be priced in the market place, i.e. if the effect is negative, no compensation is paid by the generator of the externality to the sufferer. If the effect is positive, the generator of the externality must not appropriate the gains to the third party, e.g. via some price that is charged. In the energy externality literature, the procedure of expressing the externalities in, say, cents or milli-euros (1 000th of an Euro = m€) per kWh results in an "adder". An adder is simply the unit externality cost added to the standard resource cost of energy. Thus, if an electricity source costs X m€ to produce or deliver, the final social cost of it is (X+y) m€ where y is the externality adder.

While externality adders have been researched most in the context of energy, they are increasingly being estimated for other economic sectors, notably transport [16, 18] and agriculture [22,47,54]. What are the uses of such figures?

First, such figures could be used to guide *investment decisions*. If major electricity sources remain in public or quasi-public ownership, then the full social cost of electricity by different sources could be used to plan future capacity, with preference being given to the source with the lowest social cost. Where electricity is privately owned, then full social cost can be used by regulators to guide new investment or to act as an effective environmental tax, leaving the private owners to respond accordingly.

Second, adders can be used to estimate *environmental taxes*. While the use of adder estimates is not typically used in this way, the UK has at least two taxes based on externality estimates which, in turn, contain elements of external estimates taken from energy adder studies (the aggregates tax and the landfill tax).

Third, adders may be used as an input into *modified national accounts*. Here the idea is to replace GNP (or, more correct, net national product, NNP) with a measure that accounts for the depreciation of natural resources, so that "green" NNP becomes NNP – depreciation on resource – damage to the environment).

Fourth, adder estimates may be used for *awareness raising*, i.e. simply drawing attention to the fact that all energy sources have externalities which give rise to economically inefficient allocations of resources.

Fifth, adders might help with some notion of *priority setting* for environmental policy. The basic principle would be that, as a first approximation, attention should be paid to those activities generating the highest externalities. Better still, activities should be prioritised by some cost-benefit principle, so that adders are ranked according to the ratio of reduced adders to the cost of securing that reduction.

Clearly, then, estimating externality adders is potentially very informative for policy purposes. While a huge amount of research, and a large sum of money, has gone into estimating adders, problems remain. In some cases, the search for figures that can be used has perhaps obscured the need to think more fundamentally about how adders are derived, the conditions under which they might be considered reasonably valid, and the uses to which they are put. We therefore focus on just a few of the more important issues.

2. Externality and environmental impact: consistency with economic theory

Some of the adder literature proceeds as if any "external" impact constitutes an externality. But this is not so. The second attribute of the definition given previously is that any third party impact must be unpriced, i.e. uncompensated or unappropriated. To illustrate the problem, consider occupational health effects, i.e. impacts on those who work in the energy industries, and, for that matter in industries that supply the energy industries, e.g. mining, or which dispose of waste from the energy industries. Since the adder literature adopts a life cycle approach, these impacts could be important. But one of the methodologies used to estimate the value of risks to life is in fact based on the notion that wages in risky occupations are higher, other things being equal, precisely because of the risks involved. In other words, risks are "internalised", i.e. compensated for, in wage payments. If this is true, then one cannot include occupational risks in adder estimates. Indeed, there is a contradiction in using "values of statistical life", most estimates for which come from wage-risk studies, whilst simultaneously arguing that occupational risks reflect an externality.

We can illustrate by looking at the ExternE 1995 estimates of the externalities from a pressurised water reactor (PWR) [14, p. 191]. Taking a discount rate of 3%, the estimates there suggest that the total externality in m ECU (now m) is 6.00 E-02 or 6×10^{-2} m ECU. Of this, 5.73 E-02 consists of occupational effects. In other words, over 95% of the externality is accounted for by occupational effects. But if these effects are internalised in wages, they should not be included and the resulting externality is trivial at 0.27 E-02.

Occupational effects are perhaps one of the easier sources of double counting to identify. But there are similar problems with accidents in the transportation phases of the life cycle analysis. Some accidents are undoubtedly truly external, but in many cases risks are already internalised in the decision to drive or go by train etc. The conclusion has to be that more care needs to be devoted to the consistency between the estimates and the underlying economic theory that must be obeyed if the estimates are to be regarded as useful for policy purposes.

3. Valuing statistical lives

A second major issue concerns the valuation of health effects in combined LCA and valuation studies. A glance at both the externality adder literature and cost-benefit studies of pollution controls shows that (a) health damages tend to dominate measures of externality and health benefits dominate cost-benefit studies, (b) within health effects, changes in life expectancy dominate. Table 1 shows a selection of studies relating to air pollutants and reveals that health benefits account for a minimum of one-third and a maximum of nearly 100% of overall benefits from pollution control. Moreover, in most cases these benefits exceed the costs of control by considerable margins. Health benefits therefore "drive" positive benefit-cost results. Nor is this outcome peculiar to the European Union. The US EPA's retrospective and prospective assessments of the Clean Air Act produce extremely high benefit-cost ratios, e.g. 44 for the central estimate of benefits and costs [58]. Moreover, EPA regards these as probable underestimates. In turn, the benefits are dominated by health benefits (99% if damage to children's IQ is included). The EPA's analysis has, however, been subjected to very critical analysis [29,50]. By contrast, the European studies appear not to have attracted much by way of critical comment.

It may be the case that there are very high benefit-cost ratios for air pollution control, but there are at least two reasons for a feeling of unease about the results that are being obtained:

(1) The relevant studies tend to omit ecosystem benefits, despite the fact that, for acidifying substances in the wider Europe, ecosystem protection is the driver for the UN ECE region air pollution Protocols under the Convention on Long Range Transport of Air Pollution (LRTAP). If the presumption of the Convention Parties that ecosystem damage is of dominant importance is correct, this would suggest that benefit cost ratios are substantially higher than the factors of three to five being recorded in the European studies. Some would regard this is adding to doubts about the analysis, rather than reducing them.

(2) The European studies suggest that benefits exceed costs even for scenarios defined in terms of "maximum technologically feasible reduction" (MFR) of pollutants, i.e. scenarios in which the most pollutant-reducing technologies are used. Such scenarios should be characterised by very high marginal abatement costs at very high levels of pollution reduction, precisely the context where one would expect incremental benefits to be less than incremental costs. While the benefit cost ratio does appear to fall for such scenarios relative to other more modest abatement targets, the reduction is not dramatic and benefits continue to exceed costs. Thus, AEA Technology [6] finds a benefit cost ratio of 2.17 for a MFR scenario, compared to 2.87 for practical targets based on the relevant Protocol. The incremental benefit cost ratio of going from Protocol targets to "MFR" targets is 1.6.

A similar picture emerges in the ExternE studies. Taking the UK National Implementation study, but omitting occupational health for the reasons given previously, Table 2 shows the percentage of externality due to "public health" effects and to global warming. We return to the global warming issues shortly. The estimates suggest the following conclusions. First, global warming and public health effects account for virtually all the externality from all fuel cycles[1]. Second, for coal, oil and orimulsion the two effects are broadly comparable. For gas, global warming is the overwhelming impact. For nuclear and the renewables, public health dominates and global warming is relatively unimportant.

1. This needs to be qualified because the ExternE estimates, along with other studies, generally omit ecosystem impacts beyond crop damage.

Table 1. **Health benefits as a percentage of overall benefits in recent cost-benefit studies**

Study	Title and subject area	Benefits as % total benefits
Holland and Krewitt [23]	*Benefits of an Acidification Strategy for the European Union*: reductions of SO_x, NO_x, NH_3 in the European Union.	86-94%. Total benefits cover health, crops and materials.
AEA Technology [1]	*Cost Benefit Analysis of Proposals Under the UNECE Multi-effect Protocol*: reductions of SO_x, NO_x, NH_3, VOCs.	80-93%. Total benefits cover health, crops, buildings, forests, ecosystems, visibility.
AEA Technology [2]; Krewitt *et al.*[28]	*Economic Evaluation of the Control of Acidification and Ground Level Ozone*: reductions of NOx and VOCs. SO_2 and NH_4 held constant.	52-85% depending on inclusion or not of chronic health benefits. Total benefits include health, crops, materials and visibility.
AEA Technology [3]	*Economic Evaluation of Air Quality targets for CO and Benzene.*	B/C ratio of 0.32 to 0.46 for CO. Costs greatly exceed benefits for benzene. Benefits consist of health only.
AEA Technology [4]	*Economic Evaluation of Proposals for Emission Ceilings for Atmospheric Pollutants.*	B/C ratios of 3.6 to 5.9. Health benefits dominate.
AEA Technology [6]	*Cost Benefit Analysis for the Protocol to Abate Acidification, Eutrophication and Ground level Ozone in Europe.*	VOSL + morbidity accounts for 94% of benefits. B/C ratio = 2.9.
IVM, NLUA and IIASA [25]; Olsthoorn *et al.*[43].	*Economic Evaluation of Air Quality for Sulphur Dioxide, Nitrogen Dioxide, Fine and Suspended Particulate Matter and Lead:* reductions of these pollutants.	32-98%. Total benefits include health and materials damage.

Note to Table 1: we have selected results using VOSL (value of statistical life) rather than “VOLY” (value of a life year) since the latter are not correctly estimated in the studies that also provide VOLY results. See text for discussion.

Table 2. **Percentage contribution of public health and global warming damages in all damages: ExternE, UK**

	Coal	**Oil**	**Orimulsion**	**Gas**	**Nuclear**	**Wind**	**Biomass**
Health	44	48	41	20	81	68	85
Gwarming	53	50	56	79	15	22	9
TOTAL	97	98	97	99	96	90	93

Source: adapted from AEA Technology [5].

Notes: health refers to public health only.

The fact that health and global warming effects dominate is important since both are very controversial both in terms of the "dose-response" literature and in terms of economic valuation. We deal first with just a few of the health issues. A fuller discussion can be found in [44].

The epidemiology literature linking air pollution to health effects is large. In terms of risk of death, there are two types of literature. The first relates acute episodes of pollution to life risks and the second, a far smaller literature, relates chronic exposure to air pollution to life expectancy. Most of the economic valuation literature deals with the former, i.e. with acute effects. But it is becoming increasingly clear that the chromic exposure epidemiology is more important, although acute studies still have a role to play. One of the problems with acute studies is that they may tell us numbers of people dying from acute effects but not the period of life that is foreshortened. There is a debate as to whether the life periods concerned are very short indeed, a matter perhaps of just a few days in OECD countries, or whether what evidence we do have on life foreshortening understates the true effects. This debate is well rehearsed in the contributions in [45]. The second problem is that, surprisingly, the epidemiology tells us little about the age groups that are affected. But the available evidence suggests, as one might expect, that the relevant deaths tend to occur in much older age-groups, usually in the over 70s. Combing these two likely facts of short duration loss of life expectancy with the age effect suggests that the relevant economic value will be the willingness to pay of over 70s to avoid days rather than years of life loss. Again surprisingly, we have limited evidence on how willingness to pay relates to age, but what we have suggests that it will be lower than the willingness to pay of median age groups involved in accidents. Yet it is studies relating to the latter that tend to determine the "value of statistical life" (VOSL) used in cost-benefit and life cycle externality studies. Figures such as €3 million are common in the ExternE studies, for example. No-one is suggesting that the values of the older generation do not matter, but we do have to question whether values such as €3 million can possibly be relevant to such impacts.

Turning to chronic mortality, while the relevant studies are far fewer and, even then, some of them simply borrow dose-response coefficients from previous studies, there is some suggestion that chronic exposure foreshortens life by perhaps six months and maybe one year. Supposing this to be true, the question then arises of what economic value we should attach to such epidemiological effects. If the effect of chronic exposure is to induce illness which may itself ultimately result in life foreshortening, then we should definitely be concerned to estimate the willingness to pay to avoid that illness. For the reduced life expectancy itself, however, it looks as if the correct value is what we are willing to pay today to extend our lives by, say, six moths when we are in our 70s or even 80s. We have very little evidence on what these sums are since the willingness to pay studies we do have relate to risks that are reduced (or increased) with effectively an immediate effect (e.g. a road accident).

Johannesson and Johansson [26] report a contingent valuation study in Sweden where adults are asked their willingness to pay for a new medical programme or technology that would extend expected lifetimes conditional on having reached the age of 75. Respondents are told that on reaching 75 they can expect to live for another 10 years. They are then asked their WTP to increase lifetimes by 11 years beyond 75, i.e. the "value" of one extra year. The results suggest average willingness to pay across the age groups of slightly less than 10 000 SEK using standard estimation procedures and 4 000 SEK using a more conservative approach, or say €600-1 500. This is for one year of expected life increase. Using the formula:

$$VOSL(a) = VOLY \sum_{t} 1/(1+r)^{T-a}$$

As suggested in [26] these values are consistent with "normal" VOSLs of 30 000 to 110 000, *substantially* less than the VOSLs being used in cost-benefit and externality adder studies. Since T-a is obviously less the older the age group, then the relevant VOSLs will decline with age.

It is perhaps worth noting that "rules of thumb" used in the ExternE work are not valid. The approach to valuing a "life year" in the ExternE studies proceeds as follows. A "value of a life year" or VOLY[2] can be thought of as the annuity which when discounted over the remaining life span of the individual at risk would equal the estimate of VOSL.

2. The ExternE notation is "YOLL" for "year of life lost".

Thus, if the VOSL of, say, £1.5 million relates to traffic accidents where the mean age of those involved in fatal accidents is such that the average remaining life expectancy would have been 40 years, then:

$$VOLY = VOSL/A$$

where $A = [1-(1+r)^{-n}]/r$ and n is years of expected life remaining and r is the utility discount rate. Examples are shown below in Table 3 for n = 40 years.[3]

Table 3. **Deriving VOLYs from VOSLs: examples**

VOSL € m	**Utility discount rate = 0.3%. A = 37.6**	**Utility discount rate = 1.0%. A = 32.8**	**Utility discount rate = 1.5%. A = 29.9**
1.0	26 595	30 460	33 445
1.5	39 894	45 690	50 167
2.0	53 190	60 920	66 890
3.0	79 787	91 138	100 000

These VOLY numbers can then be used to produce a revised VOSL allowing for age. At age 60, for example, suppose life expectancy is 15 years. The VOSL(60) is then given by:

$$VOSL(60) = \Sigma VOLY/(1+r)^{T-60}$$

where T is life expectancy. In the case indicated, this would be, at a 1% discount rate and a "standard" VOSL of €1 million:

$$VOSL(60) = (30\ 460).(13.87) = €422\ 480$$

The result is that the age-related VOSL declines with age and this appears to accord with the findings noted earlier that willingness to pay probably does decline with age. The generalised formula for age related VOSL is:

$$VOSL(a) = [VOSL(n)/A]\sum_t 1/(1+r)^{T-a}$$

where a is the age of the individual or group at risk, T is life expectancy for that group, VOSL(a) is the age-adjusted VOSL and VOSL(n) is the "normal" VOSL.

3. Another way of saying the same thing is that VOLY = VOSL/Discounted expected lifetime. Strictly, the relationship holds only when utility of consumption is constant in each time period.

There are several reasons for doubting the usefulness of the VOLY approach when it is based on a VOSL.

First, the basis of the VOLY approach is the life-cycle consumption model with uncertain lifetime. It is well known that such models assume utility depends on consumption alone and not on the length of life. Lifetime utility does indeed vary with life expectancy but the route is via consumption not via time itself. It seems unlikely that individuals are indifferent to time remaining. There are also additional restrictions on the model to ensure that WTP is proportional to the discounted value of life expectancy. Thus, it can be questioned whether the underlying theory needed to derive VOLYs from VOSLs is itself tenable.

Second, the theory forces the age-distribution of VOLYs to take on a monotonically declining form: VOLY simply declines with age. What evidence we have, however, suggests that willingness to pay follows an inverted "U" shape curve, rising to a median age and then falling. If so, the VOLY construct is a poor representation of "true" WTP over the lifetime of individuals.

What can we conclude on the health effects and valuation in externality adder studies? While the usual academic conclusion that "more research is needed" always seems frustrating to policy makers, the fact is that we do not know enough about the epidemiology and we certainly do not know enough about the economic valuation of life risks to be confident about the kinds of adders being produced in externality studies. In some cases, being more certain of the absolute magnitudes of the adders may not matter too much. For example if we simply wish to prioritise investments by social cost, a ranking may not be affected by what values we use. But if we wish to use the values to set, say, energy taxes, then the absolute magnitudes do matter.

4. Global warming

Table 2 above showed that the other major component of externality values derives from global warming effects. Everyone is familiar with the scientific uncertainties in warming studies. To those uncertainties we must add economic valuation uncertainties. Uncertainty is not a reason for neglecting economic valuation – there is a widespread but erroneous view that if we avoid trying to estimate economic values what we will end up with is a more certain base for policy making than if we do not. The reality is that whatever decisions we make about global warming policies they will all imply some set of economic values. It is better to be as explicit as we can about the numbers rather than masking them by procedures that allegedly do not use them.

Table 4 shows some of the results taken from economic studies of global warming. The relevant magnitude is the economic value of one tonne of greenhouse gas emitted now. This must allow for residence time in the atmosphere and the fact that greenhouse gases are cumulative. The relevant concept is therefore a discounted economic value of damage due to the “marginal” (i.e. extra) tonne of pollutant. This is the basis of Table 4.

Table 4 shows that the relevant values per tonne carbon vary substantially with the discount rate assumed. This is hardly surprising. Unfortunately, while economists are reasonably good at estimating social discount rates for a single nation, the relevant discount rate for the world as a whole is a more elusive concept. Yet it is the relevant one because the damages recorded in Table 4 relate to the world as a whole. More complex still, economists have not yet secured a consensus on what the relevant rate would be for very long-lived environmental effects of the kind that would typify global warming damages and, just as relevant for externality adders, nuclear waste disposal. The most promising contribution to date appears to be that of Weitzman [59] who shows that the long term discount rate should almost certainly be declining with time (Annex 1 to this paper is an attempt to derive Weitzman’s result in a much simpler way). Our first observation about global warming estimates, then, is that we need a far more rigorous look at the way in which discounting should be integrated into the damage estimates.

A second observation is that all of the estimates in Table 4 are based on the “dumb farmer” syndrome. They do not make any allowance for adaptation to global warming. Yet, if we know anything at all, we know that people do not stand idly by and do nothing in the face of environmental damage. Unfortunately, we appear to have only one set of studies that give us any idea what would happen to the damage estimates if we do assume adaptation. In an important contribution, a volume edited by Mendelsohn and Neumann [31] shows that total damages to the US economy could be zero instead or positive once adaptation is assumed.

There are problems with this claim if we wish to extrapolate it to the global damage estimates underpinning the marginal damage estimates in Table 4. First, the estimates relate to the market sectors of the US economy only. Yet it is the non-market sectors such as ecosystem functioning that perhaps give the greatest cause for concern. Second, adaptation in the USA is likely to be greater than in the developing world where fewer technological options are available. Nonetheless, the global warming damage estimates in the externality adder literature do need revisiting in light of the Mendelsohn-Neumann findings.

Table 4. **Marginal damages from greenhouse gases ($ tonne carbon)**

Study	**Estimate $ tC . Base year prices: 2000**			
Period	*1991-2000*	*2001-2010*	*2011-2020*	*2021-2030*
Nordhaus [32]	9.3	9.3	–	–
Nordhaus [33]				
p = 0.03, best guess	6.8	8.7	11.0	12.8
p = 0.03, expected value	15.4	23.0	33.9	–
Nordhaus and Boyer [34]	6.4	9.1	11.9	15.0
Fankhauser [21]				
with p =0,0.005,0.03	26.0	29.2	32.4	35.6
with p =0	62.5	–	–	80.5
with p =0.03	7.0	–	–	10.6
Cline, [9]				
with s = 0	7.4-158.7	9.7-197.1	12.5-238.1	15.1-282.9
Peck and Teisberg [46]				
with p =0.03	12.8-15.4	15.4-17.9	17.9-23.0	23.0-28.2
Maddison [30]	7.6-7.8	10.4-10.8	14.2-14.8	18.8-19.4
Eyre *et al.* [20]				
with s = 0	181.8	190.7		
with s = 1	93.4	92.2		
with s = 3	29.4	25.6		
with s = 5	11.5	9.0		
with s = 10	2.6	1.3		
Tol [57]	14.1	16.6	19.2	23.0
Roughgarden and Schneider [48]: lower bound = Nordhaus, upper bound = Tol	6.4-14.1	7.7-16.6	10.2-20.5	12.8-26.9

Notes:

- s = social discount rate. Eyre *et al.* [20], estimates are for 1995-2004 and 2005-2014 and the estimates here exclude equity weighting. p = utility discount rate and s = the overall discount rate. Roughgarden and Schneider's [48] ranges derive from placing the models of Fankhauser [21], Cline [8], Titus [55] and Tol [57] into Nordhaus's DICE model framework [33]. The upper end of the range should, strictly, coincide with the marginal damage estimates in Tol [57].
- Most original estimates are in 1990$ and we have assumed an escalation of 2.5% p.a. inflation. Note also that Table 1 shows the considerable sensitivity of estimates to discount rates. The discount rates given for Fankhauser's estimates relate to the pure time preference rate component, p, only. According to Fankhauser [21] his social cost estimates based on the distribution of values for p are equivalent to a "best guess" value of 0.5% for p. To this must be added a value for the elasticity of the marginal utility of income multiplied by the expected growth rate. Fankhauser and Eyre *et al.* [21, 20], take the elasticity to be about unity, so that the only variable is the expected long term economic growth rate of income per capita. Setting this at 1.6-1.8% pa, the discount rate would be 2.2 to 2.3%. Accordingly, the values in Eyre *et al.* [20] of 0-3% are more relevant for purposes of comparison.

While there are many problems with the global warming estimates, one that is much discussed and debated deserves some comment. In the ExternE work the global warming damages are "equity weighted". Global warming damages are spread across the world and affect both rich and poor countries. But poor countries have substantially lower incomes than rich countries so that one Euro's damage to them has a higher "disutility" value than one Euro's damage to rich countries. Cost-benefit analysis typically works with notions of willingness to pay that do not reflect this adjustment for different utilities of a money unit. But there is no unique way to do cost-benefit so it is perfectly legitimate to seek to maximise utility-adjusted benefits and costs. This is what the ExternE programme does [17]. In terms of the economic value of a tonne of carbon, the effect of equity weighting is to *raise* the value per tonne. This is because damages done to richer individuals tend to be used as the numeraire, with the result that damages to poorer individuals are increased relative to what they would have been without equity weighting. As shown in Eyre *et al.* [20], the effect of equity weighting is approximately to *double* the marginal damage estimates. Once again, if the policy issue of concern is one of prioritising fuel cycles the use of equity weighting may not matter too much. But if absolute levels of damages matter, then it is crucial to justify equity weighting. Unfortunately, studies using equity weighting are not very forthcoming on what this justification is. The ExternE reports suggest that it is consistent with maximising "utility" as opposed to willingness to pay-based measures of costs and benefits. This is correct. But several questions then arise. First, why do we seek to maximise equity-weighted net benefits in the global warming context but not in any other context? Second, if we do adopt equity weights, what is the justification for selecting one particular set of weights rather than another?

As far as the first question is concerned, the ExternE reports suggest that equity, and by implication equity weighting, is integral to the Framework Convention on Climate Change (FCCC) [20, p. 9]. This is debatable. The equity notions in the FCCC relate primarily to the concept of differentiated responsibility, i.e. since rich countries are bigger emitters of greenhouse gases they bear more responsibility and should act first. It is a substantial leap from this idea to one of weighting costs and benefits. More telling is the fact that equity weighting in cost-benefit analysis of global warming control will raise the benefit-cost ratio of taking action. In other words, more global warming control would be undertaken with equity weighting than without it. This seems fair until we recall that whatever is spent on global warming control is not spent on other things, and the other things may include foreign aid, technology transfer etc., all of which benefits the poorer countries. In turn this suggests that we should either equity weight all policy measures that have an effect on poor countries or we should not equity weight any of them. Isolating global warming and ring-fencing it as if it is unique is not a tenable proposition. The situation could be even more complex, since global warming expenditures are likely to benefit the descendants of the current poor rather than the poor now. Yet the descendants are likely to be richer than the current poor, so that the policy of weighting damages may simply reinforce a tendency to divert resources from solving the problems of the current poor. This was pointed out by Schelling [49].

Suppose, however, that we do accept equity weights. What are the relevant weights? ExternE [20] suggests that the weights reflect diminishing marginal utility but they actually select a specific value for the weighting procedure. Table 5 shows what is being implied for different values of the elasticity of the marginal utility of income function and for different ratios of income. Table 5 shows incomes differing by a factor of 2 and a factor of 20 (the latter is the ratio of GNP per capita in high income to low and medium income countries).

Table 5. **Equity weighting examples**

Elasticity of the marginal utility of income = e	-0.5	-1.0	-2.0	-5.0
Social value of high income Y1 relative to low income Y2 if Y1/Y2 = 2	0.7	0.5	0.25	0.03
Social value of Y1 if Y1/Y2 = 20	0.2	0.05	Neg	Neg

Note: the relevant formula is $w = (Y1/Y2)^{-e}$ where w is the weight on Y1.

What the Eyre *et al.*[20], and ExternE reports do is to select e = -1.0. For an income differential of 2 this would imply that we value the higher income group's marginal income as being just 50% of the value to the low income group. This seems potentially fair. But the income differential is actually 20, not 2, so that choosing -1.0 gives a weight of only 5% to income gains in the rich country. Many people might think this is also fair, but it is categorically not how we behave. If it was it would be impossible to explain why OECD countries devote far less than one per cent (on average) of their GDP on foreign aid and very much more than this on domestic life saving programmes.

The point here is not to assert that equity weighting is wrong. Indeed, not adopting equity weights amounts to choosing the equity weights implicit in the prevailing distribution of income. There is no escape from equity weighting! The issue is whether the studies adopting specific estimates for externality adders should adopt equity weights without (a) explaining why a specific set of weights is chosen, and (b) explaining why those weights are relevant to global warming but not to anything else.

5. “Disaster aversion”

A final issue that is very relevant to the externality adders literature is the treatment of disaster aversion. The idea here is that lives are at risk from energy sources. The usual procedure in the life cycle studies is to estimate accident rates and then value the resulting accidents at the relevant VOSL. In this case, use of “standard” VOSLs is probably correct because those at risk are the general population. It is widely thought, however, that the population is not indifferent between, say, 10 deaths in one accident and 10 deaths in 10 accidents each with one death. This is the notion of “disaster aversion” whereby the economic value attached to the former event would be higher than for the equivalent number of deaths in the ten accidents. The issue is obviously particularly relevant for nuclear power externalities, but it is also relevant for, say, gas explosions affecting the general public. The ExternE adder estimates do not in fact contain disaster aversion factors, but the ExternE background papers have discussed the issue quite extensively. While great ingenuity has been brought to bear on the kinds of aversion functions that might be specified, remarkably little empirical work appears to have been done to test whether people really are averse to disasters.

Ball and Floyd [7] reviewed studies of disaster aversion and concluded that “there is very little evidence for differential risk aversion by the public where this is based upon number of fatalities”. For example, Jones-Lee and Loomes [27] found that, in a transport context, the risk of large-scale accidents did not contribute to public willingness to pay for safety improvements. Ball and Floyd [7] also cited a study, Hubert *et al*. [24], which assessed the disaster aversion of senior managers in the petroleum, chemical and transport industries, as well as elected officials, and found that disaster aversion was significant in this group of people.

However, Slovic *et al*. [51], found that, in the nuclear context, the perceived risk was much greater than the actual risk, despite a perception of having the lowest annual number of fatalities compared to the other risk contexts studied. This discrepancy was assigned to the perceived potential for disaster. Slovic *et al*. [51,52] found that accidents also sent signals about the possibility and nature of future accidents, and asserted that, for nuclear accidents, these secondary impacts may be most important in this because the public perception of nuclear accidents is of poorly understood risks with potentially catastrophic impacts. A core damage accident may send ominous signals that the technology is out of control, even if the number of injuries or deaths was small, and that could be very damaging to the nuclear industry as a whole.

As it stands then, there is an urgent need to test for disaster aversion. It could have a substantial effect on externality adders for nuclear power and perhaps for natural gas. So far, however, we have little evidence to suggest that people are averse to collective deaths in the manner suggested by some of the theoretical literature.

6. Conclusions

Many other issues in the externality adder literature could have been addressed. Probably the biggest omission here is the extent to which we are justified in "borrowing" figures from other studies and using them in studies such as those by ExternE. This is the issue of "benefits transfer" and it is very much debated in the environmental economics literature. Major omissions in the adder studies relate to the absence of "meta-studies" even though some of the adder studies refer to meta-analysis. But it seems that what is usually meant is that the literature on damages has been surveyed. Proper meta-analysis involves statistical efforts to explain the variance in damage estimates and comparatively few of these exist.

What we can say is that, thanks to the substantial efforts of exercises such as ExternE, we are far better informed about externality adders than we were a few years ago. This is to be welcomed. The problem remains that the theoretical underpinnings are, in some cases, still weak.

Annex 1

Why the long run discount rate declines with time

The discount **rate**, r, needs to be distinguished from the discount factor, $1/(1+r)^t$. It is the discount factor that gives the weight applied to each time period. Suppose the discount rate and hence the discount factor is not known with certainty and is a random variable. Suppose it takes the values 1…6% each with a probability of 0.167. Table A1 shows the relevant values.

Table A1. **Values of the discount factor**

r	**DF_{10}**	**DF_{50}**	**DF_{100}**	**DF_{200}**
1	0.9053	0.6080	0.3697	0.1376
2	0.8203	0.3715	0.1380	0.0191
3	0.7441	0.2281	0.0520	0.0027
4	0.6756	0.1407	0.0198	0.0004
5	0.6139	0.0872	0.0076	0.0000
6	0.5584	0.0543	0.0029	0.0000
Sum	4.1376	1.4898	0.5900	0.1589
Sum/6	0.7196	0.2483	0.0983	0.0265
r*	**3.34%**	**2.82%**	**2.34%**	**1.83%**

Note: DF_{10} = discount factor for year 10, etc. r* is the value of r that solves the equation shown in the text.

While the weighted average (expected value) of the discount rate stays the same in all periods (3.5%), the discount factor obviously varies with time. The value of the implicit discount rate, r*, is given by the equation:

$$\frac{1}{(1+r^*)^t} = \frac{\sum DF_{t,i}}{n} \quad \text{………i = n}$$

where n = the number of possible discount rates, DF is the discount factor and t is time.

Table A1 shows that the implicit discount rate goes down over time even though the average discount rate stays the same for each period.

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Session 1

APPROACHES & ISSUES

(THEORY, CONCEPTS, DEFINITIONS, RELEVANCE TO DECISION MAKING)

THE EXTERNE PROJECT:
METHODOLOGY, OBJECTIVES AND LIMITATIONS

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Abstract

This paper presents a summary of recent studies on external costs of energy systems, in particular the ExternE (External Costs of Energy) Project of the European Commission. To evaluate the impact and damage cost of a pollutant, one needs to carry out an impact pathway analysis; this involves the calculation of increased pollutant concentrations in all affected regions due to an incremental emission (e.g. $\mu g/m^3$ of particles, using models of atmospheric dispersion and chemistry), followed by the calculation of physical impacts (e.g. number of cases of asthma due to these particles, using a dose-response function). The entire so-called fuel chain (or fuel cycle) is evaluated and compared on the basis of delivered end use energy. Even though the uncertainties are large, the results provide substantial evidence that the classical air pollutants (particles, NO_x and SO_x) from the combustion of fossil fuels impose a heavy toll, in addition to the cost of global warming. The external costs are especially large for coal; even for "good current technology" they may be comparable to the price of electricity. For natural gas the external costs are about a third to a half of coal. The external costs of nuclear are small compared to the price of electricity (at most a few %), and so are the external costs of most renewable energy systems.

1. Introduction

In recent years there has been much progress in the analysis of public health risks of energy systems, thanks to several major projects to evaluate the external costs of energy in the USA [8,15] and in Europe [3,4]. Of these, the ExternE (External Costs of Energy) Project of the European Commission has the widest scope and it is being continually updated to incorporate the latest scientific findings (external costs are damage costs imposed on others without compensation). This paper presents key results for damage costs and mortality risks due to the routine operation of the major energy technologies, as per ExternE [4].

To keep the presentation simple, we skip over much of the details and we show approximate results that are typical rather than precise for a specific site. We focus on mortality because it accounts for most of the damages (about 85% according to ExternE [4]) when they are weighted in monetary terms. Note that the damage cost estimates in the various publications are not always the same because the methodology has been evolving. In particular, there is a profound difference before and after 1996, due to the publication of a major study, by a team at Harvard University [9], of the cumulative impacts of air pollution on mortality, and due to a change in the valuation of air pollution mortality (loss of life expectancy, rather than number of deaths). The latter change is appropriate to permit meaningful comparisons between different causes of death with very different loss of life per death, in particular accidents and air pollution. Our paper is similar to Krewitt *et al.* [7], but places more emphasis on variability with site and technology, and on comparisons of nuclear with other fuel chains and with risks of everyday life.

2. Methodology

To evaluate the impact and damage cost of a pollutant, one needs to carry out an impact pathway analysis, tracing the passage of the pollutant from the place where it is emitted to the affected population. The principal steps of this analysis can be grouped as follows:

- Emission: specification of the relevant technologies and the environmental burdens they impose (e.g. kg of NO_x per TWh emitted by power plant).

- Dispersion: calculation of increased pollutant concentrations in all affected regions (e.g. incremental concentration of ozone, using models of atmospheric dispersion and chemistry for ozone formation due to NO_x).

- Impact: calculation of the dose from the increased exposure and calculation of impacts (damage in physical units) from this dose, using a dose-response function (e.g. number of cases of asthma due to this increase in ozone).

- Cost: the economic valuation of these impacts (e.g. multiplication by the cost of a case of asthma).

The impacts and costs are summed over all receptors of concern. For air pollutants (other than globally dispersing greenhouse gases) from fossil fuels the geographic range extends over thousands of km. The entire so-called fuel chain (or fuel cycle) is evaluated and compared on the basis of delivered end use energy. For the fossil fuel chains most of the damage cost arises from combustion in the power plant; for nuclear and for renewable energy, by contrast, the impacts upstream and/or downstream from the power plant are very important. The key assumptions of ExternE [4] are summarised in Table 1.

Since the results, other than for globally dispersing greenhouse gases, can vary strongly with the site where a pollutant is emitted, there is the question how to generalize site-specific results to representative numbers that may be needed for decision making. ExternE [4] has evaluated over fifty sites in different countries of the EU, and the results are sufficiently representative to permit general conclusions to be drawn [18,17]. Here we show results for central Europe, with an average (land and water) population density of 80 persons/km^2; for other regions the impacts of PM_{10} (particulate matter with <10µm diameter), SO_2 and NO_x need to be scaled in proportion to the population density within a radius of a thousand km around the pollution source.

The goal of the monetary valuation of damages is to account for all costs, market and non-market. For example, the valuation of a hospitalization should include not only the cost of the medical treatment but also the willingness to pay to avoid the suffering. If the willingness to pay for a non-market good has been determined correctly, it is like a price, consistent with prices paid for market goods. Economists have developed several tools for determining non-market costs. Of these tools contingent valuation has enjoyed increasing popularity in recent years, and the results of well-conducted studies are considered sufficiently reliable.

Table 1. **Key assumptions for the calculations in this paper [4]**

Atmospheric dispersion models	
Local range:	ISC gaussian plume model [1].
Regional range (Europe):	Harwell Trajectory Model [2] as implemented in ECOSENSE software of ExternE. Ozone impacts based on EMEP model [16], as interpreted by Rabl & Eyre [11].
Global warming	Physical impacts according to IPCC [6].
Impacts on health	
Form of dose-response functions[a)]	Linearity of incremental impact with an incremental dose for all health impacts.
Mortality	Dose-response function slope f_{DR} = 4.1 E-4 YOLL (years of life lost) per person per year per μg/m^3 derived from increase in all-cause age-specific mortality due to $PM_{2.5}$ [9], by integrating over age distribution.
Nitrate and sulfate aerosols[b)]	Dose response functions for nitrates same as for PM_{10}. Dose response functions for sulfates same as for $PM_{2.5}$ (slope = 1.7 times slope of PM_{10} functions).
Radionuclides	Linear dose-response functions: 0.05 fatal cancers/man·Sv, 0.12 nonfatal cancers/man·Sv, 0.01 severe effects hereditary /man·Sv.
Micropollutants	Only cancers have been quantified (As, Cd, Cr, Ni and dioxins).
Impacts on plants	Dose-response functions for crop loss due to SO_2 and ozone.
Impacts on buildings and materials	Corrosion and erosion due to SO_2 and soiling due to particles.
Impacts not quantified but potentially significant	Reduced visibility due to air pollution; disposal of residues from fossil fuels.
Economic valuation	
Valuation of premature death	Proportional to reduction of life expectancy, with value of a YOLL (years of life lost) derived from VSL = 3.1 M€: ^{v}YOLL = 0.083 M€ per YOLL (year of life lost)
Valuation of cancers	0.45 M€ nonfatal cancers, 1.5 to 2.5 M€ (depending on YOLL) fatal cancers, 1.5 M€ average for cancers from chemical carcinogens.
Discount rate	3% unless otherwise stated; Costs for nuclear are shown for 0% "effective discount rate" (=discount rate – escalation rate of cost).

a) For air pollutants the terms concentration-response or exposure-response function have also been used.

b) Nitrates and sulfates are secondary pollutants created from NO_x and SO_2 emissions, respectively, by chemical reactions in the atmosphere.

It turns out that damage costs of air pollution are dominated by mortality. The key parameter is the so-called value of statistical life VSL – an unfortunate term for what is really the willingness to pay for reducing the risk of premature death. In ExternE [4], a European-wide value of 3.1 M€ was chosen for VSL, close to similar studies in the USA. Unlike previous studies which simply multiplied the number of premature deaths by VSL, ExternE [4] bases the valuation on the years of life lost (YOLL). Since there had been no studies that determine the value of a life year directly, ExternE [4] calculated the value of a YOLL on theoretical grounds by considering VSL as the net present value of a series of discounted annual values. The ratio of VSL and the value of a YOLL thus obtained depends on discount rate and latency; it is typically in the range of 20 to 40. Here the value of a YOLL due to air pollution is taken as 0.083 M€.

3. Damage cost per kg of pollutant

Before proceeding to fuel chain results, we present the damage costs per kg of pollutant for typical European conditions in Table 2. Some indication of the variability with site and stack conditions is given in the notes under the table.

The uncertainties are large. Rabl & Spadaro [13] have analysed them in terms of lognormal distributions which are appropriate because the calculation of damage costs is essentially multiplicative. The uncertainties can be expressed as geometric standard deviation σ_g, interpreted in terms of multiplicative confidence intervals of the lognormal distri

bution: if a cost has been estimated to be μ_g (geometric mean ≈ median) with geometric standard deviation σ_g, the probability is approximately 68% that the true value is in the interval $[\mu_g/\sigma_g, \mu_g.\sigma_g]$ and 95% that it is in $[\mu_g/\sigma_g^2, \mu_g.\sigma_g^2]$. For the damage costs of PM_{10}, NO_x and SO_2, Rabl and Spadaro [13] estimate that σ_g is in the range of 3 to 4.

The largest sources of uncertainty lie in the epidemiology and in the "value of statistical life" VSL. While ExternE [4] assumes VSL = 3.1 M€, there is no general consensus and one could argue for other values in the range of 1 to 5 M€. There is also considerable uncertainty about the relation between VSL (which has been determined for accidents) and the value of a YOLL due to air pollution, because it involves the period at the end of life about which valuation studies are only just beginning.

The uncertainty due to the monetary valuation of a YOLL can be avoided by comparing the risks of technologies directly on the basis of YOLL per kWh. For this purpose we convert the €/kg numbers for PM_{10}, NO_x and SO_2 to

YOLL/kg by assuming as an approximate relation that 85% of the respective damage costs is due to mortality, valued at 83 000 €/YOLL. We do not indicate the YOLL due to global warming because that information is difficult to extract from the numbers in ExternE [4]. We note, however, that a large part of the damage cost estimate of 0.029 €/kg reflects loss of life due to the expected increase in tropical disease.

Table 2. **Typical damage costs per kg of pollutant emitted by power plants in Europe [18,17]**

Pollutant	**Impact**	**Cost**[a)]**,** €/kg
PM_{10} (primary)	mortality and morbidity	15.4
SO_2 (primary)	crops, materials	0.3
SO_2 (primary)	mortality and morbidity	0.3
SO_2 (via sulfates)	mortality and morbidity	9.95
NO_2 (primary)	mortality and morbidity	small
NO_2 (via nitrates)	mortality and morbidity	14.5
NO_2 (via O_3)	crops	0.35
NO_2 (via O_3)	mortality and morbidity	1.15
VOC (via O_3)	crops	0.2
VOC (via O_3)	mortality and morbidity	0.7
CO (primary)	morbidity	0.002
As (primary)	cancer	171
Cd (primary)	cancer	20.9
Cr (primary)	cancer	140
Ni (primary)	cancer	2.87
Dioxins, TEQ	cancer	1.85×10^7
CO_2	Global warming	0.029

a) Variation with site and stack conditions (stack height, exhaust temperature, exhaust velocity):
- No variation for CO_2.
- Weak variation for dioxin (non-inhalation pathways): factor of ≈ 0.7 to 1.5.
- Weak variation for secondary pollutants: factor of ≈ 0.5 to 2.0.
- Strong variation for primary pollutants: factor of ≈ 0.5 to 5 for site, ≈ 0.6 to 3 for stack conditions (up to 15 for ground level emissions in big city).

4. Results for fuel chains

Multiplying the cost per kg by the emission rates, Table 3, one readily finds the cost per kWh. However, we warn against the temptation to cite cost/kWh or

YOLL/kWh numbers out of context, for instance "the health risks of coal". Quite apart from the variation of impacts with the site of an installation, the very term "fuel chain" is misleading, because it suggests a simple monolithic system while the reality is a complex chain whose elements can consist of a variety of different processes and technologies at different sites, emitting very different rates of pollution. Furthermore, thanks to ever more stringent environmental regulations there has been a continual reduction of specific emissions (a factor of 3 to 10 during the past decade, except for CO_2 for which the reductions have been smaller).

Table 3. **Emission of air pollutants and resulting years of life lost YOLL/TWh for fossil power plants, for typical European conditions**

	Emission, g/kWh			YOLL/TWh			
	PM_{10}	SO_2	NO_x	PM_{10}	SO_2	NO_x	Total
Coal, current	0.15	6	3	23.3	601.8	439.9	1 065
Coal, new	0.06	0.30	0.50	9.3	30.1	73.3	113
Oil, current	0.15	6	1.4	23.3	601.8	205.3	830
Oil, new	0.07	0.40	0.60	10.9	40.1	88.0	139
Gas, current	negligible	Small[a)]	1.1	0.0	0.0	161.3	161
Gas, new	negligible	Small[a)]	0.2	0.0	0.0	29.3	29

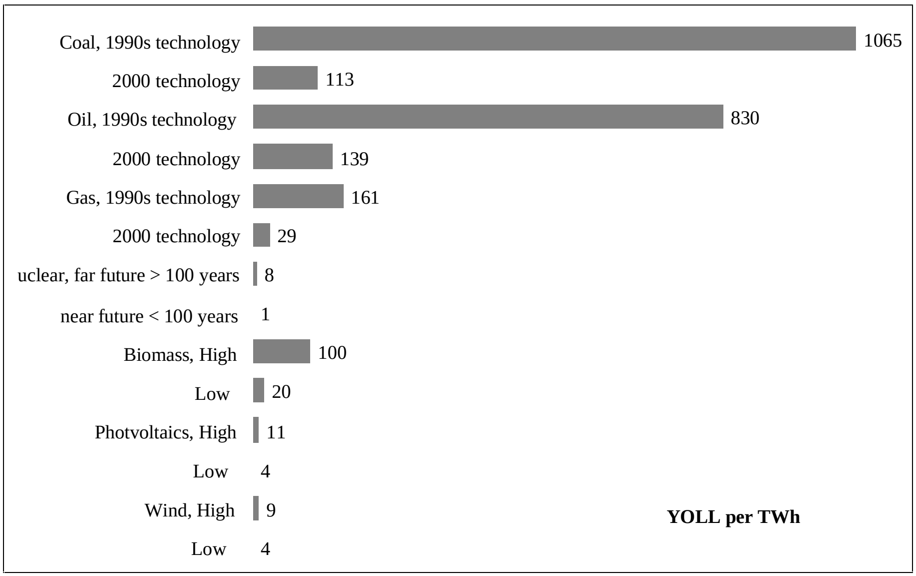
a) SO_2 emission depends on composition of natural gas; in most cases it is negligible.

Note: "Current" = with typical emissions of existing fossil plants in USA and France in 1995; "new" = with estimated emissions of large new power plants built in USA since January 2000.

To illustrate this point we list in Table 3 the measured emissions of PM_{10}, SO_2 and NO_x, for typical fossil plants in USA and France during the nineties, as well as estimated emissions for large new plants built in the EU after January 2000. For fossil fuel chains we show only emissions and YOLL due to the power plant, rather than for the entire fuel chain, because they account for almost the entire health impact; the YOLL due to emission of toxic metals is negligible, the emitted quantities being extremely small. For the renewables, by contrast, the health impacts arise mostly or exclusively from the production of the materials for the power plant. Results are plotted in Figure 1. The number for nuclear is based on the current technology of France, including reprocessing [3]; it comprises the impacts over the entire globe and a time horizon of 100 000 years. Assessments for other countries have found roughly comparable results [8,15,4].

Damage costs are plotted in Figure 2, showing separately the costs due to the classical air pollutants (PM_{10}, NO_x and SO_2), due to global warming (including upstream emissions of CO_2 and CH_4, expressed as CO_2 equivalent), and due to radiation. It is interesting to note that the damage costs for existing oil and coal fired power plants are very large, larger than the production costs of electricity, about 25 to 50 m€/kWh. The damage costs for nuclear correspond to zero effective discount rate (= discount rate – escalation rate of costs). Because of the uncertainties and controversies surrounding intergenerational discounting, we show the impacts and costs separately for the near (before 100 yr) and far (after 100 yr) future.

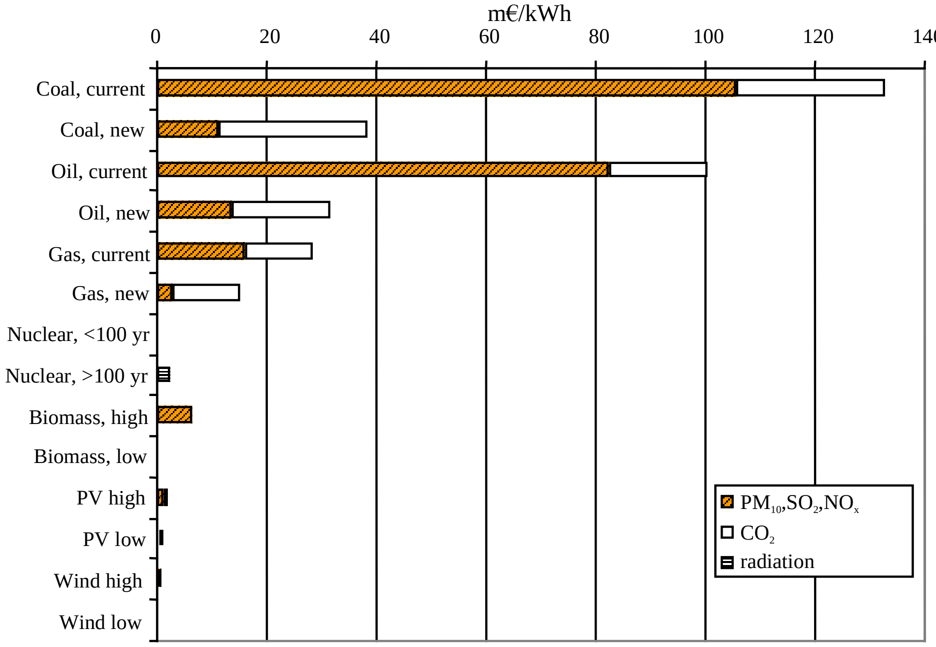
Figure 1. **Comparison of public health risks in terms of YOLL/TWh, for fuel chains in the EU, with emissions of Table 3**



Note: "High" and "low" for renewables indicate typical range of estimates of ExternE [4]. This graph includes only risks due to PM_{10}, SO_2, NO_x and radiation. For nuclear only a single technology is shown (French, with reprocessing), but YOLL are separated into near (before 100 yr) and far (after 100 yr) future.

For nuclear the numbers cover all stages of the fuel cycle, and include waste disposal and major accidents – although any estimate of the latter items is controversial. All of the damage cost of low-level radiation is due to human cancers and hereditary effects, whereas environmental impacts are negligible. Emissions of conventional pollutants by the nuclear fuel chain are negligible, if one allocates energy use upstream and downstream of the power plant to nuclear (as appropriate for an assessment of nuclear as base load electricity source for the future).

Figure 2. **Comparison of damage costs, for fuel chains in the EU, with €/kg of Table 2 and emissions of Table 3**



Note: "High" and "low" for renewables indicate typical range of estimates of ExternE [4]. For nuclear only a single technology is shown (French, with reprocessing), but costs are separated into near (before 100 yr) and far (after 100 yr) future. Note that production cost of base load electricity in EU and USA is in the range of 25 to 50 m€/kWh.

All this assumes, of course, a mature and stable political system, with strict verification of compliance with all regulations. Low external costs do not suffice to allay concerns about accidents, long lived radioactive waste, the right to impose impacts on future generations, and risks from terrorists and rogue governments; these issues involve acceptability and defy quantification in terms of external costs. The following simple example can illustrate why external costs are not the only decision criterion. Suppose someone invents an energy system that can supply the world's electricity (roughly 10^{13} kWh/yr) at the bargain price of \$0.075/kWh (about one tenth of typical current prices) – with one little catch: there is a probability of an accident occurring once every 100 years that will kill 25 000 (roughly the total expected toll for Chernobyl if one multiplies the UNSCEAR estimate of committed dose by the standard linear dose-response function).

Even at \$3 million/life and zero discount rate, the levelized cost of such an accident has an expectation value of only

$$2.5 \times 10^{4} \times 3 \times 10^{6}\ \$/(100\ \text{yr} \times 10^{13}\ \text{kWh/yr}) = 0.75 \times 10^{-4}\ \$/\text{kWh}$$

a mere 1% of this low electricity price. But would people accept such a deal?

Impacts of solid hazardous wastes are impossible to predict to the extent that they depend on future waste management decisions; the best one can do is evaluate scenarios. In principle such impacts can be kept negligible by storing wastes in well-managed leak proof facilities. But will the integrity of the containers and liners be maintained forever? In case of a leak the most likely occurrence is leaching into the ground water, and the impacts tend to be limited to the local zone and could be stopped or corrected, if appropriate measures are taken. Technologies for alternative methods of solid waste disposal are evolving; for example, coal ash is increasingly used as additive in building materials. Fly ash can be stabilized in concrete or glass. For coal none of the externality studies have succeeded in quantifying physical risks from solid wastes.

For nuclear power, the volume of waste is very small compared to that from coal but the toxicity is high. Until recently the predominant view was that nuclear wastes should be deposited in deep geological sites, to be sealed permanently. Based on certain assumed scenarios for accidental intrusion or containment failure, similar to other studies, ExternE [3] estimated the contribution of nuclear waste storage to be about 1% of the total damage cost.

However, in view of all the doubts that have arisen about permanently sealed geological sites, it may be wiser to envisage storage in depositories that are guarded and retrievable for the indefinite future (note that the cost, even over an infinite time horizon, is not excessive, even with low intergenerational discount rates [10]). After all, whatever the uncertainties for the future, in the long run technological progress is the most likely scenario, and future generations will have better means of dealing with radioactive waste, provided it is retrievable. Is the idea of leaving wastes for future generations really so shocking, if the alternative is to leave other wastes that are likely to impose even higher costs? The greenhouse gases from the use of fossil fuels pose a threat the severity of which is increasingly recognized. And as for long lived solid wastes, one should not forget that coal ash contains appreciable quantities of toxic metals, with half life much longer than radioactive waste, not to mention the natural radioactivity of coal. In view of this situation we believe that the existing assessments of waste disposal are not very satisfying and that a new set of comparative studies is needed to put the options for nuclear and for fossil fuels in perspective.

6. Risk comparisons

Since most of the health risk from nuclear power is imposed on people living in the far future, a simple number for the global YOLL is not very instructive. For another perspective let us look at the implications of using nuclear power on a large scale. The population of the world is about 6 billion and the electricity consumption 12 000 TWh/yr. Both will increase, especially the latter, although saturation can be expected eventually. Technologies will certainly evolve and nuclear fission reactors are unlikely to be more than a stopgap, perhaps for a century or so, until cleaner sources of energy mature. Since details do not matter for the following argument, we take simple round numbers.

Let us suppose a simple "100 year scenario" where the above nuclear fuel cycle is used for 100 years to produce 2×10^4 TWh/yr, for a world population of 10^{10}. The total public dose for the French nuclear fuel chain is 12.5 person·Sv/TWh; most of this dose is imposed globally rather than being limited to a local zone. This dose would imply a dose rate of

$$\text{Total dose rate} = 12.5 \text{ person·Sv/TWh} \times 2 \times 10^4 \text{ TWh/yr}/10^{10} \text{ persons} = 25\ \mu\text{Sv/yr}$$

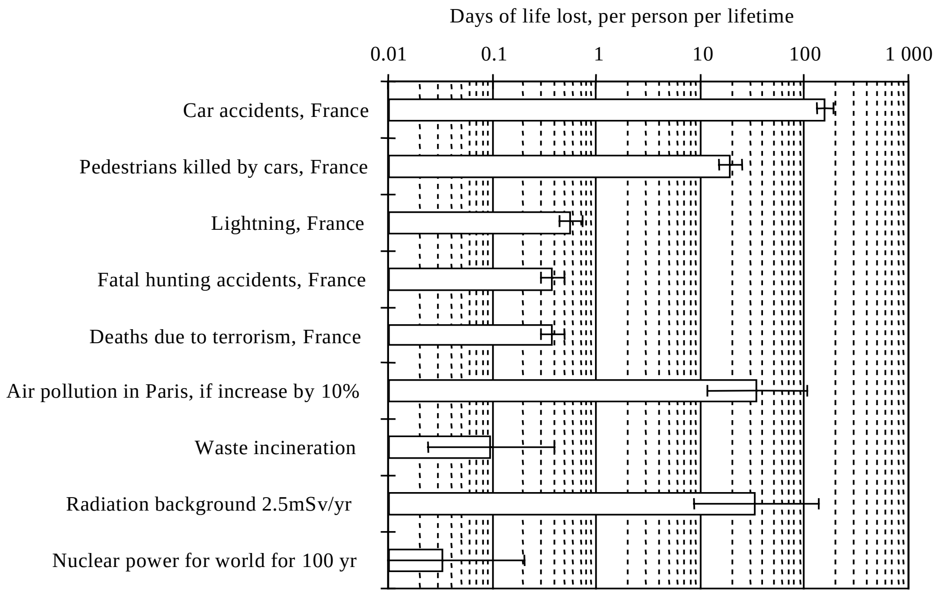
if the entire dose were incurred immediately. However, only ten percent of the dose from the production of electricity is incurred during the first 100 years. The precise time distribution of the total dose rate from the "100 year scenario" would be difficult to calculate, because of the large number of different half lives. But again a rough order of magnitude estimate is sufficient, and we simply take 10% of the total dose rate to be imposed on the population living during this "100 year scenario"

$$\text{Total dose rate first 100 years} = 2.5\ \mu\text{Sv/yr}.$$

This can be compared to background radiation from other sources that people are exposed to. A typical value is in the range of 2 to 3 mSv/yr, including medical x-rays and an average value for radon in buildings. Another interesting comparison is with the dose rate due to cosmic radiation at sea level, 260 μSv/yr, because this is the very minimum all of us are exposed to and protection from which is not practical. Thus the "100 year scenario" would increase the background exposure by, very roughly, 1% of the cosmic ray background at sea level. Generations living beyond those 100 years would of course also be exposed, but at a lower rate.

In Figure 3 we compare on a logarithmic scale the YOLL due to fatal cancers from the 2.5 μSv/yr dose rate of this "100 year scenario" with several other risks of everyday life (to get numbers that are easier to comprehend, we convert to days of life lost). Also included is the mortality risk due to another potentially significant source of renewable energy, namely waste incineration, for a scenario where all the municipal solid waste (500 kg/yr per person) is incinerated with emissions equal to the latest Directive of the EU, effective after 2001. The risks from these two controversial technologies, waste incineration and nuclear power, are very small.

Figure 3. **Comparison of some public health risks**



Note: Scenario for waste assumes all municipal solid waste is incinerated, with emissions equal to the latest EU regulation. Scenario for nuclear assumes the entire current world consumption of electricity is supplied for 100 years by the current technology of France. Data for the first 5 items are from Frémy & Frémy. [5], for air pollution in Paris from Rabl [14] and for waste incineration from Rabl *et al*. [12]. Error bars express uncertainty as estimated by Rabl & Spadaro [13].

7. Conclusion

Based on the results of the ExternE Project we have compared the public health risks and the damage costs from the routine operation of energy systems. Generally the loss of life expectancy per kWh of electricity is much higher for fossil fuels than for modern nuclear plants and for most renewables, and among fossil fuels it is much higher for coal than for gas. However, for each energy source there is much variation between different installations of a given energy source; for example, the emission of harmful pollutants from coal power plants built after January 2000 will be an order of magnitude lower than typical emissions from existing plants in the USA and EU during the nineties.

The health effects of PV (photovoltaics) and wind arise from the production of the materials rather than during operation of the plant. Among the renewables only biomass combustion entails relatively large health risks, due to the emission of PM_{10}, SO_2 and NO_x. The health risks from the routine operation of the nuclear fuel chain are very small, even though they have been summed over a time horizon of 100 000 years. Since the meaning of risks imposed over thousands of years is difficult to comprehend, we have illustrated them by considering a scenario where the entire electricity of the world for 100 years is produced by nuclear: the incremental collective dose is only one percent of the cosmic radiation background.

Even though the uncertainties are large (the numbers could be about 4 times smaller or larger), they have relatively little effect on the comparison between energy systems, except for nuclear, because they affect the impacts of the air pollutants in more or less the same way.

Acknowledgments

This work has been supported in part by the ExternE Project of the European Commission DG Research.

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Session 2

EXTERNAL COSTS OF ENERGY/ ELECTRICITY LIFE CYCLES

(RESULTS FROM RECENT AUTHORITATIVE STUDIES, LESSONS LEARNED, UNCERTAINTIES, GAPS)

A LIFE CYCLE PERSPECTIVE OF COAL USE

Louis Wibberley
BHP Sustainable Technology, Wallsend (Australia)

Context

- **Background**
  - negativity towards coal has been based on superficial “burner tip” comparisons - a misleading basis for comparison
  - policy dilemma - how to meet the development needs of the world in a sustainable, affordable manner
    - coal is expected to underpin future energy demand in many parts of the World (large reserves, diversity of supply, stability of price, ease of storage)
    - reduced proportion of the total energy is expected, but the overall consumption will increase
    - renewables need a base load energy source
- **Australia has 2 related initiatives for understanding and improving the role of coal in the transition to a more sustainable society**
  - a life cycle analysis of steel and electricity production for a range of energy sources and technologies (Australian Coal Association, WCI and others)
  - Cooperative Research Centre for Coal in Sustainable Development (CCSD)
    - a 7 year program involving industry, Government and research organisations
    - fundamentals, technology and holistic analyses of most aspects of coal use

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Approach - LCA component

- **To carry out streamlined life cycle analyses for a wide range of steelmaking and electricity process chains**
  - cradle-to-gate inventory analysis
    - limited to resource energy, GGE, NO_X, SO_X, particulates, solid waste and water
    - using (initially) Australian data
  - focus on energy/technology comparisons, improvement opportunities, detailed understanding of the production chain
- **To use streamlined impact assessment**
  - aggregation limited to GGE (taken as the main proxy for impact)
  - restricted to environmental burdens, no economics assigned
- **LCA chosen to help focus on assessment of improvement options**
- **Broader framework for assessment of socio-economics is being considered in the CRC program**

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ExternE values - coal based electricity in Europe (mECU/kWh)

Country	YOLL				Global Warming GHG Mid 3%	All Other	Total	Comments
	SOx	NOx	Other	Total				
U.K.	6.1	10.5	2.9	19.5	15	7.5	42	U.K Deep mine PF ESP, FGD Low Nox No SCR
Germany	2.9	6.3	1.2	10.4	14.3	5	29.7	DENOX FGD
Sweden	0.079	0.366	0.036	0.481	13.2	4.419	18.1	SCR for NOx > 90% reduction FGD for SOx > 88% reduction Electric filter for PM > 99.9% reduction Co generation

YOLL = Years of life lost converted to economic terms

* **Other (**= includes morbidity costs of TSP, SOx & NOx, and accidents (accidents minor contributor)

All Other = cost of impacting crops, ecosystems, materials, noise, aquatic systems & aesthetics

*__Mid 3% GHG__: A discount rate is applied to future impacts of global warming events

Reference: EUR 18528 – ExternE- Externalities of Energy Vol. 10 National Implementation

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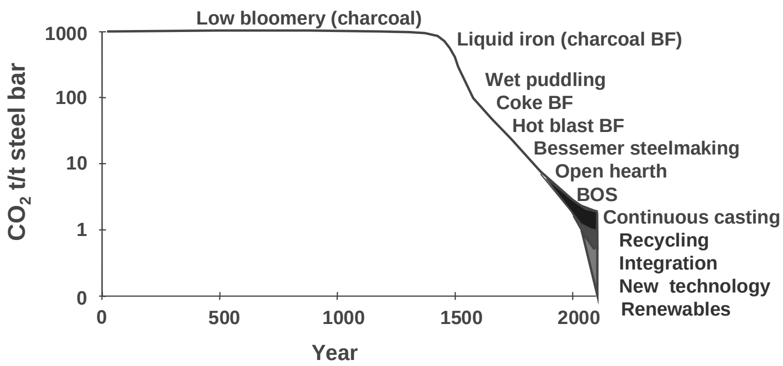
Historical perspective - iron & steelmaking

Impressive process improvements have been made by the steel industry over time, by both breakthrough and incremental technology development



Steel GGE (t CO_2-e/t cast steel)

- **Why is a systems or holistic approach so important?**

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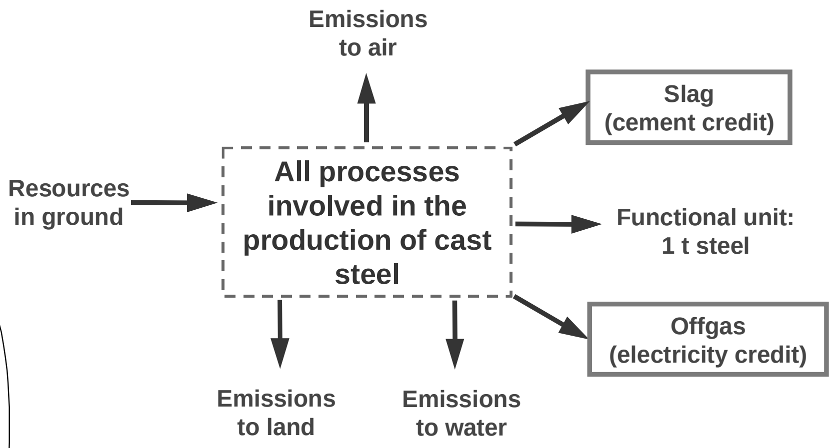
LCA system



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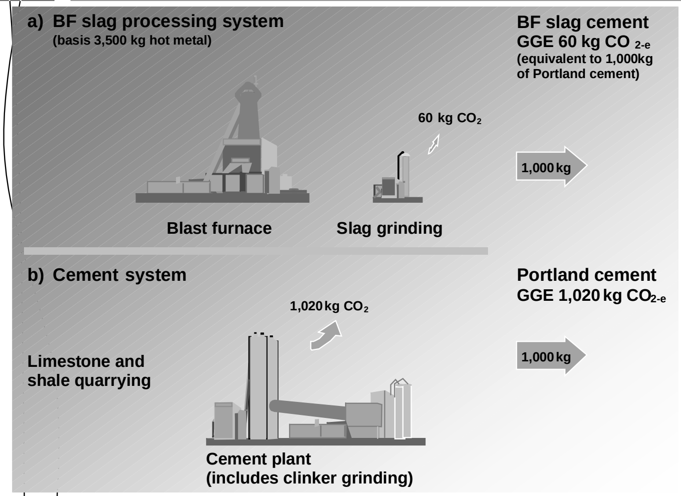
Displacement credits - slags

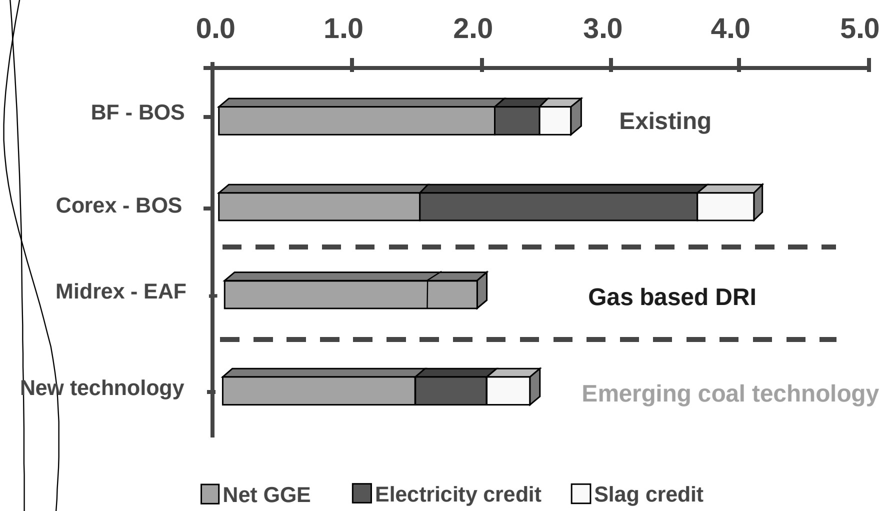


No technical or economic issues

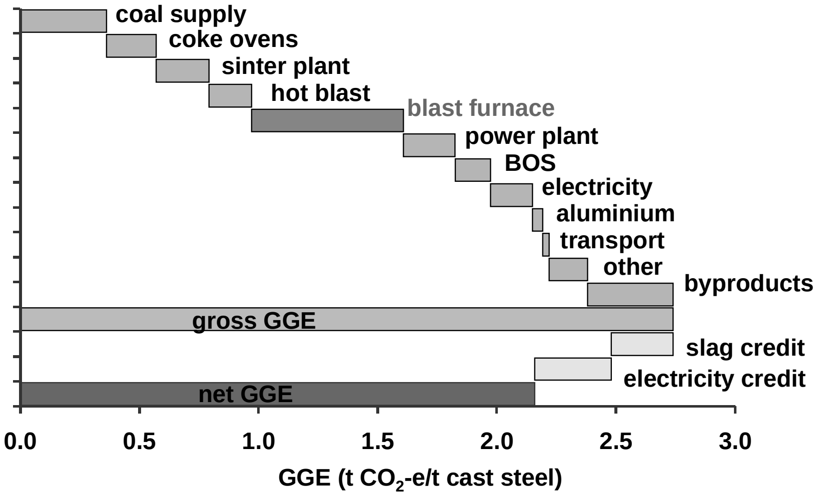
Often limited by attitudes

A product stewardship issue for both coal and steel?

Steel GGE (t CO_2-e/t cast steel)



Blast furnace is only one source of GGE



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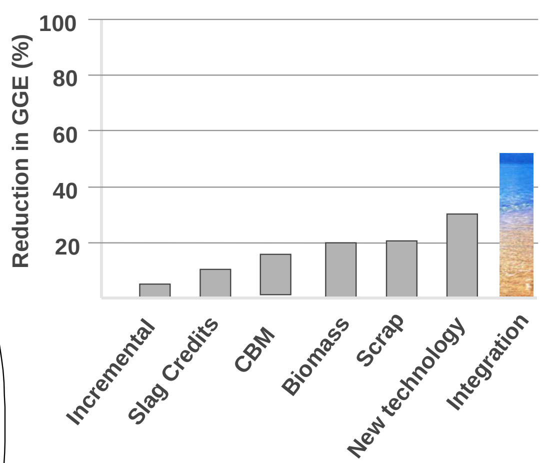
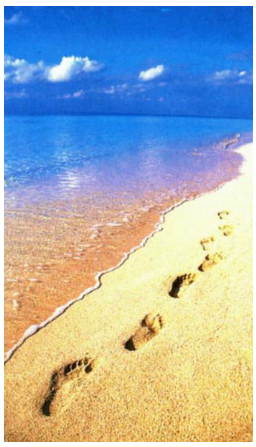
Improvement opportunities





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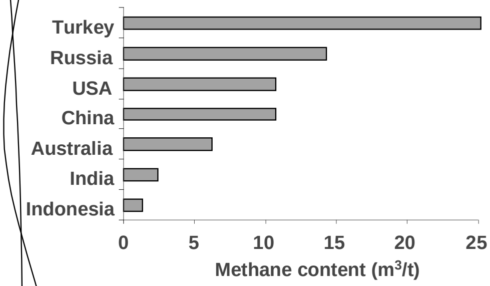
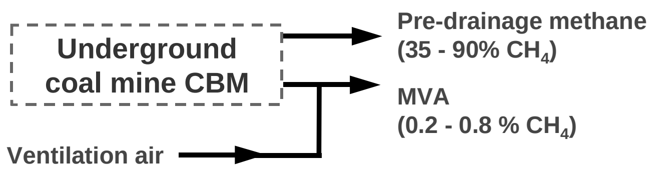
Coal bed methane (CBM)



- World total 30Mtpa?
- only 5% utilisation?
- ~50% as MVA for underground mines
- biggest GGE benefit from oxidation, power gives small additional benefit



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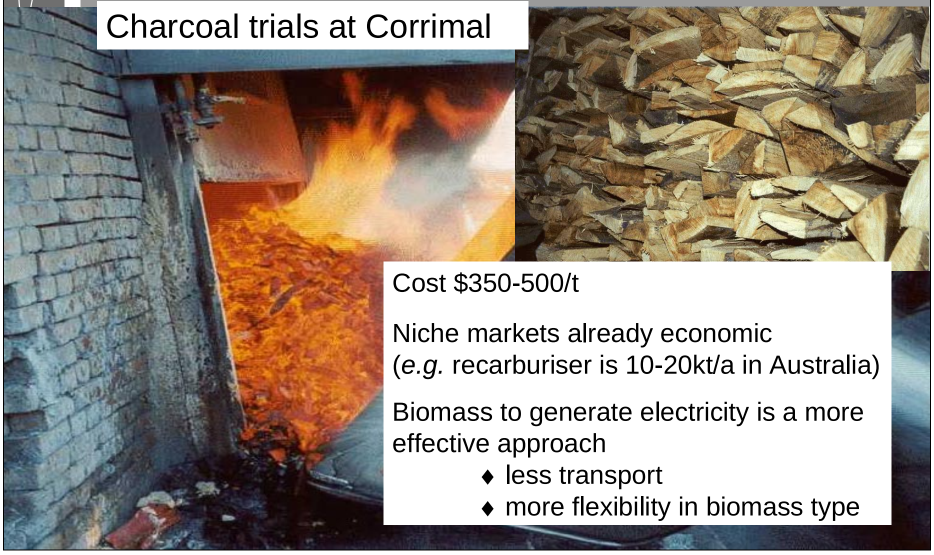
CBM utilisation at Appin and Tower

- 94MW_e using 1MW_e gas engines
- 160kt/a CH_4 utilised
  (pre-drainage gas, some MVA used for combustion)
- 3Mt CO_2-e avoided annually

Charcoal - limited applicability …

Charcoal trials at Corrimal



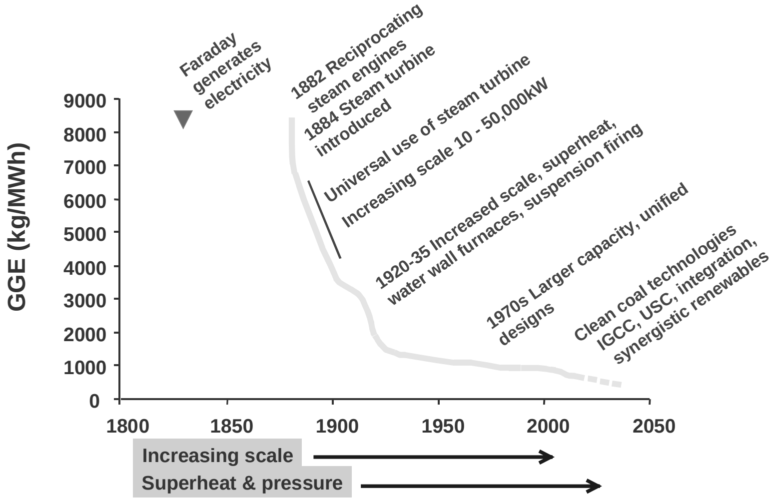
Cost $350-500/t

Niche markets already economic
(*e.g.* recarburiser is 10-20kt/a in Australia)

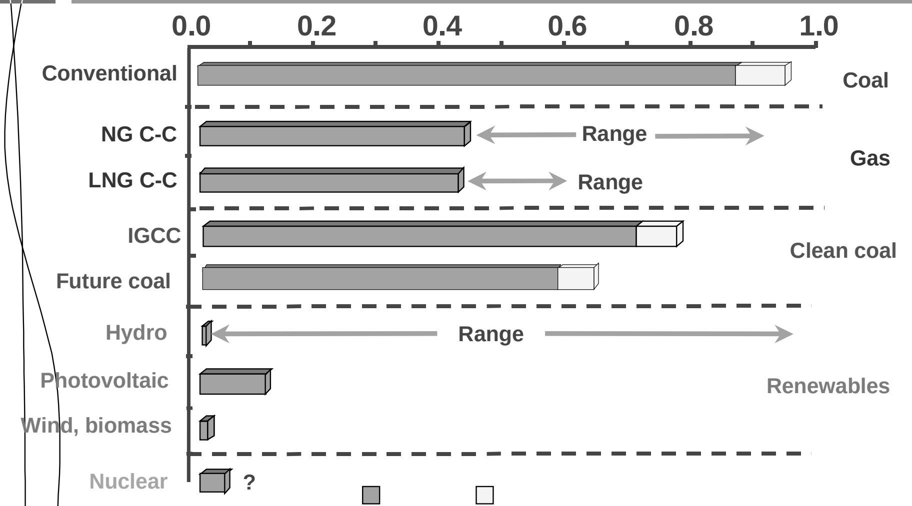
Biomass to generate electricity is a more effective approach

- ♦ less transport
- ♦ more flexibility in biomass type

Historical perspective – power generation



Electricity GGE (t CO_2-e/MWh)



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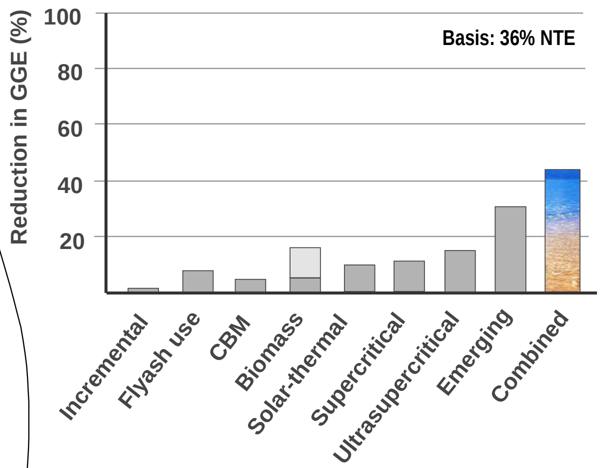
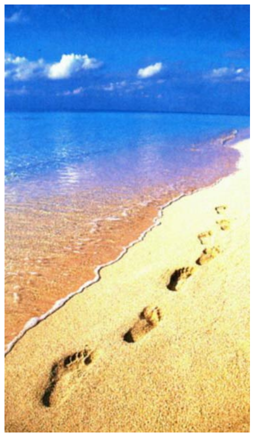
Improvement opportunities





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Reduction options

Option	Change in efficiency*	GGE reduction (%)
Incremental improvements	36 ⟶ 38%	5
Replacement		
Old coal with new	26 ⟶ 40%	25
Supercritical pf	36 ⟶ 40%	10
Ultrasupercritical pf (now)	36 ⟶ 42%	15
Ultrasupercritical pf (future)	36 ⟶ 50%	30
Emerging IGCC etc	36 ⟶ 50%	30
Flyash to cement		5-7
Biomass-coal		5-15
Solar-coal		10

** gross, sent-out*

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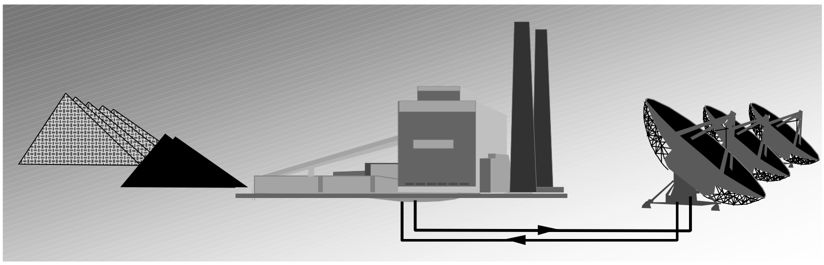
Synergies with renewables



Biomass co-firing

35% biomass conversion efficiency (20% for dedicated)

Solar thermal

30-40% solar conversion efficiency (13% for PV)

Coal can promote uptake and efficient utilisation of renewables
Coupling of renewables and fossil energy research is essential

Biomass-coal generation

Partnership** with a **Future:

Coal** and **Bagasse
*Dual-Fired Belle Vue **Power Plant***
in Mauritius

- **Guadeloupe, Reunion and Mauritius have installed 6 x 70 MWe dual fuel power stations:**
  - **bagasse (6 month season)**
  - **coal (when bagasse unavailable)**
- **Provide electricity throughout year, while maximising use of renewable energy (biomass)**
  - **economic and social benefits**
  - **enable more efficient plants to be built**

Source: Good News from Coal, WCI, Nov 1999

Solar-coal generation?

- **Several technologies have been proposed**
  - 130 MWe per km^2
- **Lowest cost routes to solar electricity**
  - A$80/MWh @ 100 MWe
- **Demonstration plant of 3 MWe (av) under consideration**

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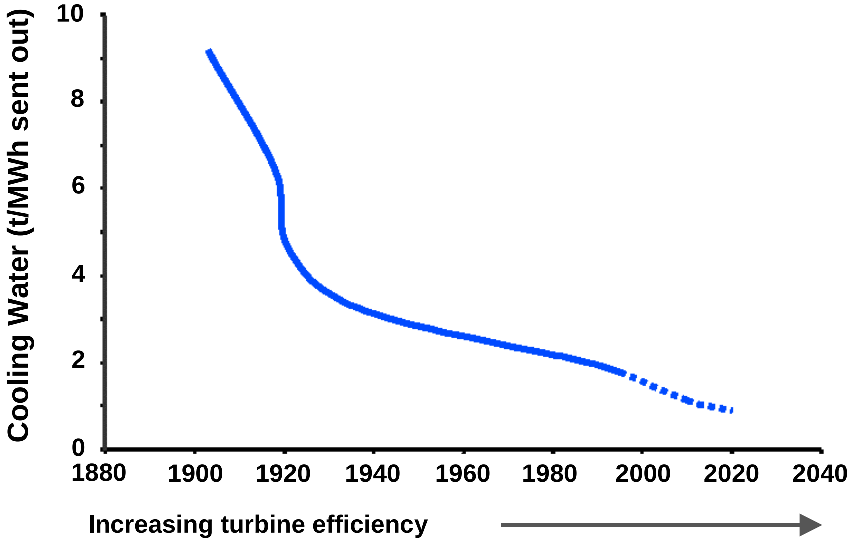
Power generation – water use



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Power generation - water use

- **Water consumption for power generation depends upon the cooling technology used and the efficiency of the conversion of steam to electricity in the turbine**
- **Majuba power station in South Africa**

	Water consumption (m3/MWh)	Efficiency (%)
Units 1-3 (dry cooling)	**0.2-0.4**	**~33**
Units 4-6 (wet cooling)	**2.0-2.5**	**~37**

Source: African Energy Vol. 1, No.3, 1999

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Final remarks

- **Streamlined LCA has provided improved data and understanding of issues for coal and alternatives**
  - for GGE, the difference between coal and alternatives has been shown to be less than previously reported
  - still many information gaps
- **Many opportunities for improvement throughout the coal chain - from coal in the ground, through to waste disposal**
  - slags and flyash are not used to maximum benefit
  - oxidation/utilisation of CBM
  - substantial improvements are available through incremental changes to “conventional pf” technologies (in addition to the new technologies)
- **Coal can underpin the introduction of renewables**
  - higher overall efficiency (and more cost effective than stand alone options)
- **Further studies are underway**
  - combining LCA, technology evaluations, socio-economics

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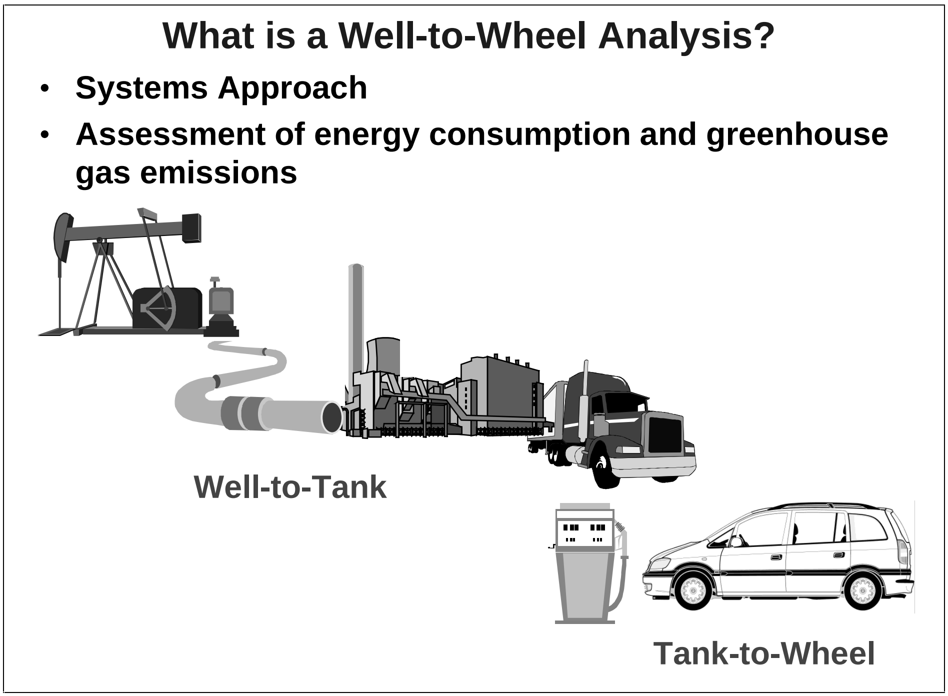
WELL-TO-WHEEL ENERGY ANALYSIS STUDY

Jean Cadu
Shell, London (United-Kingdom)

(http://www.transportation.anl.gov)



Why Do We Need Well-Tank-Wheel Analysis?

- Evaluate emerging propulsion technologies
  - Advanced Internal Combustion Engine (ICE)
  - Hybrid Electric Vehicles (HEV)
  - Fuel Cell Vehicle (FCV)
- Evaluate new fuels
- Aid public policy development and business strategy

Background

Study commissioned by GM

- Well-to-Tank work done by Argonne National Labs with input from BP, ExxonMobil, and Shell
- Tank-to-Wheel work performed by GM

- Considered 75 fuel "pathways" and 15 advanced and conventional powertrain systems

Well-to-Wheel Integration Process

Well-to-Tank

Argonne National Laboratory
GREET Model

75 Fuel Pathways
- Input data provided by Argonne, BP, Exxon Mobil, Shell

Tank-to-Wheel

General Motors
Vehicle Simulation Model

15 Propulsion Systems
- Input data provided by GM

Well-to-Wheels

General Motors
Well-to-Wheels Integration Model

27 Selected Fuel/Propulsion Pathways

14 Pathways Reviewed Today

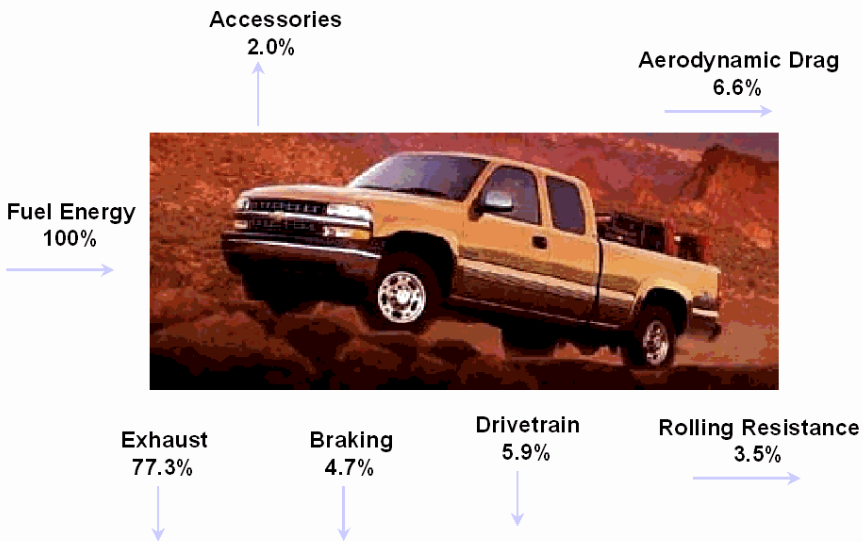
What is Different about this Well-To-Wheel Analysis?

- Uses multiple sources of information: National Labs, energy companies, and GM
- Provides range (reflecting uncertainties) as well as best estimates
- Multiple fuel pathways
- GM proprietary simulation of propulsion systems with full mapping
- True "apples-to-apples" comparison for alternative propulsion systems

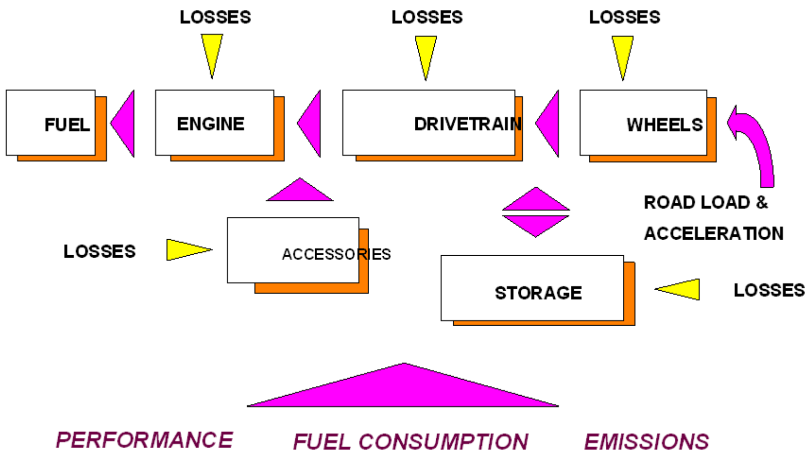
Approach for Tank-to-Wheel Analysis

- Technologies considered are targeted for 2005-2010 time frame
- Selected target vehicle class is a GM full-size pickup truck
- Validated component characteristics used in vehicle simulation models for predicting vehicle fuel economy and performance
- Performance criteria based on stringent US consumer demands
- Engine and motor sized to meet performance requirements
- Emission targets based Federal Tier 2 levels and penalty to meet targets were included in fuel economy predictions
- Fuel economy maximized with energy management and control strategies for each powertrain architecture
- Caveats in this study:
  - vehicle mass estimation needs to be addressed in greater detail based on higher fidelity of component masses
  - cold-start performance and transient responses not included in any concepts shown
  - packaging and cost not considered

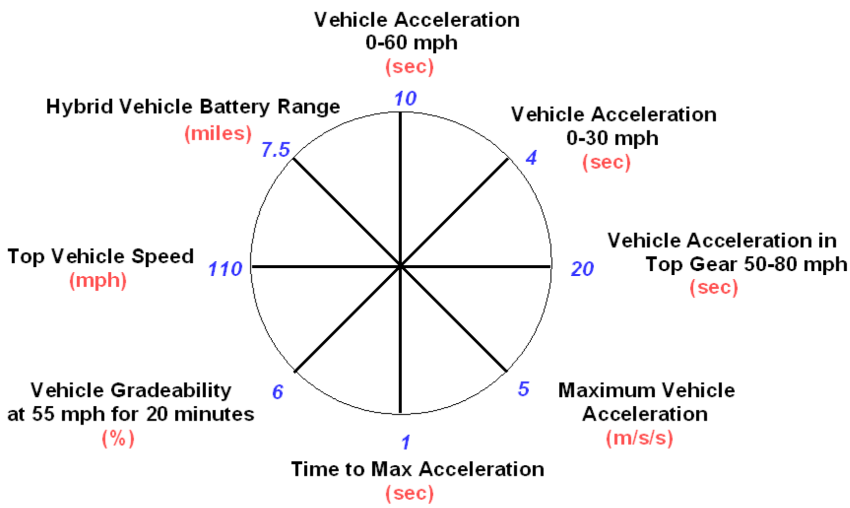
Energy Utilization in Pickup Truck

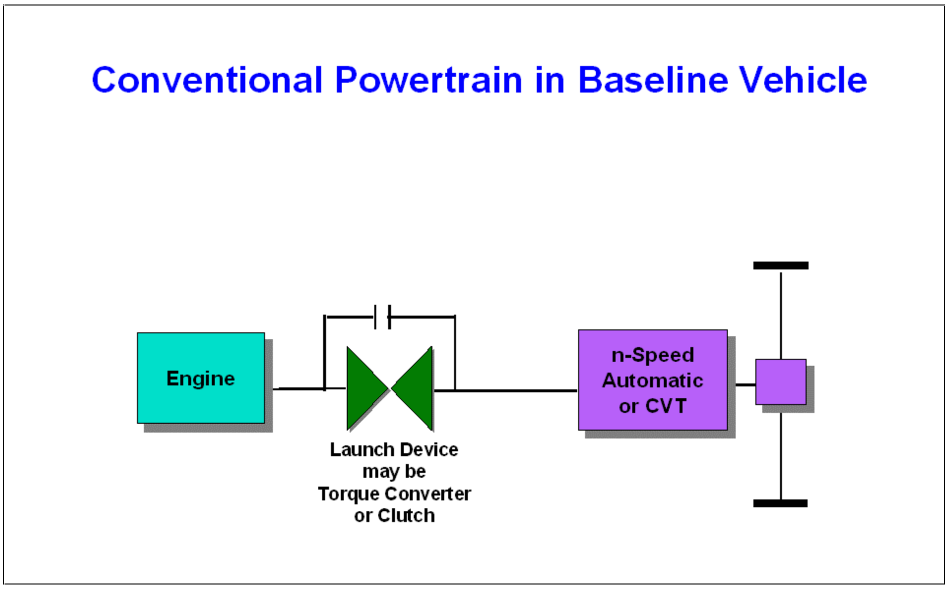


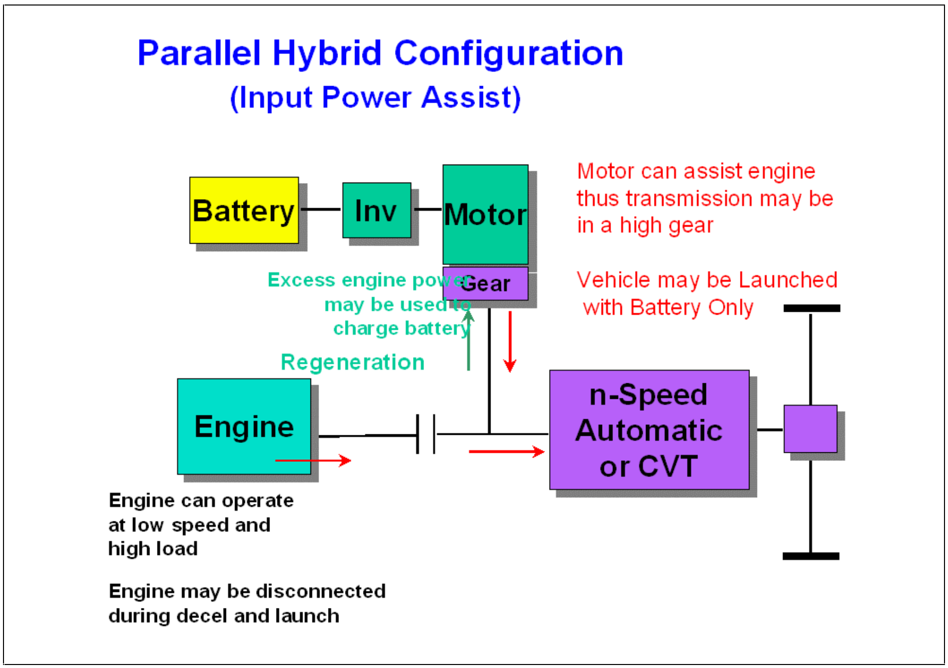
Vehicle Simulation Model



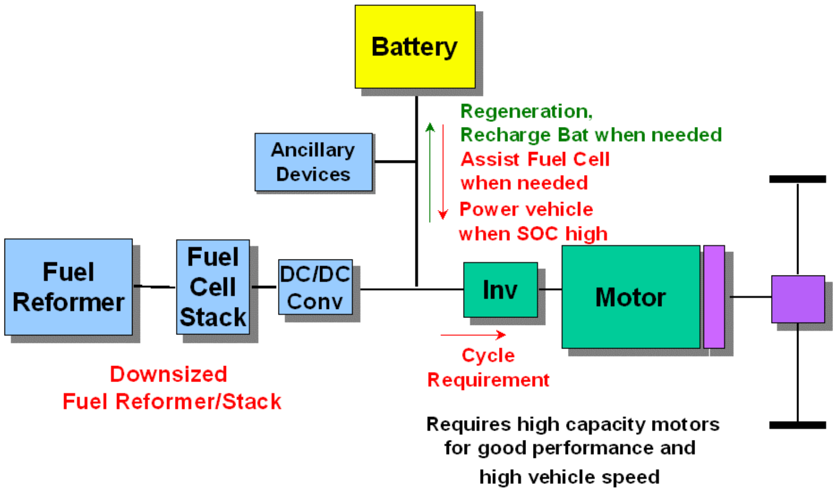
Performance Targets for Tank-to-Wheel Study




Conventional Powertrain in Baseline Vehicle
Engine
Launch Device
may be
Torque Converter
or Clutch
n-Speed
Automatic
or CVT

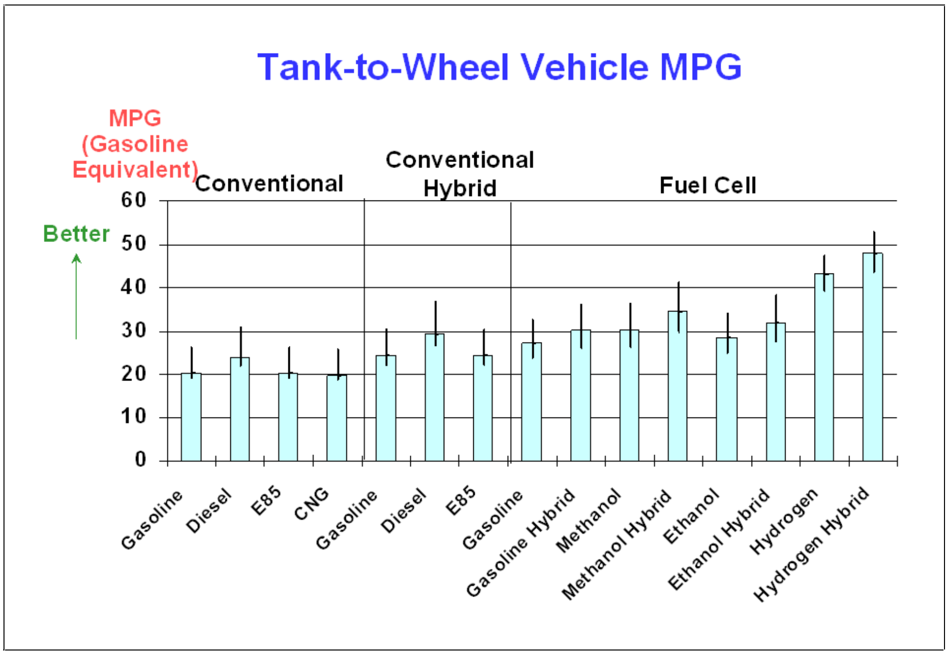

Parallel Hybrid Configuration
(Input Power Assist)
Battery
Inv
Motor
Gear
Motor can assist engine
thus transmission may be
in a high gear
Vehicle may be Launched
with Battery Only
Regeneration
Engine
n-Speed
Automatic
or CVT
Engine can operate
at low speed and
high load
Engine may be disconnected
during decel and launch

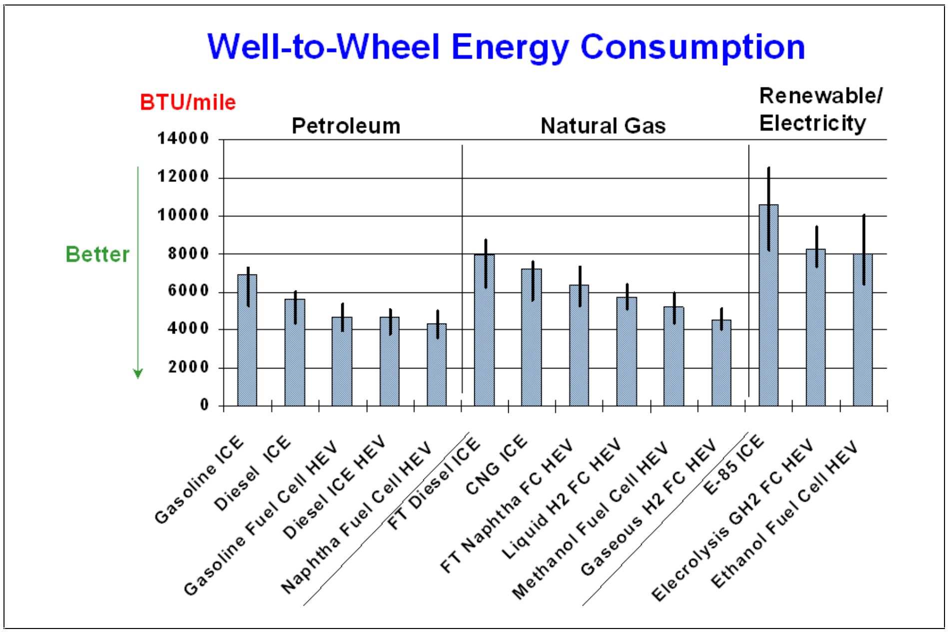
Fuel Cell Processor HEV Configuration



15 Propulsion Systems Considered

	Conventional	Conventional Hybrid	Fuel Cell Non-Hybrid	Fuel Cell Hybrid
Gasoline	X	X	X	X
Diesel	X	X		
Ethanol/ E85	X	X	X	X
CNG	X			
Methanol			X	X
Hydrogen			X	X

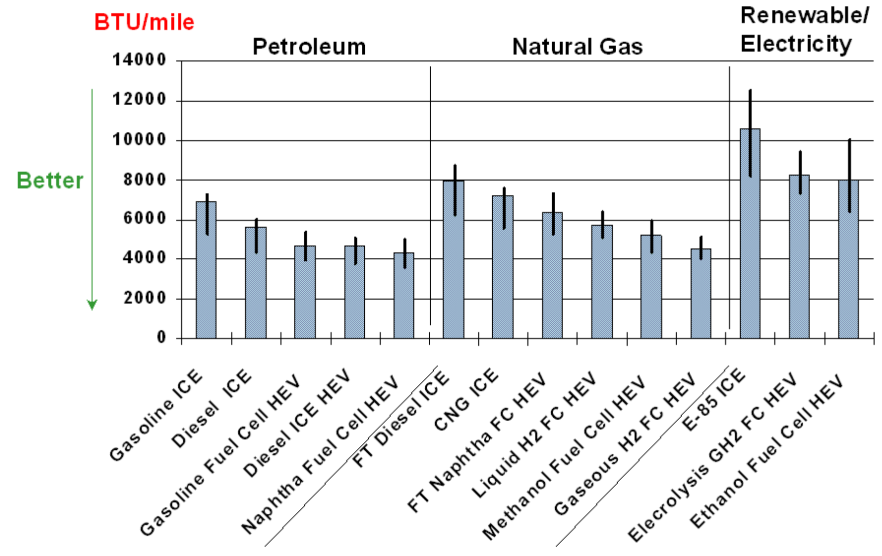

Tank-to-Wheel Vehicle MPG
MPG (Gasoline Equivalent)
Better
Conventional
Conventional Hybrid
Fuel Cell
60
50
40
30
20
10
0
Gasoline
Diesel
E85
CNG
Gasoline
Diesel
E85
Gasoline
Gasoline Hybrid
Methanol
Methanol Hybrid
Ethanol
Ethanol Hybrid
Hydrogen
Hydrogen Hybrid


Well-to-Wheel Energy Consumption
BTU/mile
Better
Petroleum
Natural Gas
Renewable/ Electricity
14000
12000
10000
8000
6000
4000
2000
0
Gasoline ICE
Diesel ICE
Gasoline Fuel Cell HEV
Diesel ICE HEV
Naphtha Fuel Cell HEV
FT Diesel ICE
CNG ICE
FT Naphtha FC HEV
Liquid H2 FC HEV
Methanol Fuel Cell HEV
Gaseous H2 FC HEV
E-85 ICE
Electrolysis GH2 FC HEV
Ethanol Fuel Cell HEV

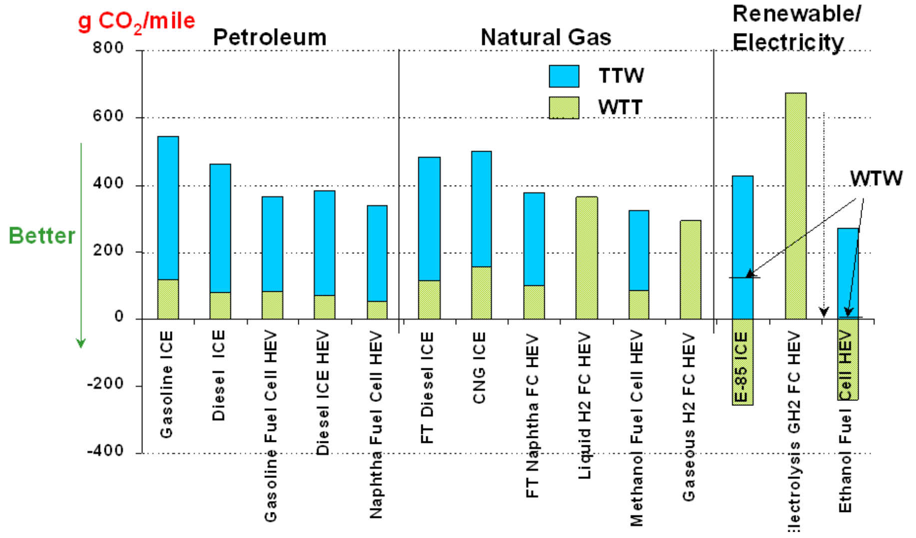
75 Fuel Pathways Examined

Feedstock -> Fuel	Pathways Analyzed
Petroleum to Gasoline, Diesel or Naphtha	8
Natural Gas to CNG	3
Natural Gas to Methanol	6
Natural Gas to Fischer-Tropsch Diesel	6
Natural Gas to Fischer-Tropsch Naphtha	6
Natural Gas to Centrally-produced Gaseous Hydrogen	6
Natural Gas to Centrally-produced Liquid Hydrogen	6
Natural Gas to Station-produced Gaseous Hydrogen	3
Natural Gas to Station-produced Liquid Hydrogen	3
US and Regional Power Generation Feedstock Mix to Electricity	4
Station-produced Gaseous Hydrogen via Electrolysis	6
Station-produced Liquid Hydrogen via Electrolysis	6
Ethanol from Renewable Feedstocks	6
E-85 with Ethanol produced from Renewable Feedstocks	6

Well-to-Wheel Energy Consumption



Well-to-Tank + Tank-to-Wheel Greenhouse Gases



US Well-to-Wheel Study Conclusions

- **Fuel cell vehicles powered by clean gasoline offer greatly reduced greenhouse gas emissions vs. today's powertrains/fuels**
- **Diesel hybrid is very competitive and a clear leader among non-fuel cell powertrains/fuels**
- **CNG does not appear to offer any benefit vs. conventional fuels for internal combustion engine (ICE) vehicles**
- **Methanol fuel cell vehicles do not offer an advantage vs. gasoline fuel cell vehicles**
- **Renewable fuels from lignocellulosic conversion offer the lowest greenhouse gas emissions**

What is not covered by this Study

- WTW Criteria pollutants (PM, NOx, CO, HC, SOx) since there is very few data on the WTT side.
- It is not a Life Cycle Assessment ie, the effects of infrastructure building and the costs are not figured

A similar study is underway in Europe for :

- The gas and oil supply are different
- The vehicle efficiency are different
- The tuning of the refining system is different
- More renewable fuel pathways are required (biofuels).

Principal Investigators

- Well-to-Tank
  - Michael Wang, Argonne National Labs (lead)
  - Norm Brinkman and Thomas Gibson, GM - R&D
  - Tony Finizza, GM - GAPC consultant
  - Andrew Armstrong and Jim Simnick, BP
  - Gilbert Jersey and John Robbins, Exxon Mobil
  - Jean Cadu, Shell
- Tank-to-Wheels
  - Trudy Weber, GM - R&D (lead)
  - Dave Masten, GM - GAPC
  - Moshe Miller, Advanced Development Corporation
  - Gerald Skellenger, GM - R&D
  - Martin Fasse and Peter Kilian, Adam Opel AG - GAPC
- Well-to-Wheels
  - Tony Finizza, GM - GAPC consultant (lead)
  - Raj Choudhury, GM/Adam Opel AG - GAPC
  - Jim Wallace, GM - GAPC consultant (lead, overall project)

FROM LIFE CYCLE ANALYSIS APPROACH TO MONETARISATION OF THE IMPACTS: AN EVALUATION IN TERM OF DECISION PROCESS[1]

Marc Darras
Gaz de France, Paris (France)

The decision process both of policy-makers and firms is never limited to a strict cost-benefit analysis. In the field of energy, a whole chain of decision is needed to allow a good and efficient use of any energy form, from the long-term investment of exploration at the upper-stream, to the development of process and the affordability analysis by the final consumer. Every decision needed along this chain affect the supply, and the security of supply, the quality of energy, the economical and social development both directly by the activity of the energy sector, but mostly indirectly through the benefit of energy, and finally the environment in a very large manner. These decisions are taken having a view of all these non-economic effects, even if the framework of decision is not always clearly organised to provide the highest level of rationality.[2]

Following Rio Conference, a strong emphasis has been put on sustainable development which promote a social, economic and environmental develop for the present and future generation. The programme Agenda 21 proposes goals in further multiple direction than the ones above-mentioned.

The life cycle analysis offers a new tool in the direction of integration of the multiple physical flows of matters around industrial processes and products, including the usage stage. Simultaneously, it stresses the multidimensional impact of the human activities, first in term of physical flows, but then in term of impact on the goals stated above.

In this perspective, integrating framework for decision are needed, especially at the more global level. One of the proposed frameworks is the

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1. This article presents the views of the author and is not a position either of Gaz de France or the International Gaz Union.
2. Concept still to be defined.

monetarisation and cost-benefit analysis, as an extension of the quantitative economy. The multi-dimensional choice, unfit for strict order choice, is reduce to one variable.In the presentation below, because of the importance of this approach in connection with Life Cycle Analysis and public policies, we concentrate on the analysis of this framework. We will show that it might shed light on the decision process, especially if associated with its uncertainties, its limits in term of process or criteria taken into account, the indication of other social criteria not taken into account, and other method of evaluation. At the end, this is mainly a source for a better multi-criteria decision process.

1. What is it all about?

Industrial activity is by nature a transformation activity involving materials from various resources; goods or services are produced. During this process wastes are generated in a gaseous, liquid or solid form.

The traditional operational field of a firm is economy, where goods are exchanged on the basis of values fixed within a market. The increased use of limited natural resources and the awareness of the impact of industrial or domestic waste and effluents have revealed another field, which is the environment. Consequences outside the economy must therefore be integrated to the human activity assessment: the economists have named them externalities.

For the Firm, these externalities often exist in two ways: external to its economy field, external to the activity field of the Firm. It really means taking into account on one hand the diminution of resources (raw materials, space, etc…) and, on the other hand, the impact to which the process effluents contribute.

The Firm must then consider theses environmental externalities in alternative positions:

- In the face of the evolution of a society wishing to take into account environmental issues when setting its options; for the Firm, this is part of the strategic positioning of its activity, of its products.
- In the face of its own production and investment options, either with the point of view of exemplarity – which is a strategic option, or plainly within the frame of regulatory and social constraints.

In either term of the alternative, rationality must be preserved in order to optimise industrial production in a way adapted to clarified social options.

The long practice of cost/benefit analysis in firms, as well as a search for global optimal economic balance have led economists to try and approach environmental impacts as if it were a cost: it is the valuation of impacts or monetarisation. This approach has two anticipated advantages:

- The attenuation of impacts of different natures, incommensurable (how can one say that a noise nuisance is of less importance than a spoilt landscape) within a common measurement unit, which can further be merged to economic costs in the process of optimised calculations.

- The assessment of the cost of the environmental impact of an activity, which would enable a polluter-payer approach.

2. What are the methods?

Two environmental cost assessment methods are commonly identified: the two-tier methods, and the direct methods.

The two-tier methods, or dose-response analysis, are the results of an analytic approach:

Emission → Dispersion → Impact → Evaluation

The direct methods progress directly through an identification of the economic value of an impact, neglecting its causal chain. Several can be identified:

- Cost of damage repairs (medical care, double glazing, …).

- The transport costs method, in which the value of an environmental “free” asset (park, forest) is assessed to the value of the transport expenses required to benefit from it.

- The hedonist pricing method, in which after the survey of various data related to an asset on different markets, the shares allocated to each environmental factor (noise, landscape, … impact on the price of housing) are valued.

- The contingent pricing method, which processes directly through questionnaires to asses the price each agent attaches to an environmental asset (willingness to pay, to receive).

Each one of these methods presents several alternatives which have been worked around since the early 70s.

In reality, and here for the purpose of assessment of the valuation stage, the merits of the direct or two-tier method are not significant. Indeed, the crucial features of these approaches are:

- The identification of any impact which will require evaluation (noise pollution, morbidity, odours, …).

- The content of the evaluation indicator, whether a valuation refers to a measuring unit or is globalised.

In the dose-response method, physical pollution indicators (waste volume and substance, particle concentration, …) are initially available, which are converted into impact through a dose-response function (toxicity, eco-toxicity), then through a monetary valuation of the response. The second approach, the direct method, also allows to deal with impacts which cannot be quantified through an intensity scale (aesthetic pleasure, …).

The effective content of the value indicator (= value) of an impact is a crucial process. Indeed, this value can represent, capture, according to the circumstances:

- The value of the assets within the economic scope directly supported by an industrial plant (cost to reforest quarries, now internalised by regulations) or collectively (cost of façade restoration as a pollution indicator).

- The value of impacts which slip out from the economy scope because they are not connected generally to markets: "option value" which referred to the preference for a delay of decision, or "quasi-option" which enables to maintain reversible options, "existence value", "legacy value".

In the latter category, individual or collective choices have made their appearance, tied in with sociological impacts (free access to the forest, sacred aspect of worship places, protection of whales, …), but as well impact in nature related to the precaution principle (long term storage of radioactive waste, biodiversity, …) very far apart from the exchange of goods. These impacts are essential in the decision process, or more precisely in the process of taking possible decisions with all the stakeholders, even if the rationality is to be clarified in some cases. The valuation indicators do not then necessarily capture all of these concepts.

3. Environmental impact valorisation difficulties and limits

The experience gained on the environmental impact valuation methods over more than twenty years, as well as the methodological consideration that surrounds this approach, allows delimiting a number of difficulties, which should be kept in mind when assessing the outcome of such surveys. We have compiled these difficulties under a few headlines, without any specific order or relative significance for each of the points.

3.1 Domain of assessment

The externalities monetary valuation system leads to questions about the nature itself of what the valuation actually represents. Therefore, it appears that there are different levels according to the valuation mode.

Firstly, and when the values are obtained either from market costs (façade restoration, structural maintenance, …), or from actual expenses (travel costs), valuations are commensurable with actual market economic costs.

However, when they are obtained through contingent cost methods, what consistency do these bear with economic costs? Let us go back to the previous example: the questioning about the willingness to pay to have clean façades does not necessarily lead on to the value of the restoration. If actors are well informed, involved in such type of operations, their answer will be biased by their knowledge of the actual economic cost, and they will tend to come close to it. However, an actor away from this problem will determine a cost in relation with the importance such an operation has for him, hence linked to his income, and, on another hand, to his sensitivity to the aesthetic of façades and other competing needs/wishes: it will then be a matter of psychological sensitivity. The connection with an economic cost becomes diluted, and the combination more precarious.

The experience acquired on contingent valuations (willingness to pay, to receive) shows the great variety of appraisal and the sensitivity to psychological factors: structure of the questionnaire, type of questions, … Finally, there are impacts which escape valuation since they remain inconceivable to the individuals questioned: can forest cease to be a free asset when one lives in Finland?[3]

Therefore, there is a necessity to clearly make a distinction between valuations actually emerging from the economy sector from those conveying or capturing in part a social or psychological value (opportunity of a CMU unit,

3. Not being a Finn, it is just a guess on my own.

Contingent Monetary Unit, which will not mix will true monetary exchange or only on certain evaluation phases?).

3.2 Aggregation of the effects

For a method, within an analysis:

$$\text{emission} \rightarrow \text{transfer} \rightarrow \text{impact} \rightarrow \text{valuation},$$

the valuation stage will allow to aggregate the impacts. This is a delicate procedure. Indeed, some joint emissions have an impact that is different from the sum of impacts of each emission, either because the impact has been added twice, or because the joint effect is of a different nature. In this way, a recent survey of urban pollution was found difficult to wrap up Thus must be monitored the consistency between types of impacts identified, the estimate of the dose-response function and of the impacts assessed. Finally, the same emission can partake in several impacts of strongly diversified natures (particles produced by motor vehicles can lead to respiratory problems and to the soiling of façades). The valuation is no more connected to the emissions, but to a complex application based on the emissions.

On another hand, the mode of identification of values, and particularly using the hedonist methods, leads to limit the use of these values in domains of impact of the same order of magnitude than with the observations that have been used as a basis for the identification. Any extrapolation would be hazardous since there is no identification of causalities. This is actually the case for any empirical rule statistically determined and based on a given set of observations.

3.3 Temporal and spatial aggregation

The traditional economic calculation uses temporal aggregation using present value ratio, which shows the preference for the present. There is massive literature dealing with the effective value to be considered. However, this approach requires that only actions be considered that do not include any "irreversible" effects, since present and future actions must be interchangeable by nature.

Moreover, when considering environmental effects that might be delayed or spread over a very long period (greenhouse effect, long-life radioactivity, …) the present value estimate of any effect becomes negligible at a 50-100 year time-horizon. The issue here is no more taking a risk as in a traditional cost-benefit present value calculation, but more an ethical problem towards the future generations. Will this point be lifted just counting on a technical evolution to come, thus allowing future generations to efficiently compensate technically and economically for an impact passed on by the present generation? Finally for certain distinct class of risks, impacts, of major importance even if not with a quantified probability of occurrence, a precautionary principle forbids this type of temporal aggregation.

Inter-temporal aggregation sets a second problem due to an ambiguity. We have indeed determined that contingent valuation is more a social appraisal indicator than a value in itself. It therefore rests on the social distribution of the values, and on the connected rights and duties. This distribution of the values varies in time, and even more when we move away from fundamental ethical values towards preferences, tastes … It is therefore impossible to trim down these future preferences to the present ones, hence to assess them. In this context, an option based upon identified present social values and leaving major options for the future open is the only alternative.

Spatial aggregation raises a problem of similar nature, particularly when there are spatial heterogeneities. As an example let us take the impact assessment of the establishment of a mine. If it appears that this establishment must take place in a low-income area, then the impact based on the cost of housing is assessed to be low. A similar problem exists when the morbidity based impact is valued through the economic cost of an economic agent (education, production to be, …): the impact is then smaller in less developed areas. It is therefore necessary to check that the environmental impact design to help the decision process does not introduce an heterogeneity between the populations, eventually correct in economical terms, but inconsistent for the social choice purpose.

3.4 Accuracy of the assessment

The accuracy of the global assessment rests on numerous aspects:

- Variation/accuracy of individual impact assessments.
- Accuracy of the assessment of the emission, transfer and response (dose-response function).
- Good sampling of all potential impacts, particularly when the analysis is based on emissions.
- Appropriate identification of potential cumulative effects.

Each of the above stages is subject to a more or less obvious ambiguity. This uncertainty must hence be estimated, taking into account the cumulative effect through the valuation procedure.

4. What is the use of monetary evaluation of externalities for the firm?

As previously stated, the externalities monetary assessment appears within two situations:

- A global context in terms of social valuation in which is encompassed the activity of the firm; this is the case for the full cost of energy.

- A micro-economic context, directly linked to the activity of the firm.

Taking into account the actual context of externalities monetary valuation, we will now study how this approach can be better used within the firm decision process.

4.1 Global appraisal of the environmental impact of energy

Global appraisals of the environmental impact of energy have been attempted over the recent years by several organisations, and notably by the European Union on the cost of electricity (ExternE).

The approach of externalities seems to suggest here a reasonable framework for the debate. What about it? On the basis of the above analysis, is appears that:

- The traditional economic approach calculates an economic optimum for the exchange of goods; taking into account full externalities should lead to an economic under-optimum, or to an optimum under "social" constraints.

- The procedure is kept within the bounds of economy as long as it integrates only the share of environmental effects (external or internal) which is relevant to the economy, otherwise an added monetary flux is added in the system (contingent monetary unit flux).

- When monetarisation is extended to effects beyond this economic scope, the whole of the environmental effects is still not taken into account: it is therefore a light shed on only a part of the environmental impact.

- The appraisal of environmental impact hence requires further investigations complementary to monetary appraisals.

- Concerning the monetary valuation, the value uncovered must be associated with a survey covering the uncertainty and sensitivity to the monetary unit values selected.

In fact, we are finally confronted to a bundle of indicators, some of which are given in monetary or quasi-monetary terms. In order to analyse the positioning of a decision, each one of these indicators must be assessed so that the most relevant opinions can be expressed and help improve the consistency of the decision process. One example would be to take into account the outcome of a strict economic survey (including, of course, the environmental effects which have been internalised according to the regulation or voluntarily by the firms, the consumers, ...), then include one or another environmental effect depending on what the stakeholders can valorise, may negotiate or might take into account. The whole set will act as a support to the decision process.

However, reasoning over the consistency of a decision options can be compared with the above remark to a multi-criteria analysis, in which irreducible data are compiled and clearly weighed in accordance with the options set by the various parties (firms, administration, consumers …) through the procedure aiming at defining the weight for each parameter. One will notice that this procedure can be implemented directly at impact identification level, associated or not to an extensive valuation. Furthermore, it forces the reflection on the basis of the weights of each impact.

Finally, let us note that there is no rational link between a monetary evaluation of externalities which lay outside the economic domain, and the value of a tax which is directly affected in the economy *stricto sensu.*

4.2 From life cycle analysis to monetary valuation of a specific activity of a firm

The review of a specific activity of the firm in terms of environment can be the basis of a life cycle analysis. There again, the remarks made earlier relatives the consistency of a cost-benefit using monetary valuation of impacts in the decision process following the life cycle analysis. Therefore it is more interesting to look at the various option opened by a Life Cycle Analysis.

First of all, it should be pointed out that numerous industrials underline the gain achieved by the simple analysis of the material flux within the processes, or for products, a sharper approach than the traditional globalised cost-benefit assessment.

Insofar as various valuations appear which must be taken into account simultaneously, a multi-criteria procedure can be used. Practically, it is already a more or less formal approach, because a number of criteria are integrated such as the maintenance of a process, the development potential, … which are not always fully economically assessed in terms of advantages. Then, noise, air and water impact can be added to this approach as independent criteria. It is for instance one of the basis of eco-efficient approaches promoted by the WBCSD. Furthermore, such an approach may be a good basis for a multi-stakeholders decision, within and outside the firm.

This approach leads to study then the processes in terms of cost-efficiency, and no more in terms of absolute cost-benefit. The environmental externality is integrated to the constraints of the procedure as a standard to comply with: the various processes or products can then be compared in terms of cost, the costs being related to actual economic values. This approach is the cost efficiency approach. It is to be underline that it is a common industrial practice under the format of technical clauses (and sometimes administrative) within the supply specifications for instance. As an add-on to this approach, the sensitivity of the result to the level of the standard can be tested.

5. Summary of the conclusions

With regards to the consideration of the stakeholders for their options, and even if the apparent simplicity of:

cost (economic cost + monetarised externalities) – benefit

seems to be a fair option, the valorisation of externalities is tainted with too many uncertainties for such an approach to be regarded as rational. Alternative approaches must be considered: multi-criteria methods on the impacts, cost-efficiency approach, … In all cases however, the stage where only a limited numbers of environmental effects taken into account must be selected is unavoidable: this selection is a decision to be made in accordance with the environmental approach defined by the firm, the priorities of the authorities.

The inter-temporal aggregation leads to uncertainty and may rise ethical issues which are not solved by the standard economy procedure (present value). On the long term, the value of the inter-temporal rate of actualisation should be selected very carefully. Furthermore, when the risk corresponds to a major change, such as corresponding to the application of the Precaution Principle, the aggregation is no more founded.

Confronted to an approach based on the monetary evaluation of externalities, for example the environmental cost of energy, each and every constituent must be scrutinised, and the robustness of the outcome must be assessed according to the variability of the elementary valuations. Moreover, and for those impacts, which remain outside the economic scope, it is necessary to study the benefits of the monetary valuation carried out. The whole operation must allow the stakeholders to position themselves, to negotiate on a better understood rational basis.

Negotiations based on the multi-criteria methods can be consistent and interesting insofar as they can help to bring to light and define the context. Such approach can take full advantage of the result of a Life Cycle Analysis. However, the implementation of such methods requires a strong involvement of the actors concerned to develop the motivation behind the weights given to the various criteria.

There is no link between the monetary value of externalities of non-economic nature, and the value of the tax designed to internalise externalities. Furthermore, the geographical distribution of impact should be taken into account in the repartition of the corresponding regulation in order to develop the adequate adaptation. This is particularly important in recycling the revenue, if a taxation is applied.

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LIFE CYCLE ANALYSIS AS BASIS FOR EVALUATING ENVIRONMENTAL IMPACTS OF ENERGY PRODUCTION[1]

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1. LCA methodology

To evaluate what LCA is suited for and what it is not suited for, it is necessary to take a brief look at the methodology. The methodology and formal framework has developed significantly during the last ten years. The most commonly used frameworks are those given by SETAC and in ISO 14040. The latter divides a Life Cycle Assessment (LCA) (see Figure 1) into four steps:

1. Goal and scope definition.
2. Inventory analysis.
3. Impact assessment.
4. Interpretation.
5. (Improvement).

2. Goal and scope definition

The problems of defining the goal and scope of a study should not be underestimated. What is the purpose of the analysis? Do we for instance want to compare two specific production chains for one identical product? Example: Should we build a coal fired or a gas fired power station, each with a specific technology and specified fuel sources, at a specific site? Or do we want to compare average production chains in a country. The results of those two analyses will most probably be quite different.

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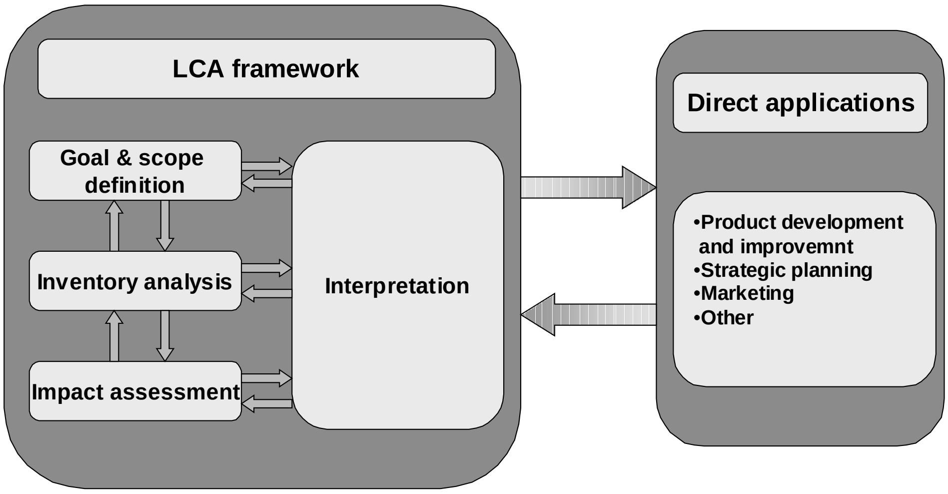
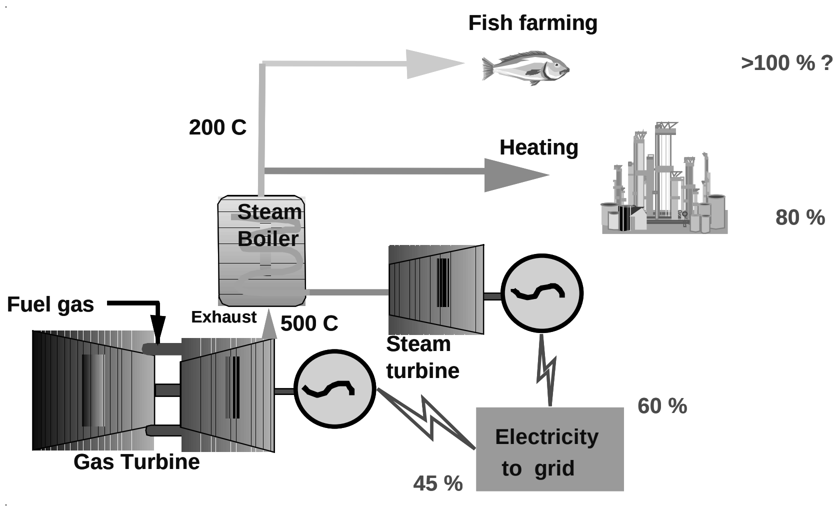
1. This article presents the views of the author and is not a position either of Statoil or the International Gaz Union.

Figure 1. **Life cycle assessment (LCA) framework**



Production chains will in most cases have several products that may be spun off at different places in the chain. Natural gas is in most cases produced together with oil that ends up in a large number of products. A gas power station produces a lot of heat that may be used for district heating, in nearby industrial plants etc. and the low temperature heat may be used for e.g. fish farming (see Figure 2). How do we allocate the impacts in the production chain to each of the products?

Figure 2. **Energy chain**



Consider various gas power stations as an example, a simple gas turbine power plant will have a typical efficiency of 45%. A combined cycle gas power plant, where the turbine exhaust heat is used to produce steam which again drives a steam turbine, may typically have an efficiency of 60%. The combined cycle plant will then have a 25% lower CO_2 emission per kWh. If the exhaust heat instead is used to produce hot water for home or industrial use, the overall efficiency may exceed 80%, but the electrical efficiency may drop to, say, 50%. The CO_2 emissions per kWh will then increase again by 20% if only the electricity generation is considered, but decrease with 25% if the total energy production is considered. Both results may be correct, depending on the purpose of the LCA. If the low temperature water is used for fish farming, what is then the functional unit of the total plant?

Some industrial plants have surplus energy, which may be used to generate electricity for export to the grid. The electricity may then be regarded as a by-product. How should the impacts of the plant be allocated to the electricity?

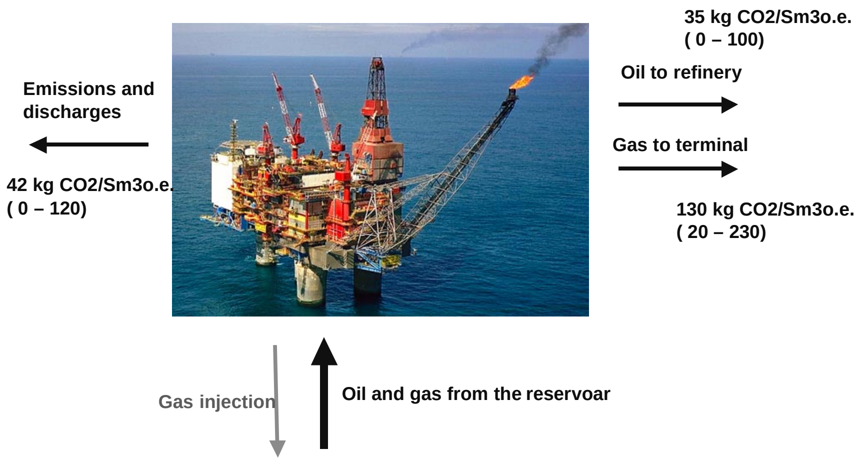
These simple examples serve to show that it is very important to be completely clear on the purpose of an LCA, and also illustrates that results from one LCA can not uncritically be used for other purposes.

3. Inventory analysis

Inventories for "traditional" parameters like resource consumption, emissions, discharges, energy use, waste generation and treatment, land use, noise etc. will normally be of good quality (high accuracy), at least for simple, well-defined, production chains. It is more difficult to describe other parameters like visual intrusions, aesthetical disturbances, changes in migratory patterns for animals or birds etc. Non-regular phenomena, like accidents pose special problems, particularly major accidents with low probability but severe consequences.

With a view back to the first step of the analysis, goal and scope: How representative is the inventory that is established? An example: The CO_2 emissions for gas production and transport from the Norwegian North Sea ranges between approx. 20 and 230 g CO_2/Sm^3 gas with an average of approx. 130 g/Sm^3, depending on which field the gas is coming from (Figure 3). This corresponds to approx. 1-10% of the energy content in the gas, with an average of approx. 5%. For one field, the value may vary by a factor of 3-4 over the field's lifetime. Inventories from two seemingly identical production chains may therefore show large differences. Again, it is very important to be completely clear on the purpose of the analysis. An inventory that is adequate, or even absolutely correct, for one purpose may be completely misleading for another. In fact: If you tell me the answer (e.g. energy consumption or emissions) that you want (within reasonable limits) regarding oil and gas production, I will find a production scenario in the Norwegian North Sea that will provide your results in a way that is absolutely correct according to the LCA methodology. The same possibilities for “manipulating” the inventory exist for many production chains, implying that externalities based on LCA inventories must be treated with appropriate care.

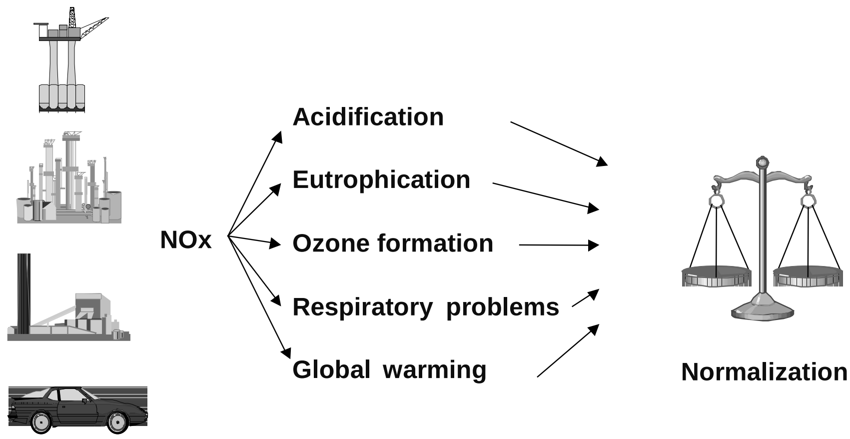
Figure 3. **CO_2 emissions from Norvegian oil and gas production**



4. Impact assessment

The impact assessment is a much more complex and difficult step than the inventory analysis. Contributions to local and regional effects cannot simply be added throughout the production chain. NO_x emissions for instance, which contribute both to air quality/health impacts, acidification, eutrophication, ozone formation and global warming, will have different impacts depending on the geographical site of emission (see Figure 4). Both the total impact and the relative contributions to these impact categories will be different depending on the location. Where as air quality/health impacts will probably be the most important category in urban areas, eutrophication and acidification will probably be relatively more important in rural areas. It may therefore, for instance, be sensible to spend more energy in a refinery, and thereby increase its (NO_x) emissions, to produce a gasoline quality that may lead to reduced (NO_x) emissions from cars driving in urban areas.

Figure 4. **Impact assessment**



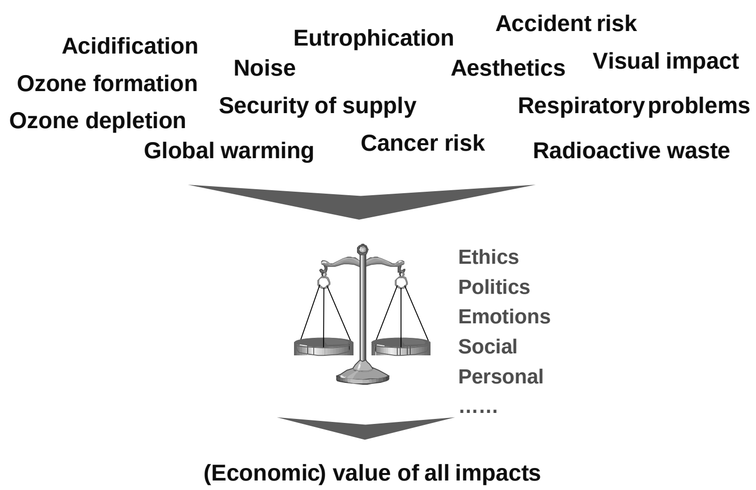
Such “relocation” of emissions may, however, lead to other effects that may be hard to evaluate in an LCA. Example: we performed an LCA of conventional gasoline and gasoline with MTBE some years ago. According to the “normal” environmental parameters, the two gasoline qualities seemed to give comparable total impacts but parts of the emissions were “relocated” from the roads to the sites of the refinery and MTBE plant. A 10% increase in energy consumption, and thereby emissions, to produce gasoline with MTBE, was offset by 0-5% reduction in NO_x, CO and HC emissions from the cars. What we did not foresee and include, was possible negative impacts of MTBE on e.g. groundwater and on people’s health that has come into focus in the last few years. Even if we had known them, how could we have brought them into the analysis in comparison with the other impacts to evaluate the total effects? If we should have tried to use monetary values, how should we value people’s feeling of (lack of) well-being?

The ExternE study also clearly shows that equal emissions/loads may result in widely different (monetary) impacts in different countries.

Impact assessment is therefore a troublesome task, even for products or production chain where the main impact categories are similar.

When the impact categories are widely different, there is no objective way, perhaps even not a scientifically valid way, to compare the total impacts of different products or production chains. A number of methods for “normalisation” have been proposed and are regularly applied in LCAs. Such methods, including monetary valuation, may give valuable information when interpreting the results from an LCA, but they may also be completely misleading. The simple reason for this is that there is no unambiguous way of comparing the effects like acid rain, climate change, peoples perception of noise, visual impact, aesthetics, risks for major accidents, health impacts, security of supply, risk for proliferation of radioactive material, peoples perception of risks etc. All attempts to compare these will inevitably involve political, ethical and emotional aspects as well as personal and societal dimensions and priorities will have to be made (Figure 5). The way these priorities are made are obscured in an analysis where the simple output is a monetary value and it will require a significant effort and quite a lot of skill to follow the thoughts and assumptions made by the analyst or developer of the method. These steps should therefore rather be performed as an open, transparent political/public discussion than a calculation in an LCA. A further complication is that even the comprehensive ExternE project has not been able to value several important impact categories. Other speakers will cover the challenge of setting monetary value on different potential impacts in more detail.

Figure 5. **No objective scale**



LCAs are therefore not suited for comparing impacts of widely different electricity generation options like fossil, wind, hydropower, nuclear and solar and therefore not suited as basis for political decisions regarding the balance between these. They may, though, be suitable for comparing different transportation fuels. Statements like “New research reveals the real costs of electricity in Europe”, which have been used to describe the results of from the ExternE project, are therefore simply not true. ExternE at best reveals a possible way of setting monetary value to some of the impacts of electricity generation.

5. Interpretation

This final step of an LCA should evaluate the results of the analysis and all choices made during the course of the analysis in terms of soundness and robustness, and overall conclusions should be drawn.

As should be evident from the above discussions, an LCA that is designed to give answers to specific questions, as posed in the goal and scoping phase, may give anything from valid to misleading answers to other questions. The results from an LCA made by an industrial company for evaluating different production technologies before erecting a new factory, may therefore be unsuitable, even misleading, for authorities when deciding taxes or other incitements for driving the development in preference of certain products or production technologies.

The discussions and few examples above illustrate that LCAs have many limitations and pitfalls. The results from LCA should handled with care and with due consideration of its goal and scope. On the other hand, LCAs may give valuable information in many decision processes if they are used for what they are suited for and as one of several tools to clarify positive and negative effects of a product, a process or activity.

According to our experience and opinion, LCAs may be suited for:

- Systematic description of resource consumption, environmental stressors (emissions, discharges, waste, noise), possibly also accident risks and consequences and to describe where (geographically) the environmental burdens occur.

- Highlighting environmental challenges and to pinpoint where in the production chain they can be found.

- Comparing impacts of products or production chains with similar impact categories, like electricity production from gas, oil and coal.

- Comparing alternative technology options for a factory, including technologies for reduction of emissions and discharges.

- Comparing alternative locations for a facility.

LCAs are not suited for:

- Comparing impacts of products or production chains with widely different impact categories, like electricity production from fossil fuels with e.g. hydropower, wind or nuclear.

- Internalisation of externalities for energy production and supply.
- Making decisions on taxes or other instruments to influence power generation technologies unless used in combination with other tools and evaluations.
- Industry evaluation of "green" taxes.

The work on externalities is important and interesting, but one should be very conscious of the limitations of the available methods and that they will never give objectively correct answers. The current state of the art regarding externalities of energy production and products is far from giving a complete picture, although significant progress has been made during the last decade. The results from studies of externalities should therefore never be regarded as more than one of several inputs in any decision process, whether in industry or in politics.

THE EXTERNAL COST OF THE NUCLEAR FUEL CYCLE

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Abstract

The external cost of the nuclear fuel cycle has been evaluated in the particular context of France as part of the European Commission's ExternE project. All the steps in the fuel cycle which involve the use of cutting edge technology were taken into consideration, from mining of uranium ores to waste disposal, via construction, dismantling of nuclear power plants and the transport of radioactive materials. The general methodology adopted in the study, known as the "Impact Pathway Analysis", is based on a sequence of evaluations from source terms to the potential effects on man and the environment, and then to their monetary evaluation, using a single framework devised for all the fuel cycles considered in the ExternE project. The resulting external cost is in the range of 2 to 3 mEuro/kWh when no discount rate is applied, and around 0.1 mEuro/kWh when a discount rate of 3% is considered. Further developments have been made on the external cost of a nuclear accident and on the integration of risk aversion in its evaluation. It appeared that the external cost of a nuclear accident would be about 0.04 mEuro/kWh, instead of 0.002 mEuro/kWh without taking risk aversion into account.

1. Introduction

In the context of the Joule Programme of the European Commission, the ExternE project has been implemented from 1991 to 1995 in order to assess the external costs of various fuel cycles used in the production of electricity [1]. Within this project, the CEPN (Centre d'étude sur l'Évaluation de la Protection dans le domaine Nucléaire) has completed the assessment concerning the nuclear fuel cycle in the particular context of France [2]. All steps of the fuel cycle which involve the use of cutting edge technology were taken into consideration, from mining of uranium ore to waste disposal, via construction, dismantling of nuclear power plants and the transport of radioactive materials. The general methodology adopted in this study, known as the "Impact Pathway Analysis", is based on a sequence of evaluations from source terms to the potential effects on man and the environment, and then to their monetary evaluation.

The aim of this paper is to present the main results of this study and to discuss the meaning of the different indicators and assumptions adopted in the evaluation of the external costs. As far as the nuclear fuel cycle is concerned, a first set of questions arises from the risk assessment (especially the long term and global impacts, as well as the integration of potential consequences associated with severe reactor accident). Beyond this quantification of the physical impacts on man and the environment, the evaluation of the external costs associated with the nuclear fuel cycle also points out some questions on the use of economic indicators (monetary value of life, discount rate, risk aversion, …). Although the evaluation of the external costs implies the adoption of various assumptions, it allows to put into perspective the health and environmental effects of the different fuel cycles as far as their external costs have been evaluated within a similar framework.

2. Methodology

2.1 Stages of the nuclear fuel cycle

The French nuclear fuel cycle was broken down into 8 separate stages (see Table 1). Reference sites and 1990's technology were chosen to represent the total nuclear fuel cycle, as it exists today. In addition, the transportation of material between the sites was considered. The facilities are assessed for routine operation, except in the cases of electricity generation and transportation, where accidental situations are evaluated. The impacts of construction and decommissioning of a facility are included in the electricity generation stage. It is important to stress that this methodology does not employ a worst case scenario analysis, as is usually done for safety or regulatory compliance assessments, but intends to evaluate the impacts expected from the operations. In a few cases, however, when no reasonable alternative seemed possible, conservative values were used.

Table 1. **The different stages of the nuclear fuel cycle considered in the French evaluation of the external cost**

Stage of the fuel cycle	Site	Technology used
Mining and milling	Lodève	Underground and open pit mines
Conversion	Malvesi and Pierrelatte	Yellowcake conversion to UF_6
Enrichment	Pierrelatte	Gaseous diffusion
Fuel fabrication	Pierrelatte	Conversion of UF_6 to UO_2 pellets
Electricity generation	Belleville, Flamanville, Saint-Alban, Nogent, Paluel	1 300 MWe PWR
Reprocessing	La Hague	PUREX process
Waste disposal	Aube	Surface disposal
	Auriat (hypothetical site)	Underground disposal
Transportation		Road and rail

The impact pathway approach requires an inventory and assessment of all potential impacts, however, within the context of the ExternE project it has not been possible to consider all of these. Therefore, only the most important impacts, called priority impacts, have been included. Releases of radioactive material to the environment, which potentially impact public health, were given the highest priority. Occupational health impacts, from both radiological and non-radiological causes, were the next priority, even though the extent to which occupational health impacts can be considered as externalities has not been addressed in this study.

2.2 Impact assessment

Assessment of the impacts was organized by the type of routine emissions: atmospheric releases, liquid releases and solid wastes. The analysis of impacts of releases from severe accidents involves additional complex issues, therefore, it was treated as a distinct category. Health impacts to the workers – radiological and non-radiological – were also accounted for separately.

The most important choices for the assessment of the nuclear fuel cycle concern the definition of temporal and spatial boundaries. In this study a distinction has been made between local impacts (0 to 100 km), regional impacts (100 to 1 000 km), and global impacts (above 1 000 km) in the short term (first year), the medium term (1 to 100 years), and the long term (100 to 100 000 years) (see Table 2). The selection of a 100 000 year limit to the assessment was arbitrary, however the most significant part of the impacts are included.

Table 2. **Distribution of impacts from routine operation of the nuclear fuel cycle**

	Local	**Regional**	**Global**
	0-100 km	**100-1 000 km**	**>1 000 km**
Short term (<1 year)	Non-radiological impacts on workers Traffic accidents		
Medium term (1-100 years)	Radiological impact on workers and the public	Radiological impact on the public	^{85}Kr, ^{3}H ^{14}C, ^{129}I
Long term (100-100 000 years)	Radiological impact on the public	Radiological impact on the public	^{14}C, ^{129}I

The radiological health impacts for routine operations, and the number of potential fatal cancers, non-fatal cancers, and severe hereditary effects in future generations were estimated using recommendations of the International Commission on Radiological Protection [3]. The non-radiological impacts of accidental deaths and injuries for the general public, and the number of deaths, working days lost and permanent disabilities for the workers were based on published statistics. In accidental situations, additional impacts of the immediate (deterministic) radiological impacts and the costs of the radiation protection countermeasures were calculated using the COSYMA code [4].

The final stage of the impact pathway methodology is the economic valuation of the impacts. The economists involved in the ExternE project set a common value of a statistical life of 2.6 MEuros, based on a literature review of contingent valuation studies, adopted for all the fuel cycle assessments in the project [5]. Nevertheless, recently, lower values have been observed, especially in a French contingent value concerning road accidents. In this survey, the value of statistical life was about 1 MEuros [6]. Such a difference may largely modify the result of the external costs of the nuclear fuel cycle, which is mainly due to health effects. Furthermore, a working day lost and a permanent disability were considered to be the equivalent of 65 Euros and 19 kEuros, respectively. These values are based on those used by the French national health insurance system. The value of 0.25 MEuros, adopted for a non-fatal cancer, was based on information on direct and indirect costs from the United States of America. The costs were calculated with an annual discount rate of 3%, adopted as the reference value. Nevertheless, because of the large sensitivity of the results according to the value of the discount rate, additional calculations were performed for an annual discount rate of 10% as well as without discount rate.

2.3 Severe accidents

Accidents are one of the most controversial features of environmental assessment of the nuclear fuel cycle. Within the scope of this project, this type of assessment has been confined to the electricity generation stage and the transportation of radioactive materials between sites. Although facilities at other stages of the nuclear fuel cycle handle very large inventories of radioactive material, their activities are generally believed to be of a lower risk. The probabilistic assessment of the transportation of materials between all the fuel cycle facilities includes risks from both conventional traffic accidents and releases of radioactive material. These have been found to be relatively small.

At this time, there is no general consensus on a methodology to assess the external costs of severe nuclear reactor accidents. In this project, a risk-based approach has been adopted. Due to the complexity of the assessment and the difficulty in finding facility-specific or generally accepted input data, the evaluation that was completed provides indicative results for this type of methodology.

The source term considered in this study corresponds to a release of about 1% of the core (ST21). This source term is in the same order of magnitude as the reference accident scenario used by the French national safety authorities. To illustrate the sensitivity of the results the impacts of three other source terms are presented. The largest can be considered as release that would occur after a core melt accident with a total containment breach. The fraction of the core released, based on a source term used in an international inter-comparison study, is about 10% of the core inventory. The smallest release can be considered to represent the situation after a core melt accident where all the safety measures have operated as planned and there is only leakage from the intact containment (0.01% of the core inventory).

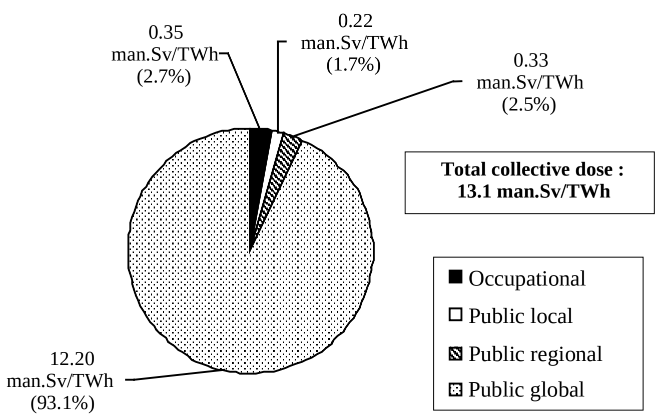
The probability of a core melt accident, based on a French assessment of a major core melt accident at a 1 300 MW PWR reactor, is taken to be 1.0E-5 per reactor.year. This is broadly consistent with other similar assessments based on engineering fault tree analysis, although a wide range of estimates have been proposed. The conditional probabilities of the large and small releases that would occur after a core melt accident are taken from a US Nuclear Regulatory Commission report, and are 0.19 for the three largest source terms and 0.81 for the lowest.

3. Results

3.1 Doses

The total collective dose for all the stages of the fuel cycle, except for the severe accident analysis, integrated for a time period of 100 000 years into the future, is 13.1 man.Sv/TWh. A closer look shows that the total local collective dose is about 0.22 man.Sv/TWh and the total regional collective dose is 0.33 man.Sv/TWh, leaving over 95% of the public dose due to the global dispersion of certain radionuclides (^{14}C, ^{129}I). If the global doses are not included, the occupational doses become a dominant contributor to the overall impacts (about 40% of the doses received) (see Figures 1 and 2).

Figure 1. **Distribution of the total collective dose associated with all stages of the nuclear fuel cycle (without accident), integrated over 100 000 years**



When all the categories of the doses are considered, the reprocessing stage, with a collective dose of 10.3 man.Sv/TWh contributes the largest portion (79%) of the total collective dose, followed by the electricity generation stage (18% of the total). If the global collective doses are excluded, the reprocessing stage diminishes in importance and is replaced by the electricity generation (0.38 man.Sv/TWh) and the mining and milling (0.29 man.Sv/TWh) stages. The doses from the enrichment stage are the least important. Tables 3 and 4 present the order of magnitude of the doses associated with the different stages of the fuel cycle, respectively for occupational and public exposures.

Figure 2. **Distribution of the collective dose associated with all stages of the nuclear fuel cycle (without accident), without global assessment**

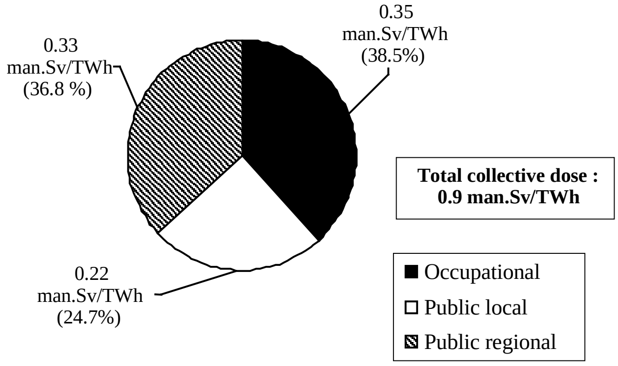


Table 3. **Order of magnitude of the occupational exposure**

Stage of the fuel cycle	**Occupational exposure (0-100 years)**	
	Individual average (mSv/year)	**Collective dose (man.mSv/TWh)**
Mining and milling	2 to 5*	112
Conversion	2	2
Enrichment	2	<1
Fuel fabrication	7	6
Electricity generation	3	202**
Dismantling	n.d.	22
Reprocessing	1	~1
Transportation	n.d.	~1

* Respectively open pit and underground mines.
** Average value for 1 300 MWe PWR.
n.d.: value not available.

In case of a severe reactor accident, an indicative total collective dose for the population (for a radius of 3 000 km) for the four accident scenarios has been estimated. The impact of the reference scenario ST21 (core melt with a 1% of the core released) is a collective dose of about 58 000 man.Sv. For the other scenarios considered, the expected risk (consequences x probability of occurrence) varies between 0.001 and 0.08 man.Sv/TWh.(See Table 5)

Table 4. **Order of magnitude of the public exposure**

Stage of the fuel cycle	Public exposure (0-100 years) (man.mSv/TWh)		
	Local (0-100 km)	**Regional (100-1 000 km)**	**Global (>1 000 km)**
Mining and milling	83	90	<1
Conversion	<1	<1	<1
Enrichment	<1	<1	<1
Fuel fabrication	<1	<1	<1
Electricity generation	~1	16	140
Dismantling	<1	0	0
Reprocessing	<1	84	480
Transportation	~1	0	0

Table 5. **Expected collective doses for a major reactor accident (ST21: reference scenario for France)**

Source term ***(% of core released)***	**Core melt probability (per reactor.year)**	**Conditional probability**	**Collective dose (man.Sv)**	**Collective dose x probability (man.Sv per reactor.year)**	**Risk* man.Sv/TWh**
ST2 *(10%)*	1E-05	0.19	291 200	0.55	0.078
ST21 ***(1%)***	**1E-05**	**0.19**	**58 300**	**0.11**	**0.016**
ST22 *(0.1%)*	1E-05	0.19	12 180	0.02	0.003
ST23 *(0.01%)*	1E-05	0.81	1 840	0.01	0.001

* 7 TWh/reactor.year.

3.2 Human health impacts

3.2.1 Routine operations

The radiological health effects resulting from the normal operation of the nuclear fuel cycle are directly proportional to the total collective doses. The expected number of health effects were calculated assuming no lower threshold for radiological impacts, using internationally accepted data from Publication 60 of the International Commission on Radiological Protection. The total number of expected health impacts per TWh are: 0.65 fatal cancers, 1.57 non-fatal cancers, and 0.13 severe hereditary effects. These results include the long-term global dose assessment.

The number of estimated deaths for the European population due to the routine annual operation of one additional 1 300 MWe PWR (about 7 TWh/y), integrated over 100 000 years, would be less than 1 fatal cancer (0.1). This can be compared to the approximate value of 800 000 fatal cancers reported in Europe each year.

It is estimated that the production of 1 TWh will result in 0.02 deaths, 0.96 permanent disabilities and 296 working-days-lost (non-radiological health impacts) in the work force for the nuclear industry. Worker accidents during the construction and the decommissioning of the reactor are the most important contributors to these values.

3.2.2 Accidental situations

The transportation of the radioactive materials between the different sites and the transportation of the materials involved in the construction and the decommissioning of the reactor result in traffic accidents involving the general public. The number of non-radiological health impacts estimated are: 3E-4 deaths and 1.7E-3 injuries per TWh. Assuming an incremental annual production of 7 TWh, less than 1 death (0.002) can be expected per year. This is insignificant when compared to the nearly 10 000 traffic accident deaths that occur in France each year. In accidental situations occurring during the transportation of hazardous radioactive materials such as UF_6, the toxicological health impacts estimated are even smaller (2E-9 deaths/TWh and 7E-5 injuries/TWh).

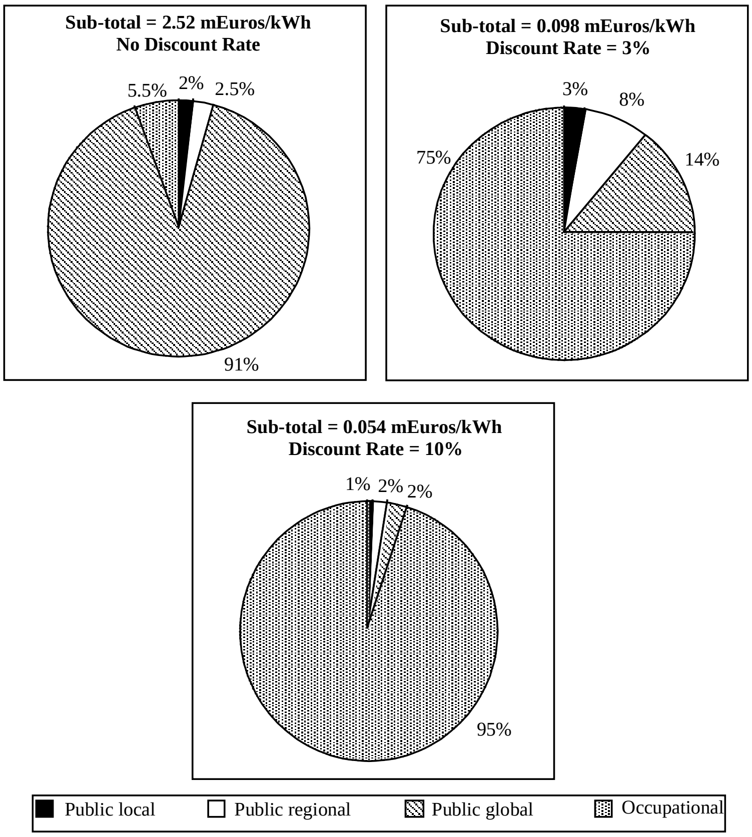
The radiological health effects from reactor accidents can be divided into two categories: the early health effects (deterministic effects) and the stochastic effects as cancers or severe hereditary effects. For the four accident scenarios considered in this study, only the two most severe accidents lead to deterministic effects, but no early deaths are expected for the reference scenario ST21. For the stochastic effects, as for normal operation, they are considered to be directly proportional to the collective doses. Depending on the scenario, the number of expected fatal cancers varies from 1E-4 to 3.9E-3 per TWh.

3.3 Monetary valuation

3.3.1 Routine operation

The sub-total of the cost presented for all the stages of the nuclear fuel cycle is about 2.5 mEuros/kWh, if no discount rate is applied. When 3% and 10% discount rates are used, the cost is reduced to 0.1 and 0.05 mEuros/kWh, respectively (see Figure 3). The current base load electricity generating costs in France are on the order of 35-40 mEuros/kWh.

Figure 3. **Distribution of the costs for the 0%, 3% and 10% discount rates**



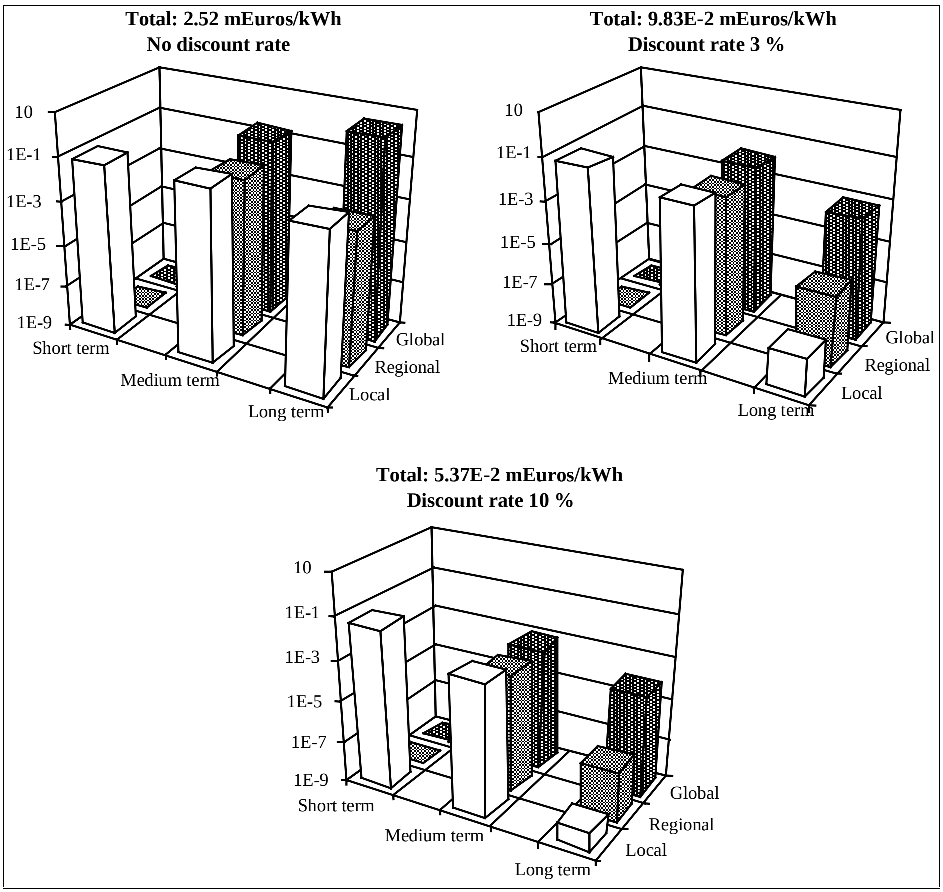
The dominant contributor to the total cost is the reprocessing stage (76%), followed by electricity generation (18%), when the 0% discount rate and the global impact assessment are implemented. When the 3% discount rate is applied (see Table 6), the construction of the reactor becomes the most important because discounting does not reduce the costs of the very short-term impacts assessed (40%). This is followed by mining and milling and electricity generation (19% and 17%, respectively). Six percent of the overall cost of the fuel cycle, at 0% discount rate, is due to occupational health impacts. This proportion increases in importance when a discount rate is applied (75% of sub-total for a discount rate of 3% and 95% of sub-total for a discount rate of 10%).

Table 6. **Distribution of external costs of the nuclear fuel cycle with the different stages of the nuclear fuel cycle, for public and workers**

Stages of nuclear fuel cycle	**External cost (routine operation, local and regional areas, short and medium term impacts, 3% discount rate) (mEuros/kWh)**	
	Public	**Workers**
Mining and milling	5.24E-03	1.22E-02
Conversion	9.90E-08	4.50E-04
Enrichment	5.06E-08	7.47E-04
Fuel fabrication	2.66E-07	6.91E-04
Reactor construction	2.16E-04	3.70E-02
Electricity production	3.81E-03	1.02E-02
Dismantling	1.06E-02	6.41E-03
Reprocessing	1.21E-03	1.91E-03
LLW disposal	1.87E-12	2.46E-06
HLW disposal	0.00E+00	6.04E-09
Transportation	2.23E-04	2.80E-05
Total	**2.14E-02**	**6.96E-02**

The sensitivity to the discount rate used is due to the relatively large portion of medium and long-term impacts associated with nuclear fuel cycle (see Figure 4). For example, it can be seen that for no discount rate, the predominant costs are due to the global assessment, however, if a 3% discount rate is used, the short term, mostly occupational impacts dominate the final result. Another example is the waste disposal stage where the relatively small costs disappear if discount rates are applied. It has been for this reason that the use of this type of impact pathway methodology for the assessment of waste disposal and global impacts has been questioned.

Figure 4. **Distribution of the costs for the 0%, 3% and 10% discount rates with time and space (log scale)**



3.3.2 Accidental situations

The costs assessed for transportation impacts in general are extremely small. The portion attributed to accidental conditions can be considered insignificant due to the low probabilities and transportation packaging.

These results reported for the four accident scenarios are indicative of a risk-based methodology. The reference scenario, considered to represent a core melt accident followed by a release of 1% of the core, resulted in a 0.005 mEuros/kWh cost. The risk for the other scenarios varies between 0.02 and 0.0005 mEuros/kWh (see Table 7). The portion of these costs that might be internalised by nuclear accident insurance has not been addressed.

Table 7. **Results of accident analysis at a 1 300 MWe PWR, for 4 different scenarios including public health effects and costs of countermeasures**

Source term (approximate % of total core released)	**Core melt probability (per reactor.year)**	**Conditional probability**	**Total cost (MEuros)**	**Cost x probability (MEuros per reactor.year)**	**Cost* (mEuros/kWh)**
ST2 (10%)	1E-05	0.19	83252	0.158	0.023
ST21 (1%)	**1E-05**	**0.19**	**17093**	**0.032**	**0.0046**
ST22 (0.1%)	1E-05	0.19	3339	0.006	0.0009
ST23 (0.01%)	1E-05	0.81	431	0.003	0.0005

* 7 TWh/reactor.year.

Further work must be done to evaluate other potential social impacts and external costs such as, *inter alia*, public perception, risk aversion, disruption of electricity supply, and decommissioning of the destroyed reactor. Besides the difficulty in assessing these impacts and costs, the partition of externalised versus internalised costs must also be evaluated.

4. Discussion

As mentioned above, the evaluation of the external costs for the nuclear fuel cycle is confronted firstly to the difficulty to integrate over long periods of time and large areas small individual levels of exposure (and thus their translation into monetary terms) and secondly to the probabilities associated with the accidental scenarios. The following paragraphs do not intend to propose solutions for integrating these impacts into the external costs but rather to point out some of the various dimensions of these specific impacts.

4.1 Nuclear accident

Calculation of the economic consequences of a severe reactor accident points out the difficult question of the choice of a reference scenario for the source term. The reference source term for the French safety analyses is focused on about 1% of the core released after meltdown. Although there is a large consensus within the scientists to consider a reference probability for the core meltdown around 10^{-5} per reactor.year, there is less analyses related to the consequences of this meltdown: it is generally considered that in most of the cases, the releases should be limited. Thus at this stage, simplified assumptions were used in the ExternE project in order to aggregate the probabilities and consequences. It is clear that this approach is limited as far as it does not consider the risk perception associated with the potentiality that a severe accident occurs as well as the indirect consequences on the economic activity.

4.1.1 Indirect costs

Thus, further to the classical evaluation performed for the health and environmental consequences of a nuclear accident, it has to be considered that there will be a decrease or an interruption of most of the economic activity (essentially agricultural and industrial productions) in the affected territories during a significant period of time. The importance of this interruption will notably depends on the size of the accident. In terms of monetary indicators, this disturbance of the economic activity will meanly induce a loss of value added (this indicator corresponding to the different direct and indirect incomes of the various “economic agents”). In order to derive the order of magnitude of the indirect consequences, calculations have been performed using the COSYMA code [4] and the value of life adopted in the ExternE project, and the following results were derived [7]:

- The reference accident is supposed to induce an indirect cost which represents about 10% of the regional gross domestic product during the first two years.

- Similarly, it represents about 0.2% of the national gross domestic product.

- The indirect costs lead to an increase of 25% of the direct external costs of a nuclear accident.

In fact, based on this calculation, one can consider a multiplying coefficient of 1.25 to be applied to the direct external costs calculations in order to derive the total external costs of the accident. This simplification seems to be reasonable as soon as we are dealing with accident scenarios leading to significant radioactive releases into the environment. In that case, the external cost associated with the nuclear accident is 0.0057, for a reference source term of 1%, instead of 0.0046 without taking into account the indirect costs.

4.1.2 Risk aversion

The main criticism of this approach is that there is a discrepancy between the social acceptability of the risk and the average monetary value that in principle corresponds to the compensation of the consequences for each individual of the population affected by the accident. In fact, it appears that there is a need to integrate risk perception – risk aversion within the calculation of this external cost.

Some economic developments have been made, based on the expected utility approach, in order to integrate risk aversion within the evaluation of the external cost of the nuclear accident [8]. One of the interest of this approach is the availability of experimental data concerning risk aversion coefficient. Although a large range of values has been published for this coefficient, mainly based on the analysis of financial risks, it seems reasonable to adopt a risk coefficient around 2 for the specific case of nuclear accident. This leads to estimate a multiplying coefficient approximately equal to 20 to be applied to the external cost of a nuclear accident corresponding to a release of about 1% of the core.

According to the previous results, the external cost of a nuclear accident would be in the order of 0.092 mEuros/kWh, instead of 0.0046 mEuros per kWh without taking risk aversion into account. In this case, it would represent about 3.6% of the external cost of the nuclear fuel cycle (with no discount rate, and without accident).

4.2 Long-term global impacts

The major difficulty from the methodological point of view with regard to the calculation of the external costs of the nuclear fuel cycle concerns the evaluation of the impacts associated with releases of radionuclides such as H^3, ^{14}C, ^{129}I and ^{85}Kr, and their translation into monetary terms. These releases induce long term and global impacts because of their radioactive half-life or their transfer into the environment. They were estimated using models and hypotheses (migration into the environment, dose calculations, dose-response relationship, constant population, etc.) that have been internationally agreed but which are still being debated extensively.

4.2.1 Methodological considerations

The evaluation of long term and global impacts is raising various theoretical and complex issues related to the validity of the quantitative assessment of what could be the future risks but also to the ethical position we are adopting towards future generations. However from a practical point of view, a responsible attitude implies to use in the best way the available information we have about the possible consequences of our present actions even if this information is just reflecting limited knowledge taking into account the weakness of our present instruments to assess far future consequences.

Although the concept of collective dose has not to be considered as the "best indicator", it allows to express the impact on populations in space and time and provides additional information to the evaluation of individual exposures for far future exposures. Nevertheless, one has to keep in mind the two types of arguments that have been raised against the use of collective dose to assess the impact far in the future:

- The aggregation on large population of extremely low individual doses leads to significant collective doses.

- The existence of increasing uncertainties especially with time are weakening the pertinence of estimation of impacts in the far future.

These criticisms are very valid, however any responsible attitude cannot avoid taking into account the magnitude of the release of radionuclides in the environment, the length of the period during which these radionuclides remain a source of exposure and how wide they are dispersed geographically, i.e. how large is the exposed population.

So far the use of the individual dose and collective dose concepts allow to determine the order of magnitude of the long term and global impacts and to assess if these impacts do not induce any problem in the future in terms of individual risk or public health. In fact, there is a need to consider these various indicators for evaluating the impacts because of the complexity of the problem. In this context, their translation in monetary terms is difficult: any aggregation of the complexity will reduce the dimension of the problem.

4.2.2 Order of magnitude of the impacts

Assuming a constant world population of 10 000 million people, the impacts associated with the releases of long live radionuclides (mainly ^{14}C and ^{129}I) are at the maximum of about 20 man-Sv per TWh, when integrating individual doses over 100 000 years at the global scale. In terms of individual exposure, the average doses are bellow 10^{-6} mSv/year for the various releases of the French nuclear fuel cycle associated with one year of production. For comparison, it should be kept in mind that the individual dose associated with ^{14}C which is naturally present into the environment is of the order of 1.2 10^{-2} mSv/year.

A second aspect concerns the mining residues. On the basis of an optimistic assumption, that after the closure of the site, these residues are managed in order to catch the radon emanations (over the natural background), the long term impacts are negligible. In the absence of any specific treatment, the collective dose associated with these residues is about of 200 man-Sv/TWh over a period of 10 000 years according to the estimates performed by UNSCEAR [9] for local and regional populations. In the French evaluation, assumptions have been made on the coverage of the old mines, and only the differential with the natural background of radioactivity has been considered. for these reasons, the collective dose considered in the study are far below the evaluation of UNSCEAR without specific treatment (around 200 man.mSv/TWh).

Concerning the management of radioactive waste, the evaluation of collective doses is rather limited according to the available studies in this field. In terms of individual dose, a European study (Everest project [10]) provides some evaluations for impacts associated with the storage of high level waste. For a granite storage, the maximum individual doses are of the order of $2\ 10^{-4}$ mSv/year after a period of 20 000 years in the case of the “normal evolution” of the site which is designed to receive the waste associated with the reprocessing of about 100 000 tons of spent fuel. As far as the intrusion scenarios are concerned, the maximum individual dose is of about 2 mSv/year. For both cases, the doses are associated with ^{129}I. For low and intermediate waste disposal, the maximum individual doses for the public are of about $4\ 10^{-3}$ mSv/year (impact associated with the releases of ^{3}H) during the monitoring phase (300 years) and $8\ 10^{-3}$ mSv/year above this period.

For the evaluation of the external costs, it appears that regardless of the radionuclides and the period of time considered, the individual impact remains insignificant (maximum annual individual dose estimated to be in the region of 10^{-8} mSv/year per TWh). Integrating in space and time for the population as a whole, the global external cost increases by at most 10 to 20% if a discount rate of 3% is considered. This situation is quite different without discounting the external costs of long-term impacts: in that case, the external costs should be multiplied by a factor 10. In this context, the validity of an annual discount rate and of the integration over very long period of time should be questioned.

5. Conclusion

From this assessment, it clearly appear that the main impacts are on workers operating nuclear installations, with an individual level of exposure in the range of a few millisieverts. As far as the public is concerned, the level of exposure for the present generations (0-100 years) due to direct radioactive releases from facilities into the environment is extremely low: a very small fraction is added to natural background exposure and the corresponding individual risk can be considered as negligible.

Concerning the long term and global consequences, the exposures of the future generations remain negligible in terms of individual risk and one can state that the total collective exposure that can be estimated by integration over a large time period and the global population will never be a public health problem as the worldwide collective exposure for a given generation (about 30 years) is at the most a few man-sieverts.

A remaining issue is the potential for major accidents which may have significant health and environmental consequences. The prevention of such accidents calls for a high level of vigilance and on-going improvement of safety.

In this context, the monetary evaluation of these impacts points out some issues for the economic theory. The development of contingent valuation surveys during the last decade has improved the valuation process of these impacts, but further analyses should be focused on the validity of aggregating indicators using discounting procedure for very long term and global impacts as well as indicators based on expected value in the case of major accident. Due to the complexity of the health and environmental impacts, the monetary valuation process has to be used with caution in order to reflect the various dimensions of these impacts.

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HYDROPOWER – INTERNALISED COSTS AND EXTERNALISED BENEFITS

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Abstract

The benefits of hydropower consist of the minimal level of noxious and greenhouse gas emissions, it's energy security from political instability, and its renewable, non-depletable nature. The costs of hydropower consist of negative effects on the river ecosystem and of social changes in communities in the vicinity of large projects. Public awareness of these costs has increased dramatically during the past two decades, and new hydro projects will not get approval unless adequate mitigation measures are taken to avoid, offset, or compensate for adverse environmental and social effects. To a very large extent, the hydropower industry has internalised what were previously social and environmental externalities. However, hydropower operators do not receive any compensation for the benefits, and to date their competitors (coal, natural gas, oil) have not been required to internalise their adverse environmental externalities. (emissions, depletion of supplies, and sometimes dependence on imported primary energy sources). This creates an uneven playing field, and the hydropower industry enthusiastically welcomes a discussion of this issue, and eventually measures to rectify the situation.

The IEA Hydropower Agreement has completed a major international study on the environmental and social impacts of hydropower, and one major component of this study was a Life Cycle Assessment and comparison of all the most important electricity generation technologies. The main conclusions were:

- The LCA methodology can give useful results for some specific environmental parameters, and could probably be broadened to include some social parameters.

- The LCA methodology is heavily dependent on the initial assumptions, and the outcomes for a specific parameter can have a very wide range of values, sometimes of two or three orders of magnitude.

- The LCA methodology does not currently take into account the important issue of level of service, also known as ancillary services.

- In the case of hydropower, and probably in the case of other generation technologies, a separate LCA analysis would be needed for each power plant because the results can not be generalised to the industry as a whole.

- Some environmental parameters, such as biodiversity, and some social parameters, such as esthetics or visual amenity are difficult or impossible to quantify.

- The multiple impacts of any major infrastructure project are judged not only by economic value systems (i.e. economic and financial rate of return) but also by cultural value systems (health, education, public safety, esthetics, heritage, social equity, conservation) which differ from country to country.

- The LCA methodoly is especially strong in two parameters of greatest importance to hydropower, noxious emissions and greenhouse gas emissions. It could and should be used to internalise these externalities either through imposing a cost on emitters, or paying a benefit to non-emitters, or a combination of the two.

1. Introduction

Hydropower is characterised by its complete site specificity, each hydropower project is located on a particular site with a particular topology on a particular river in a particular climate zone and in a particular eco-system. Hydropower projects can be found high in the mountains above the tree line and down in lush valleys. They are found in arctic and sub-arctic regions as well as tropical regions, in desert or semi-arid regions as well as regions with heavy rainfall. They are found in completely un-inhabited areas and in densely populated river valleys. It is not surprising that few generalisations can be made about hydropower, and that for almost every generalisation that is made exceptions can be found.

Aside from the high site specificity of hydropower, there are broad categories of hydropower projects with very differing environmental impacts, the main ones are run-of-the-river projects, projects with a large reservoir, and multi-purpose projects (irrigation, flood control, navigation). Some governments have also made distinctions between large hydro projects and small ones, but hydropower professionals in general believe that on a per kWh basis small hydro projects can have more environmental impacts than large ones. A recent study by the German Ministry of the Environment supported this conclusion, at least in the case of Germany.

In 1998, hydropower generated 2 643 TeraWatt hours of electricity (2 643 × 10^9 kWh), which was 18.4% of the world's total electricity production.[1] If the average wholesale price of electricity is taken to be about 3 cents US per kWh, then the annual 1998 production would be worth $79 billion. It would have required 1 586 million tons of coal to generate this same amount of electricity.[2]

2. The IEA Hydropower Agreement Study of Social and Environmental Impacts

The IEA Implementing Agreement for Hydropower Technologies and Programmes, as it is officially known, conducted a major international study of the Environmental and Social Impacts of Hydropower during the period 1995-2000[3]. It involved six participating countries, who held 11 international meetings and workshops. In all 112 experts from 16 countries, the World Bank, and the World Commission on Dams have participated, and 29 professional papers have been presented at meetings. One major component of this study was a sub-task entitled: "Environmental and Health Impacts of Electricity Generation – A Comparison of the Environmental Impacts of Hydropower with those of Other Generation Technologies". This sub-task was essentially an LCA comparison of all the important electricity generation technologies: coal, oil, natural gas, nuclear, biomass, hydropower, wind and solar. It will be called "the LCA sub-task" in the remainder of this paper.

The LCA sub-task did a very extensive literature survey to collect from various sources published data about the environmental impacts of the various generation technologies. It did not do any original data acquisition of its own. Although all six participating countries contributed to a greater or lesser extent, the main work was done by two organisations, Vattenfall in Sweden and Hydro Quebec in Canada. One result of their work is Table 1, below, which is a reduced version of a larger table.

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1. IEA, World Energy Outlook 2000.
2. The assumption is that it takes about 0.6 kg of coal to generate 1 kWh of electricity.
3. Available on the web-site http:\\www.ieahydro.org.

Table 1. **Emissions produced by 1 kWh of electricity based on life cycle analysis**

Generation option	Greenhouse gas emissions gm equiv CO_2/kWh	SO_2 emissions milligram /kWh	NO_x emissions milligram /kWh	NMVOC milligram /kWh	Particulate matter milligram /kWh
Hydropower	2-48	5-60	3-42	0	5
Coal – modern plant	790-1 182	700-32 321+	700–5 273+	18-29	30-663+
Nuclear	2–59	3-50	2-100	0	2
Natural Gas (combined cycle)	389–511	4-15 000+[4]	13+-1500	72-164	1-10+
Biomass forestry waste combustion	15–101	12-140	701-1950	0	217-320
Wind	7–124	21-87	14-50	0	5-35
Solar photovoltaic	13–731	24-490	16-340	70	12-190

The table clearly shows that the amount of noxious emissions (SO_2, NO_x, NMVOC – non methane volatile organic compounds, and particulate matter) and greenhouse gas emissions are much less for hydropower, nuclear, and wind, than they are for fossil fuels (coal and natural gas). This fact has been widely known for a long time, and in a qualitative sense it is nothing new. However, being able to quantify these results by means of an LCA is an important step forward, and should, in turn, enable policy makers to quantify the taxes or other policy instruments that they may use to internalise these externalities.

3. Extension of the LCA approach

Further research work to expand the LCA approach to issues of energy security would certainly be welcomed by the hydropower industry (and no doubt by the other renewable energy industries as well). Although many rivers cross international boundaries, and there are some international hydro power projects, most hydro is a national resource and not dependent on political stability of energy exporting countries. However, the amount of hydroelectricity available in a given year is dependent on the amount of rainfall, and can easily

4. The sulphur content of natural gas when it comes out of the ground can have a wide range of values, when the hydrogen sulphide content is more than 1%, the gas is usually known as "sour gas". Normally, almost all of the sulphur is removed from the gas and sequestered as solid sulphur before the gas is used to generate electricity. Only in the exceptional case when the hydrogen sulphide is burned would the high values of SO_2 emissions occur.

vary by more than 20% from year to year. Currently, reservoir levels in most of Brazil and in the US North-West are low, and the electricity produced is less than in average years. Hence energy security depends not only on political factors, but also on the weather.

Similarly, work to expand the LCA approach to issues of energy depletion would also favour all forms of renewable energy. Hydropower is quintessentially a long-term investment, some plants have now been in continuous operation for more than 100 years. Current economic thinking and net present value calculations give little importance to cost or benefit streams that occur more than 15 or 20 years from the present. Yet the hydropower industry abounds with examples of communities or utilities which constructed hydro plants thirty or forty years ago, and made little profit while the debt was being amortised. However, they now own money spinning assets, which unobtrusively and reliably produce thousands or millions of dollars of revenue year after year. A net present value calculation thirty or forty years ago when the decision was made to build such plants would have rated them as relatively poor investments compared to others which might have had a higher internal rate of return. Some of the current economic theories do not appear to give adequate guidance when dealing with the long term future, whether it is long term benefit streams or long term costs, such as depletion of fossil energy resources. An extension of the LCA methodology to deal with these long-term issues would be an important step forward.

4. Hydropower – Internalised costs and externalised benefits

The past two decades have seen a paradigm shift not only in environmental issues, but also with respect to the rights of individuals, small groups, and local communities affected by large infrastructure projects. The hydropower industry has changed accordingly, and the way hydro projects are done today is very different from the way they were done 20 years ago. Full environmental assessments are done, and extensive consultations are held with affected groups and communities, in a broad participatory decision making process. Most (but not all) adverse impacts of a project are established during the planning stage, and the design of the project will avoid impacts if possible, or include mitigation measures, or provide for compensation. Modern hydro projects have to pay for most of their social and environmental impacts, otherwise they will not get government approval to proceed. In some cases reserve funds are set aside to pay for impacts which are discovered once the project is completed and in operation. It is not uncommon for hydro projects to be given the stewardship responsibilities for an entire river basin and to have to pay for the impacts of other polluters. For example, the Bonneville Power Administration in the North Western USA pays not only for the environmental impacts of the hydropower industry, but also for those due to forestry, agricultural run-off, and other industries.[5] In most jurisdictions, the hydropower industry has fully internalised the cost of adverse environmental and social impacts, and in some case it has gone beyond.

In most countries, the main competitors of hydro power, i.e. coal and natural gas generation, do not have to pay for their environmental impacts, and especially their emissions of noxious gases and greenhouse gases. This could be corrected by requiring them to pay for their emissions, i.e. internalising their externalities, or by paying a bonus to renewable energies for not emitting, or by a combination of the two. Creating a level playing field is important for the hydropower industry, and the LCA methodology could make a very useful contribution in this respect.

5. Experience with the LCA approach

The LCA sub-task has documented various aspects of the LCA approach, which need to be taken into account in decision making.

5.1 Wide range of values

Firstly, the numbers in Table 1 give a very wide range of values, up to three orders of magnitude in some cases. This is due to the specific site in some cases, and due to the variety of assumptions that can be made about the materials and manufacturing processes going into a project. Cement can be made in gas fired kilns or coal fired kilns, the gas and coal can have a high sulphur content or a low sulphur content. The electricity used in manufacturing processes can come from thermal sources or renewable sources. All these assumptions have a determining effect on the outcome of the LCA. The emissions of solar voltaic are a good illustration, because a lot of electricity is required to manufacture a photovoltaic cell. If this electricity is derived from coal, the greenhouse gas emissions during the manufacture of the photovoltaic cell can be of the same order of magnitude as natural gas combined cycle generation.

The range of values for a generation technology as a whole is so wide that it does not provide very useful numbers for policy making. Instead, a specific project will have to be selected first, and the LCA calculations made for that project. Even then, it may not be known in advance from which country the steel and cement will be bought, and the resulting calculations could have

5. Personal communication, the Bonneville Power Administration has a budget of about US $ 400 million per year for environmental improvements in its service area.

widely different answers as a result. If the LCA approach becomes more widely used, the LCA parameters for all the main materials will be more readily available, and with the use of computers the overall LCA for construction of a plant can be calculated more easily.

The operation of a plant will also cause variations in environmental impacts. The operation of a hydropower reservoir can make significant differences both in the reservoir itself and downstream. The effects of a coal or natural gas plant will depend on the sulphur content and other characteristics of the fuel that is being used at any given time. These variations during operation give rise to a range of values, even for a specific plant, and they are more difficult to handle from the regulatory point of view.

5.2 Level of service

The consumer expects electricity to be available on demand, when he or she turns on a switch the light should go on. Since electricity itself can not be economically stored, utilities face a major challenge in continually adjusting the amount generated to be equal to the amount demanded. Some generation technologies, such as hydro or diesel, can easily adjust the amount of electricity produced, whereas others such as coal or nuclear can only operate economically at a constant output level. Yet others, such as solar or wind are subject to interruption, either because the sun doesn't shine or the wind doesn't blow. The ability of a generation technology to easily match the amount produced to the amount demanded by consumers is known as level of service. In the power industry the words ancillary services are used. In privatised electricity markets, the exact market rules and the generation mix in the market area determine whether, and to what extent, the ancillary services are compensated. In some cases these market rules are satisfactory, and in others less so. The main point is that economic mechanisms do exist to deal with the differing levels of service. The LCA methodology does not in its present form include the issue of level of service, and it is probably not necessary to do so because of the existence of these other economic tools. However, the interpretation of LCA results should always be done with the caution in mind that differences in levels of service are not reflected.

5.3 Quantification of some environmental and social parameters

Some environmental parameters, such as biodiversity, and some social parameters, such as esthetics or visual amenity are difficult or impossible to quantify. For the hydropower industry, the biodiversity issue is important, especially as related to fisheries of migratory species (salmon, trout), and to the creation of new wetlands downstream or the inundation of existing wetlands by a reservoir. The question of esthetics and visual amenity can be quite controversial, some will be very happy with a beautiful new lake which can be used for recreational and other purposes, whereas others will be unhappy because a certain area of land is flooded. In many cases new transmission lines have to be constructed for hydro projects, and this may detract from the visual amenity of a landscape. The fact that these environmental and social parameters are difficult or impossible to quantify means that an LCA analysis can not tell the complete story about the desirability or otherwise of a proposed project. Consequently, the LCA may be helpful in internalising some of the costs, but it can not internalise all of the important effects.

5.4 Cultural value systems

The multiple impacts of any major infrastructure project are judged not only by economic value systems (i.e. economic and financial rate of return) but also by cultural value systems (health, education, public safety, esthetics, heritage, social equity, conservation) which differ from country to country.

An LCA analysis may provide numerical values for several environmental and social parameters, and it may then be possible to attach a monetary value to each numeric environmental parameter, and then to internalise the cost. Once each parameter has been translated into a monetary value, it becomes easy to compare and manipulate them. One effect will cost 0.5 cent per kWh, another 0.1 cent, a third 1.5 cent, etc. Decisions about trade offs, cost of mitigation measures, taxes or surcharges, etc. become straightforward.

When dealing with cultural value systems there is no quantitative methodology for comparing either the costs or the benefits of a project against each other. Bringing electricity to a town or village for the first time may start a whole chain of events leading to increasing prosperity: light in the evening may stimulate adult education, refrigeration allows for better storage of medicines and food, craftsmen can increase their productivity with electric tools, water may be delivered by electric pumps, etc. To do this, it may be necessary to construct a coal fired generation plant, to build a transmission line through a scenic landscape, and over the years a few unfortunate persons in a large population may be electrocuted. The importance attached to each of these factors, and the ultimate decisions made, will be based on cultural value systems, and will differ from country to country. Even if some of these parameters can be quantified, for example the increased literacy rate and the accidental electrocution rate, they cannot be compared against each other. Even if the LCA methodology enables a number to be attached to an environmental or social factor, the next step of attaching a monetary value to that number may not possible, and then the internalisation of that factor is also not possible.

6. Conclusion

The LCA methodology is especially strong in two parameters of greatest importance to hydropower, noxious emissions and greenhouse gas emissions. It could and should be used to internalise these externalities either through imposing a cost on emitters, or paying a benefit to non-emitters, or a combination of the two. If the LCA methodology can be extended to other issues such as security of energy supply and depletion/renewability, the hydropower industry would join the other members of the renewable energy community in supporting this wider application.

At the same time, there are limits and challenges facing LCA methodology and the process of going from a quantified environmental parameter to a monetary value. Some important environmental and social parameters are difficult or impossible to quantify. Some other parameters may be quantifiable, but the next step of going from an environmental quantity to a monetary value is difficult or impossible. Ultimately, decisions about approval of projects or internalisation of their costs to society are based not only on economic values, but also on cultural values. These cultural values differ from country to country and are not easily incorporated into a quantitative framework.

The IEA Hydropower Agreement welcomes and encourages all efforts to more fully internalise the costs to society of electricity generation. These efforts should apply to all producers of electricity, at the present time the hydropower industry has internalised its costs and some of its competitors have not.

LIFE CYCLE ASSESSMENT OF RENEWABLES : PRESENT ISSUES, FUTURE OUTLOOK AND IMPLICATIONS FOR THE CALCULATION OF EXTERNAL COSTS

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Abstract

In principle, Life Cycle Assessment (LCA) is certainly appropriate for estimating external costs of renewables, since major environmental impacts of the latter are generated in phases of the life cycle other than use. In practice however, several issues still remain. They are related to the availability and quality of Life Cycle Inventory (LCI) data, to the fast technological development of renewable energy technologies (RET), to the existence of many different applications of the latter and to a strong dependency on local conditions. Moreover, a “static” picture of present technologies is not enough for policy indications. Therefore some kind of dynamic LCA is needed. These LCA issues are reflected in the calculation of external costs.

First, the paper discusses these issues on the examples of two main technologies, namely Photovoltaics (PV) and wind. Second, it discusses the results of ExternE for these two specific technologies and gives an outlook for the future. Future needs for a better use of LCA as a support tool for the calculation of external costs are identified. Finally, a new research project funded by the European Commission focused on LCI of renewables is briefly introduced and presented.

1. Introduction

In principle, Life Cycle Assessment (LCA) is certainly appropriate for estimating external costs of renewables, since major environmental impacts of the latter are generated in phases of the life cycle other than use. For example, solar energy technologies have zero emissions during the use phase; a significant share of the environmental impacts of wind is generated during the production of the wind turbines. The CO_2 emissions of biomass energy plants are compensated by the CO_2 absorption during the tree growth. All this justifies the use of a life cycle approach in general and of LCA in particular as a supporting tool for the estimation of external costs of renewable energy technologies.

However, in practice there are several open issues. Some main problems are listed here:

- Availability and quality of LCA data. Many studies of LCA of renewables have been carried out in the past. However, large differences in results are reported in literature. This is discussed more in detail later on for the case of photovoltaics (PV).

- Some renewable energy technologies are under fast technological development (e.g. PV). Therefore, a "static picture" of present technologies is not enough. Some kind of dynamic LCA and of technological forecasting are needed in order to provide useful recommendations for policy.

- Some RET have several different application scenarios. For instance, PV can be used in independent Solar Home Systems (SHS) with electric battery storage, in open-field power plants or integrated in buildings. In the latter case, they might play a multi-functional role, as they substitute building cladding materials, the may act as sun-shading (and thus energy-saving) systems. Finally, they might also be used as mini-hybrid systems with recovery of heat.

- All renewables strongly depend on local resources. Solar technologies depend on the local solar irradiation, wind and biomass depend on the local specific resource.

- As shown later, significant life cycle impacts of PV and wind are generated during the use of (conventional) electricity for their production. This implies that LCI results are strongly dependent on the local electricity mix used for calculations. This rather reflects the present technological status of a National system than of the technology itself.

All these aspects obviously strongly influence the final result in terms of external costs per produced kWh. The issues for PV and wind and their implications for the calculation of external costs are discussed in the next paragraphs.

2. Photovoltaics

2.1 Introduction

Several LCI or LCA studies of PV systems have been published in the period 1989-1998 in Germany, Switzerland, Netherlands, UK, US, Australia, Japan, Italy [1,3,6,10,12-15]. Large differences in results emerge from these studies, which were not immediately explainable at first glance. In 1997, an International Workshop entitled “Environmental Aspects of PV Power Systems” was organised at the Dept. of Science, Technology and Society of the University of Utrecht within the activities of the International Energy Agency Implementation Agreement on Photovoltaics, (IEA PVPS Task 1 – see for instance [2]. The goal of the workshop was precisely to collect most of the up-to-date LCA information on PV systems and to try to understand such important differences arising from different studies. Another international workshop followed in Keystone, Colorado, the year after, organised by the US Brookhaven National Laboratory. As an outcome of these workshops and of their own research activities, several authors have reviewed and discussed the energy requirements for the production, the energy payback time (EPBT) and the contribution for CO_2 mitigation of PV systems [4,7,5]. These authors have reviewed the main literature and have identified min. and max. values for primary energy requirements for the production of PV systems. As a consequence, they have been able to determine min. and max. estimates for EPBT, for energy yield ratios (EYR) and for specific CO_2 emissions over the life cycle of PV systems. Moreover, they have highlighted the importance of the so-called “Balance of System” (BOS), in particular for future PV systems.[1] Some of their results are summarised and discussed in the next sections.[2]

1. The BOS is the ensemble of mechanical structures and electrical devices other than the module necessary for the operation of PV systems (e.g. supporting structures, electric inverters, etc.).
2. This part is mostly taken by earlier publications by the author [Frankl & Gamberale, 1998, Alsema, Frankl & Kato, 1998].

2.2 Energy requirements for the production of PV systems

2.2.1 Crystalline silicon

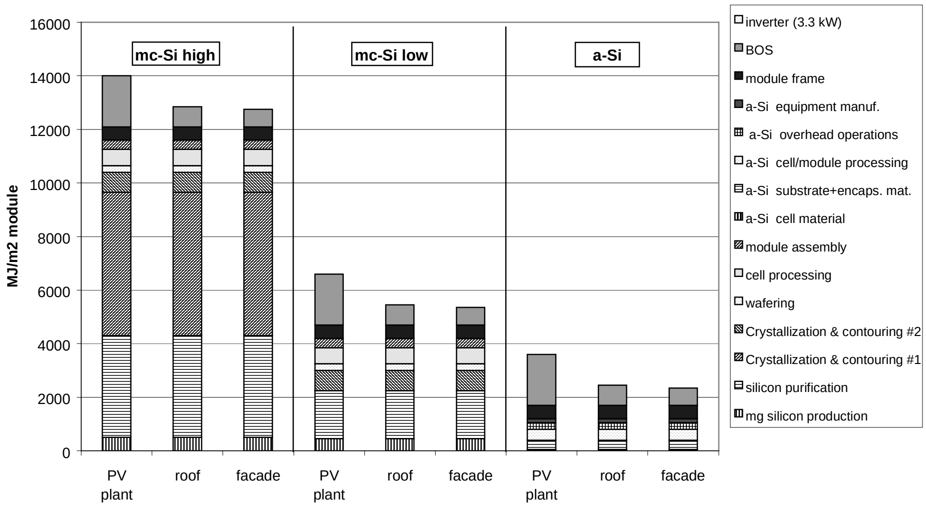
The study by [4] points out that the published estimates (as of 1997) for the energy requirement of present-day crystalline silicon vary considerably: between: 4 200-11 600 MJ/m^2 for multi-crystalline silicon (mc-Si); and between 6 000-13 900 MJ/m^2 for single-crystalline silicon (sc-Si). Partly, these differences can be explained by different assumptions for process parameters like wafer thickness and wafering losses. However, the most important source of differences is the energy requirement estimation for the silicon feedstock used to produce PV wafers. Currently the majority of PV cells are made from off-spec silicon that is rejected by the micro-electronics industry. As a matter of fact, the major source of uncertainty is the preparation of silicon feedstock from electronic industry scraps, involving two crystallisation steps. The present manufacturing energy requirement very strongly depends on:

- The allocation criteria for the primary crystallisation step.
- The silicon content of the cell.
- The specific energy consumption rates for silicon purification.

This situation is depicted in Figure 1, which shows the primary energy requirements for "present" (1997) mc-Si PV systems highlighting the large difference between the max. estimate (mc-Si "high") and the min. one (mc-Si "low". For comparison, also the energy requirements for amorphous silicon (a-Si) are reported. Moreover, the results for the BOS for three different applications (PV plant in the open field, roof integration and facade integration) are also shown. The latter figures on a-Si and BOS are discussed in the next sections.

In the future the introduction of solar-grade silicon preparation processes might significantly reduce energy requirements down to 2 600 MJ/m^2 and make the discussion about one or two crystallisation steps obsolete [4]. The significantly lower expected energy consumption values will mainly be caused by three factors, namely: i) a much higher silicon mass efficiency (that is a much better use of silicon feedstock input per kWp output, ii) internal recycling and iii) less specific energy consuming processes (i.e. faster Czochralsky and/or directional solidification processes).

Figure 1. **Primary energy requirements for silicon PV systems (1997)**



2.2.2 Amorphous silicon

The differences in published estimates for the manufacturing of amorphous silicon modules (710-1 980 MJ/m^2) can be explained by the choice of substrate and/or encapsulation materials, and by whether the overhead auxiliary energy use and energy consumption for equipment manufacturing are taken into account or not. The cell material itself accounts for only a few percent of the total energy requirements. The best estimate for present primary energy requirement for amorphous silicon manufacturing is around 1 200 MJ/m^2 [7]. Assuming a 6% module stabilized efficiency, this corresponds to specific energy requirement of 20 MJ/W$_p$, which is significantly lower than the one of present crystalline silicon modules (35-96 MJ/W$_p$ for mc-Si). However, lower efficiencies and thus higher BOS requirements can cancel out this advantage, at present and in the future. The potential for improvement is lower than for crystalline silicon modules (max. 30% energy requirement reduction) with current encapsulation materials.

2.3 The importance of BOS

P. Frankl in [6] has carried out a detailed analysis of building-integrated PV systems and highlighted the importance of taking into account the BOS in LCI calculations. For the comparison of PV systems two major categories are identified, namely “conventional” installations (array field PV power plants), and PV systems in buildings. The latter can be further classified into subcategories, corresponding to the part of the building on which the PV system is applied (terrace or flat rooftop, tilted roof, facade, etc.). Furthermore, the classification depends on whether the PV system is mounted on existing structures (retro-fit systems) or designed together with a new building (integrated installations). Finally, integrated hybrid systems with heat recovery are also considered. Mean European values for the energy requirements and emissions related to the production of materials are used for calculations.

The study has taken into account several applications on rooftops and building facades. It also has included for comparison the analysis of a large PV plant in the open field, namely the 3.3 MW power plant in Serre, Italy.

The results show that the primary energy content of a PV power plant is in the range of 1 900 MJ/m^2. Whi1e most of the systems in buildings have a total primary energy content of around 600 MJ/m^2. The high value for power plants is caused by the high amount of concrete and steel needed for this kind of installation in open fields. It is not expected to drop significantly in the future. On the contrary PV applications in architecture profit from the bui1ding structures. The analysis also has shown that in the future the primary energy needed to install building-integrated systems might well further drop down to around 400 MJ/m^2 for tilted roofs and 200 MJ/m^2 for facades. These improvements can be obtained by reducing absolute quantities of materials and/or using large fractions of recycled materials (especially in the case of aluminium). It is worth reminding that the actual roof and facade systems analysed were almost the first pilot cases of PV building integration in several countries. Therefore, it might be assumed that the use of materials has not been optimised. Furthermore, it is worth noticing the significant contribution of module frames in present-day systems. Its wide range of energy content (300-770 MJ/m^2) is due to large differences in the amount of a1uminium used for the frames. In any case, PV modules are expected to be frameless for all future applications.

2.3.1 Future installations – Possible optimisations

In the future, PV installations in buildings will likely be designed taking into account the full life-cycle of materials. This is necessary for an energy-conscious, energy efficient and environmentally sound design of the systems. Two approaches can be followed, *namely*: to minimise absolute quantities of materials and to use a large fraction of recycled, secondary materials. Figure 2 shows the possible primary energy content of future optimised PV systems. The scenario depicted is characterised by the following assumptions:

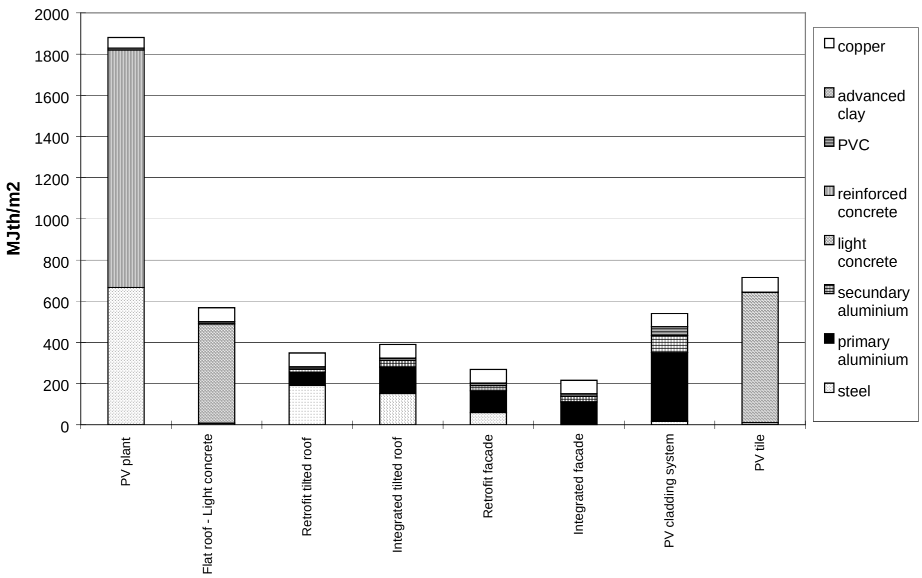
- Future installations will contain 80% of secondary aluminium. This strongly decreases energy consumption for most PV systems in buildings.

- Light concrete supporting structures will likely be used for PV systems on flat roofs, both for economic reasons and for the simplicity of installation and maintenance.

- An advanced type of clay will be used for PV tiles, which allows energy consumption to be reduced by about 30% [6].

If all the above mentioned factors are taken into account, the comparison between the BOS energy content of PV plants and PV systems in buildings becomes radically favourable to the latter, as clearly illustrated in Figure 2.

Figure 2. **Possible future BOS primary energy content of optimised PV systems**



2.4 Energy profiles of present and future silicon PV systems

2.4.1 Present systems (1997)

Figure 1 shows the primary energy requirements for present PV systems. As a consequence of the high energy requirement for crystalline silicon module manufacturing, the contribution of BOS is of minor importance, at least in the case mc-Si “high”. As a matter of fact, in this case the Energy Pay-Back Time

(EPBT)[3] is slightly higher than eight years, even if the system is installed in a place with a relatively high sun radiation, such as Central Italy.[4] Because of the large contribution of PV modules, the installation of PV systems in buildings reduces the EPBT only to a limited extent (max. 18% for roofs). Facades show even worse results because of the bad exposure to the sun at these latitudes. The most effective PV system seems to be the simple installation on flat roofs [7].

However, in the case of mc-Si "low" and a-Si, the contribution of the BOS is proportionally higher. Therefore, the benefits of the integration in buildings are more significant. Even the contribution of the aluminium module frames is not negligible.

The proportional importance of BOS will increase further with future PV modules and is described in more detail in the next paragraph.

2.4.2 Future prospects

In future, the manufacturing of crystalline silicon cells will require significantly less energy. Whatever the specific technology (single- and/or multi-crystalline silicon derived from electronic industry, or solar-grade silicon), the production chain will be optimised for solar energy cell manufacturing. A much smaller amount of silicon feedstock will be required to produce a cell. The cell and module efficiencies will also increase. Also the technology of amorphous silicon is expected to improve significantly. Table 1 summarises the expected technological evolution of silicon modules adapted from P. Frankl [6] and E. Alsema *et al.* [4].

Figure 3 shows the EPBT of future multi-crystalline silicon PV systems. Results are subdivided according to the various manufacturing steps of crystalline silicon PV systems, namely the preparation of high-purity silicon feedstock, the cutting of silicon ingots into wafers, the manufacturing of cells, the assembling of modules, and the BOS. Moreover, the contribution of process electricity and primary energy content of materials are distinguished.

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3. The EPBT is the time needed for the PV system to supply the amount of energy consumed for its production. It is defined as: *EPBT (years) = Consumed primary energy for system production/Annual primary energy produced by the system.*
4. Other parameters used for calculations are: *i*) PV plant electric BOS efficiency: 85%; *ii)* efficiency of Italian electricity production mix: 39,1%; grid distribution losses: 7%; *iii)* for integrated systems, the primary energy content of the building materials substituted by the PV components have been subtracted from the BOS primary energy content.

Table 1. **Technological parameters of present and future silicon PV modules**

	Silicon PV technologies					
	Multi-crystalline		Single-crystalline		Amorphous	
	Present	Future	Present	Future	Present	Future
Cell efficiency	14%	16%	15.5%	18%	–	–
Module efficiency	12.1%	14.5%	12.7%	14.8%	6%	10%
Primary energy content for module manufacturing (MJ/m^2)	4 200-11 600	2 600	6000-13 900	3 100	1 200	840
Module lifetime (years)	25	30	25	30	10	15

As far as this is concerned it is worth noticing the large contribution of (conventional) electricity consumption for the production and purification of the silicon feedstock..

As a consequence of manufacturing and efficiency improvements, the expected EPBT of such "optimised" power plant is reduced by more than a factor three (from 8 down to 2.3 years) with respect to present power plants.

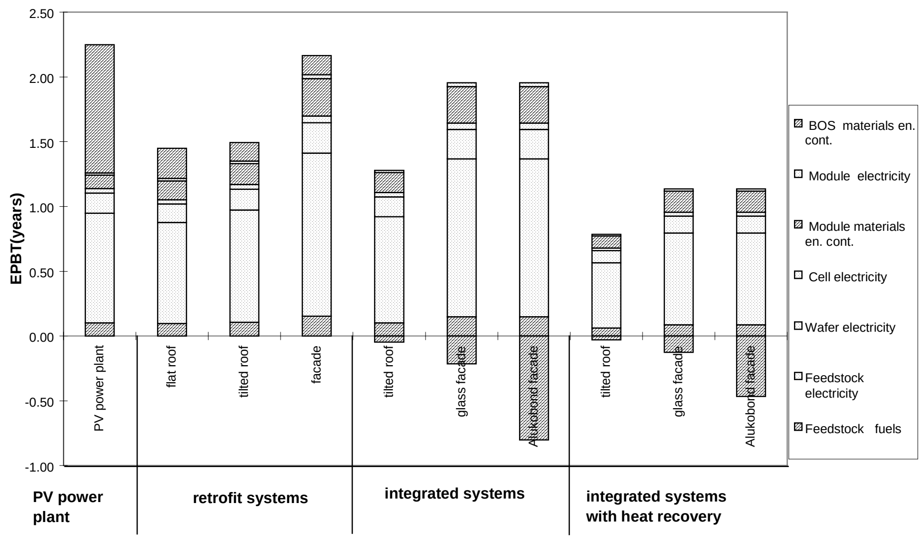
Moreover, as already mentioned the BOS plays a more important role in the total energy balance. This means that the integration in buildings gives proportionally more benefits than today. The EPBT of a fully integrated future mc-Si PV roof system is expected to be about 40% smaller than that of a future PV plant.

Moreover, the EPBT is further strongly reduced if heat recovery is taken into account. In integrated systems, at least part of the heat dissipated by the PV panels can be recovered by means of an air channel between the back-plates of the modules and the roof (or facade) itself. This air flow has a double effect: first, it allows the warm air to be used in the building for air conditioning and/or pre-heating of water; second, it cools down the cells, thus increasing their efficiency.[5] In this case, the thermal energy recovery in tilted roof can reduce the EPBT by almost a factor 3 with respect to a PV power plant. As a matter of fact the expected EPBT of an integrated tilted roof with heat recovery is lower than 10 months.[6] It is also worth noticing that the PV facades become interesting

5. In this study an annual mean value of 2 kWhth recovered heat per kWhel produced by the PV system is assumed
6. To calculate the corresponding primary energy, the substituted heat has been supposed to be produced by methane boilers.

when equipped with a heat recovery system (Figure 3). However, given the difficulties to effectively recover and use the thermal energy throughout the whole year, these results have to be interpreted with care.

Figure 3. **Expected energy pay-back times for future optimised multi-crystalline silicon PV systems**
(mean annual insolation: 1 700 kWh/m^2 year on a 30° tilted, south-oriented surface; cell efficiency: 16.0% ; module efficiency : 14.5%)



Finally, the negative BOS contribution for PV integrated facades and roofs should be remarked. This (theoretical) result reflects the possible use of PV modules to replace conventional building cladding materials. The result is particularly significant for the case of Alukobond panels[7]. The energy needed to manufacture a 1 mm thick aluminium foil is very high, larger than the BOS energy content of a PV facade-integrated systems. As a consequence, the BOS contribution is negative. The planning and design of a PV facade instead of an Alukobond facade can be therefore considered as a conceptual energy saving measure. Although purely theoretical, this result highlights the need for energy-conscious architects and engineers to be aware of the hidden energy contents of building materials.

All these results are even more significant in the case of future amorphous silicon modules. In this case the EPBT of PV roof systems is always lower than

7. An Alukobond panel is made by a sandwich of two thin aluminium foils (total thickness 1 to 3.5 mm) with a hard rubber layer in between. These panels are often used in modern office buildings.

one year. With heat recovery it further drops down to less than six months. If the substitution of Alukobond panels occurs, the total (theoretical) EPBT is zero!

2.5 Environmental benefits

Environmental benefits are evaluated here in terms of avoided emissions of CO_2. The indirect air emissions have been calculated according to the Italian electricity production (0.531 kg CO_2/kWh_{el}) and distribution (0.567 kg CO_2/kWh_{el}) mix [1].[8] For thermal energy production, a specific emission factor of 0.198 kg CO_2/kWh_{th} has been taken into account (natural gas boilers).

As expected, at present the CO_2 emissions produced during the manufacturing and installation of all systems are significant, especially if compared with the emissions avoided by the systems during their estimated life-time (25 years). This can be also indicated in terms of CO_2 yield ratio, defined as:

CO_2 yield ratio = gross CO_2 emissions avoided during lifetime of PV system/CO_2 emitted during production of PV system.

Today, conventional m-Si PV power plants save only 2.6 times the amount of CO_2 generated during their manufacturing, whereas for PV roofs with heat recovery this value increases up to 5.4. It is worth recalling that results concerning hybrid systems should be interpreted with some care, since they still require more detailed investigations and further tests. More detailed LCAs of hybrid systems are needed in the future, in order to take into account the downstream use of the recovered heat.

In any case, the environmental benefits of PV systems in buildings will significantly increase in future, as energy consumption and emissions during manufacturing of modules will strongly decrease, and at the same time efficiencies and lifetimes are expected to increase. Indeed, PV systems have a relevant potential for improving their environmental performances. CO_2 yield ratio values range from present worst case of multi-crystalline silicon power plants (around 2) to the best future cases of optimised multi-crystalline integrated silicon roofs (20 for simple roof; 34 for roofs with heat recovery; 38 if the substitution of Alukobond panels is considered). Energy and CO_2 yield ratios of future amorphous silicon are in the range of 15-120 times (see also Table 2)

The figure also indicates that the environmental benefits of integrating PV systems in buildings with respect to conventional PV power plants will

8. PV systems in buildings have no distribution losses, therefore their environmental benefits are higher.

proportionally increase in the future. For example, the CO_2 yield for an integrated PV roof with heat recovery is expected to be around three times higher than that of a conventional power plant.

The significant improvement achievable by PV systems can also be expressed in terms of specific emissions during lifetime. Today, a monocrystalline silicon PV power plant has a specific emission value of around 0.2 kg CO_2/kWh_{el}. This is mainly caused by indirect emissions deriving from high (conventional mix) electricity consumption during manufacturing of modules. In future, this value is expected to drop as low as 0.06 kg CO_2 /kWh_{el} for PV power plants and 0.04 kg CO_2/kWh_{el} for integrated PV roofs [6]. Expected figures for future a-Si are even significantly lower.

2.6 External costs of PV

Within the ExternE project, two building-integrated systems have been assessed in Germany. Table 4 summarises the main results.

As already mentioned in the earlier sections, a large share of the impacts and related external costs is caused by airborne emissions of fossil fuel electricity production. This reflects the fact that present crystalline PV production technologies are not optimised and have a large consumption of electricity. Of course much lower costs would result if one assumed that electricity used for production was supplied by PV itself or other renewable energy technologies. It is worth noticing that this is a feasible option nowadays with the progressive liberalisation of energy markets.[9]

In absolute terms, the damage estimates are low, but they are higher than the ones for wind. Again, this result reflects the large consumption of electricity produced by fossil fuels. Results based on rather high emission levels, reflecting the present electricity production mix and the present PV technologies. Although not at the highest level of estimated primary energy requirements shown in Figure 1, the figures are rather high if one takes into account the rapid technological change of PV production. This implies several recommendations for future calculations of external costs of PV, which will be described in the next paragraph.

9. Environmental-oriented firms might want to purchase just green electricity

Table 3. **Expected energy yield ratio of different future optimised PV systems**

Mean annual radiation: 1 700 kWh_{th}/m^2 on a 30° tilted, south-oriented surface; lifetime: 30 years for crystalline silicon, 20 years for amorphous silicon

	sc-Si	***mc-Si***	***a-Si***
PV plant	11.7	13.0	15.7
Retrofit flat roof	17.8	20.7	38.3
Retrofit tilted roof	17.2	20.1	42.9
Retrofit facade	11.8	13.9	23.4
Integrated tilted roof	20.4	24.3	71.3
Integrated glass facade	14.4	17.2	37.8
Integrated tilted roof with heat recovery	33.3	39.7	123.5
Integrated glass facade with heat recovery	24.7	29.7	65.5

Table 4. **External costs of PV fuel cycle**

		Life cycle				**MECU/kWh**
	Site, size	Visual impact	Global warning	Human health	Other	Sub-total
DE	Emstal-Riede, 4.8 kW, Roof	ng	0.2-7.7	0.9	0.02	1.1-8.1 (1.9-3.3)
DE	Bielefeld, 13 kW, Facade	ng	0.2-7.0	0.3	0.02	0.6-7.6 (1.4-2.8)

(Source: ExternE Infosystem).

In the calculations carried out within ExternE, some benefits of building-integration taken into account, i.e. the fact that building-integration implies no land-use impacts. Furthermore it is assumed that good architectural integration causes no visual impacts.

However, other advantages of building-integration are not taken into account. For example, substitution of building cladding materials is not included. As we have seen in the case of amorphous silicon facades, this can lead to (theoretical) negative EPBT. Moreover, the multi-functional use of PV systems in buildings is not considered. The possibility of heat recovery in mini-hybrid PV-Th collectors for water pre-heating or air heating is not taken into account. Nor is the amount of energy saving caused by sun-shading PV systems, decreasing the energy needs for room cooling. All these factors might significantly reduce the external costs of PV, in particular for future systems, for which the resource requirements and related emissions are expected to decrease significantly.

On the other hand, potential impacts of other substances released during Si purification (e.g. chlorosilanes) have been treated just qualitatively. This has to be considered in future research, at least as long crystalline silicon PV industry uses “classical”, (although optimised for larger quantities and lower purity) silicon purification and feedstock preparation methods. However, it is also worth noticing that several direct purification methods of metallurgical grade (MG-Si) into solar-grade (SG-Si) silicon are currently experimented. This development has to be taken into account very seriously, because the off-spec silicon scraps of electronic industry are no longer sufficient to supply PV demand, due to the rapid growth of PV worldwide market.

3. Wind

Wind is in some way similar to PV, in the sense that a significant share of the environmental impacts is generated during the production phase. On the contrary of PV however, other local impacts related to the power generation phase, such as noise and visual amenity, are significant.

Within ExternE, 7 wind plants in 6 different countries have been assessed. Table 5 summarises the main results.

As mentioned in the ExternE national implementation report, the damages estimated are very low, the lowest of all fuel cycles studied. In fact, the higher damages correspond to the indirect pollutant emissions produced during turbine manufacturing. However, there are large variations among damages relate to noise and visual impact, which means that wind plants have to be assessed specifically for each site.

Apart from the LCI studies carried out within the ExternE project, several other LCI or LCA studies of Wind systems have been published in past years (see for instance [11,9]). All of them report the importance of the employed materials for turbine manufacturing over the whole life-cycle.

Table 5. **Overview of results of the wind fuel cycle**

	Site, size	Power generation			Other fuel cycle stages		
		Noise	**Visual impact**	**Other**	**Human health**	**Other**	**Sub-total**
DE	Nordfriesland, 11.25 MW	0.064	0.06	ng	0.31	0.03-1	0.37-1.3 (0.47-0.67)
DK	Tunø Knob, 5 MW off-shore	0.004	ng	0.009	0.5	0.1-3	0.6-3.6 (1-1.6)
DK	Fjaldene, 9 MW	0.02	0.2	0.02	0.3	0.1-2	0.6-2.5 (0.9-1.3)
ES	Cabo Villano, 3 MW	0.008	ng	0.95	0.8	0.02-0.7	1.7-2.7 (1.8-1.9)
GR	Andros, 1.6 MW	1.12	ng	0.14	0.9	0.03-1.14	2.2-3.3 (2.4-2.6)
NO	Vikna, 2.2 MW	ng	ng	0.003	0.4	0.06-2.1	0.5-2.5 (0.5-1.1)
UK	Penrhyddlan, 31 MW	0.07	ng	0.2	0.8	0.03-1.3	1.2-2.4 (1.3-1.5)

Source: ExternE, Vol. 10, "National Implementation", p. 598.

In particular, I wish to mention and discuss the recent Environmental Product Declaration (EPD) of a wind farm carried out by Sydkraft in Sweden. On one hand an EPD has certainly to be acknowledged with great favour, for various reasons, including:

- The commitment of a Private utility towards the use of LCA for external communication and "green marketing" of its products and services.

- The fact that the multi-stakeholder approach of EPD, involving all important actors from the very beginning of the certification process, guarantees good rules on how to carry out the LCA and on how to report its results. Credibility of results is further guaranteed via the certifying institution.

However, with respect to the use of these data for the calculation of external costs, there are also some limits:[10]

- Results are reported in mid-points (e.g. kg of CO_2-equivalents, kg of CFC-11eq, etc.) and not in end-points.

10. Of course, this does not apply if the LCI data, which are available in principle only to the certifier, are published entirely.

- In the Swedish EPD, results are reported in aggregated manner over the whole life cycle. As far as this is concerned, it is worth mentioning that the forthcoming pilot Italian EPD-system will require data to be presented in clearly separate manner for each phase (production, use, end-of-life).

Therefore, at the time-being, these data are of limited usefulness for the calculation of external costs.

4. Future outlook

From the previous sections, it clearly appears that LCA is certainly appropriate as a main supporting tool for the calculation of the external costs of renewables. However, it is also clear that the current level of detail is not enough, and that several refinements of analysis will be needed in the future.

4.1 PV

Present calculations of external costs of PV systems are clearly too limited. In the future, many more studies shall be done. They should take into account several aspects, including:

4.1.1 PV modules:

- Enlarging the scope of the analysis to other types of semiconductors, in particular thin films.

- Taking into account the fast technological improvement of production processes both for crystalline silicon cells and for thin film modules.

- Considering optimised silicon feedstock production methods, both based on the "classical" Siemens process for the purification of MG-Si into electronic-grade silicon (EG-Si) and on emerging alternative direct purification processes of MG-Si into solar-grade silicon (SG-Si).

- As far the Siemens process is concerned, taking into account all impacts of silicon purification, including the ones related to other emissions just treated qualitatively in ExternE

- Including the dismantling and recycling phases of modules (some LCI studies already exist).

4.1.2 Integration in buildings:

- Analysing different types of building integration and related BOS efficiencies.

- Assessing the quality of data about employed materials for BOS.

- Making sensitivity analysis on the use of primary vs. recycled materials.

- Taking into account the potential multi-functional use of PV systems in buildings (e.g. for sun-shading or heat recovery) and assessing the relative credits.

4.1.3 Local aspects:

- Assessing local aspects (solar radiation and energy mix), which cannot be generalised. Therefore, different cases shall be explicitly shown or rules on how to make sensitivity analysis shall be provided.

- Making Sensitivity analysis on the use of the "conventional" energy mix in a specific country *vs.* the use of hydro and/or wind for PV module production. It is worth reminding that this a very concrete option within the current and future market liberalisation process.

4.2 Wind

The crucial issue for LCA of wind systems is the quality of LCI data of materials. This issue might be solved either by:

- The diffusion of more and more precise public or half-public[11] data-bases, such as the recent Italian National data-base I-LCA on materials, energy, transport and waste management systems, or the IISI database on steel production.

- The use of precise specific data, as foreseen for example in Environmental Product Declarations (EPD).

Of course, local impacts of wind, e.g. noise and visual amenity, cannot be generalised; therefore site-specific assessments shall be carried out.

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11. Only for experts in the field.

5. The new project *"eclipse"*

To conclude I wish to briefly introduce a new research project in this area, financed by the European Commission and called *ECLIPSE* ("ECological Life cycle Inventories for present and future Power Systems in Europe"). *ECLIPSE* will last 2 years, from January 2002 to December 2003 and will be carried out by 8 research partners (5 research institutes and 3 utilities) from 7 different European countries:

- Ambiente Italia - Istituto di Ricerche, Italy (co-ordinator).
- DLR - Deutsches Zentrum fuer Luft- und Raumfahrt, Germany.
- ESU Services, Switzerland.
- IER/Univ. Stuttgart (Germany).
- KEMA Nederland, The Netherlands.
- EdF (Électricité de France), France.
- Fortum, Finland.
- Vattenfall, Sweden.

The general objective of the research project *ECLIPSE* is to overcome some current limitations of the use of Life Cycle Inventories (LCI) for energy modelling and planning. More specifically the goal is to provide potential users with:

- A coherent methodological framework, with application-dependent methodological guidelines and data format requirements related to the quantification of environmental impacts from power generation in Europe based on a life cycle approach.

- A harmonised set of public, coherent, transparent and updated LCI data on new and decentralised power systems, in a format which will make them comparable to existing data of other energy technologies, easily adaptable to local conditions and technological improvement and up-datable.

In particular, the main objective is to fill the current gap of consistent and reliable data on present and future new and decentralised technologies, by creating a harmonised and coherent LCI data-set, which is of crucial importance for any energy modelling, forecasting and planning. The work will cover about 100 different configurations of PV, wind, fuel cells, biomass and CHP technologies. Sensitivity analysis to tackle with rapid technological improvement and local conditions will be carried out. As far as this is concerned, an accompanying manual illustrating the different technologies and showing how to make sensitivity analysis will be provided.

Project results will increase the credibility, diffusion and exploitation of LCI as a support tool for energy-environment-economy-modelling (like ExternE) and planning as well as for other uses.

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Session 3

COMPARATIVE ASSESSMENTS IN ELECTRICITY AND TRANSPORTATION

LCA AND EXTERNAL COSTS IN COMPARATIVE ASSESSMENT OF ELECTRICITY CHAINS. DECISION SUPPORT FOR SUSTAINABLE ELECTRICITY PROVISION?

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1. Introduction

The provision of energy and electricity plays an important role in a country's economic and environmental performance and the sustainability of its development. Sustainable development of the energy and electricity sector depends on finding ways of meeting energy service demands of the present generation that are economically viable, environmentally sound, and socially acceptable and do not jeopardize the ability of future generations to meet their own energy needs.

As liberalised electricity markets are becoming widespread, according to neo-classical welfare economics, getting the prices right is a prerequisite for market mechanisms to work effectively towards sustainable development.

Life Cycle Assessment (LCA) and external cost valuation are considered to offer opportunities to assist energy policy in a comprehensive comparative evaluation of electricity supply options with regard to the different dimensions of sustainable energy provision as well as in the implementation of appropriate internalisation strategies.

The paper addresses life cycle assessment and external cost analysis carried out for selected electricity systems of interest under German conditions. Results from a comprehensive comparative assessment of various electricity supply options with regard to their environmental impacts, health risks, raw materials requirements as well as their resulting external cost will be summarised. The use of LCA based indicators for assessing the relative sustainability of electricity systems and the use of total (internal plus external) cost assessment as measure of economic and environmental efficiency of energy systems will be discussed. Open problems related to life cycle analysis of energy chains and the assessment of environmental damage costs are critically reviewed, to illustrate how in spite of existing uncertainties the state of the art results may provide helpful energy policy decision support. The paper starts with some remarks on what the concept of sustainability in terms of energy systems means.

2. The concept of sustainable development: What does it mean for the energy system?

According to the Brundtland Commission, and the Rio Declarations, the concept of "sustainable development" embraces two intuitively contradictory demands, namely the sparing use of natural resources and further economic development. The Brundtland Commission defines sustainable development as a "development that meets the needs of the present generation without compromising the ability of future generations to meet their own needs".

Even if this definition has arisen against a background of environmental and poverty problems, it nevertheless represents an ethically motivated claim which is derived from considerations of fairness with future generations in mind. The challenge is to simultaneously help to deliver economic prosperity, to reduce and eventually eliminate poverty, to provide environmental quality and social equity and to maintain the natural foundations of life.

Therefore, the aim of sustainable development is to bequeath future generations with a stock of natural resources which will enable them to satisfy their needs at least at the level we enjoy today. This general definition of sustainability, which is acceptable to many, is not very specific about how we can guarantee satisfying the needs of future generations, for example with reference to the energy supply. It is both vague and open-ended and therefore leaves room for different interpretations.

Any attempt to define the concept of sustainability in concrete terms can only be sound if – as far as the material-energetic aspects are concerned – it takes the laws of nature into account. In this context the second law of thermodynamics which the chemist and philosopher Wilhelm Ostwald called "The law of happening" [Das Gesetz des Geschehens] acquires particular significance. The fundamental content of the second law of thermodynamics is that life and the inherent need to satisfy requirements is vitally connected with the consumption of workable energy and available material.

Within the context of defining the concept of sustainability in concrete terms the need to limit ecological burdens and climate change can certainly be substantiated. It becomes more difficult when confronted with the question of whether the use of finite energy resources is compatible with the concept of “sustainable development”, because oil and natural gas and even the nuclear fuels which we consume today are not available for use by future generations. This then permits the conclusion that only the use of “renewable energy” or “renewable resources” is compatible with the concept of sustainability.

But this is not sound for two reasons. On the one hand the use of renewable energy, e.g. of solar energy, also always goes hand in hand with a claim on non-renewable resources, e.g. of non-energetic resources and materials which are also in scarce supply. And, on the other hand, it would mean that non-renewable resources may not be used at all – not even by future generations. Given that it is, therefore, obviously impossible to pass on un-changed the non-renewable resource base, the important thing within the meaning of the concept of sustainable development is to bequeath to future generations a resource base which is technically and economically usable and which allows their needs to be satisfied at a level at least commensurate with that which we enjoy today.

However the energy and raw material base available is fundamentally determined by the technology available. Deposits of energy and raw materials which exist in the earth’s crust but which cannot be found or extracted in the absence of the requisite exploration and extraction techniques or which cannot be produced economically cannot make any contribution towards securing the quality of life. It is therefore the state of the technology, which turns valueless resources into available resources and plays a joint part in determining their quantity. As far as the use of limited stocks of energy is concerned this means that their use is compatible with the concept of sustainability as long as it is possible to provide future generations with an equally large energy base which is usable from a technical and economic viewpoint. Here we must note that in the past the proven reserves, i.e. energy quantities available technically and economically, have risen despite the increasing consumption of fossil fuels. Moreover, technical and scientific progress has made new energy bases technically and economically viable, for instance nuclear energy and part of the renewable energy sources.

As far as the environmental dimension of sustainability is concerned, the debate should take greater note of the fact that environmental pollution, including those connected with today’s energy supply, are primarily caused by anthropogenic flows of substances, by substance dispersion i.e. the release of substances into the environment. It is not, therefore, the use of the working potential of energy which pollutes the environment but the release of substances connected with the respective energy system, for instance the sulphur dioxide or carbon dioxide released after the combustion of coal, oil and gas. This becomes clear in the case of solar energy which, with the working potential – solar radiation – it makes available is, on the one hand, the principle source of all life on earth but is also, on the other hand, by far the greatest generator of entropy, because almost all of the sun's energy is radiated back into space after it has been devalued as heat at the ambient temperature. Since its energy, the radiation, is not tied to a material energy carrier, the generation of entropy does not produce any pollution in today's sense of the word. This does not, of course, exclude the release of substances and associated environmental pollution in connection with the manufacture of the solar energy plant and its equipment.

The facts addressed here are of such particular significance because this entails the possibility of uncoupling the consumption of energy and the pollution of the environment. The increasing use of working potential (energy) and a reduction in the burdens on the climate and the environment are not, therefore, a contradiction in terms. It is the emission of substances that have to be limited, not the energy flows themselves, if we want to protect the environment.

In addition to expanding the resource base available, the economical use of energy or rather of all scarce resources is, of course, of particular significance in connection with the concept of "sustainable development". The efficient use of resources in connection with the supply of energy does not only affect energy as a resource since the provision of energy services also requires the use of other scarce resources including, for instance, non-energetic raw materials, capital, work and the environment. The efficient use of all resources as can be derived from the concept of sustainability also corresponds to the general economic principle, however. Both allow for the conclusion that an energy system or an energy conversion chain for the provision of energy services is more efficient than another if fewer resources, including the resource environment, are utilised for the energy service.

In the economy costs and prices serve as the yardstick for measuring the use on scarce resources. Lower costs with the same use mean an economically more efficient solution which is more considerate on resources. The argument that can be raised against using costs as a criterion for evaluating energy systems is that the external effects of environmental damage for instance are not currently incorporated in the cost-calculations. This circumstance can be remedied by an internalisation of external costs. Without addressing the problems associated with external cost valuation here, the concept of total social costs that is combining the private costs with the external ones could serve as a suitable yardstick for measuring the utilisation of scarce resources. Total social costs could therefore serve as an integrated indicator of the relative sustainability of the various energy and electricity supply options and it would be appropriate if, in this function, they were again to be afforded greater significance in the energy policy debate. Furthermore, cost efficiency is also the basis for a competitive energy supply in order to secure the energy side of economic development and adequate employment and it is also the key to avoiding intolerable climate change. Both of these issues are central aspects of the concept of “sustainable development”.

Following this clarification of the concept of sustainable development with regard to the supply of energy we will now like to examine various electricity production options as regards their contribution towards a sustainable development of energy supply. The assessment will be based on a set of sustainable development indicators, including emissions to the environment, the requirement of both energetic and non-energetic non-renewable resources, health impacts and economic performance.

3. LCA results – a first comparison of energy systems with a view to sustainability

The approach of Life Cycle Assessment (LCA) provides a conceptual framework for a detailed and comprehensive comparative evaluation of electricity supply options with regard to their resource, health and environmental impacts as important sustainability indicators. Full scope LCA considers not only the direct emissions from power plant construction, operation and decommissioning, but also the environmental burdens and resource requirements associated with the entire lifetime of all relevant upstream and downstream processes within the energy chain. This includes exploration, extraction, fuel processing, transportation, waste treatment and storage. In addition, indirect emissions originating from material manufacturing, the provision and use of infrastructure and from energy inputs to all up- and downstream processes are covered. As modern technologies increasingly tend to reduce the direct environmental burdens of the energy conversion process, the detailed assessment of all life cycle stages of the fuel chain is a prerequisite for a consistent comparison of technologies with regard to sustainability criteria.

The LCA was carried out for a set of important electricity generation’s option, which is considered as representative for near-future technologies to be operated in Germany. Table 1 summarises some central technological parameters for the selected reference technologies.

The following figures and tables will summarise results for some of the key impact categories. Although based on our present level of knowledge this is not a complete and comprehensive comparison of all the indicators that are important from the point of view of sustainability, but it does provide an initial indication of the potential contribution of specific electricity supply options to a future sustainable energy system.

Table 1. **Characterisation of the reference electricity production technologies**

	Technology	Power installed	Efficiency	Life
Coal	Pulverised fuel firing	600 MW	43.0%	35 a
Lignite	Pulverised fuel firing	800 MW	40.1%	35 a
Gas combined-cycle	Combined-cycle	777.5 MW	57.6%	35 a
Nuclear (PWR)	Actual PWR	1375 MW	34.0%	40 a
PV (poly) PV (amorphous)	Poly-crystalline amorphous	5 kW 5 kW	9.5%[1)] 4.5%[2)]	25 a 25 a
Wind	5.5 m/s[2)]	1.5 MW	–	20 a
Hydro	Run-of-river	3.1 MW	90%[3)]	60 a

1) System-efficiency.
2) Average windspeed p.a.
3) Efficiency of turbines.

3.1 Cumulative energy requirements

The generation of electricity is associated with partly quite intensive energy consumption by power plant construction, and – in the case of fossil and nuclear energy sources – also by fuel supply and waste treatment. The cumulative energy requirement as shown in Table 2 for different power generation systems includes the primary energy demand for the construction and decommissioning of the power plant as well as for the production and supply of the respective fuel. The energy content of the fuel input is not included in the figures.

The indirect primary energy input per produced kWh of electricity for hydro, wind and nuclear systems is in the range of 0.04 to 0.07 kWh. For natural gas and coal the necessary energy input per produced unit of electricity is in the range of 0.17 to 0.30 kWh which is basically determined by the energy required for the extraction, transport and processing of the fuel. The corresponding figures for today's photovoltaic systems are 0.67 to 1.24 kWh. This is also reflected in the energy amortisation time which is approximately 6 to 12 years in the case of photovoltaic systems using today's technology and is by far the longest compared to any of the other systems.

Table 2. **Cumulative energy requirements (CER) and energy payback periods (EPP)**

	CER (without fuel) [$kWh_{Prim}/kWh_{el.}$]	**EPP [months]**
Coal (43%)	0.3	3.6
Lignite (40%)	0.17	2.7
Gas CC (57.6%)	0.17	0.8
Nuclear (PWR)	0.07	2.9
PV (poly) PV (amorph.)	1.24 0.67	141 76
Wind (5.5 m/s)	0.07	6.4
Hydro (3.1 MW)	0.04	10.9

3.2 Raw material requirements

Electricity production involves consumption of non-energetic raw materials such as iron, copper or bauxite. Sustainability also means the efficient use of such resources. Table 3 shows the cumulated resource requirements of the power generation systems considered here for selected materials. It covers the raw material requirements for power plant construction, fuel supply, and for the supply of other raw materials. The table only includes a small part of the various raw materials required and is therefore not a complete material balance. However, results indicate that the relatively small energy density of solar radiation and of the wind leads to a comparatively high material demand. This high material intensity for wind and solar energy is an important aspect with regard to the generation costs.

3.3 Pollutant emissions

Figure 1 compares the cumulative emissions of selected pollutants of the power generation systems considered. It is obvious that electricity generated from solid fossil fuels (hard coal and lignite) is characterised by the highest emissions of SO_2, CO_2 and NO_x per unit of electricity, while emissions from the nuclear system, hydropower and wind are comparatively low. Electricity generation from natural gas causes emissions that are significantly lower than those from coal-fired systems.

Table 3. **Total life cycle raw material requirements**

	Iron [kg/GWh$_{el.}$]	**Copper [kg/GWh$_{el.}$]**	**Bauxite [kg/GWh$_{el.}$]**
Coal (43%)	2 310	2	20
Lignite (40%)	2 100	8	19
Gas CC (57.6%)	1 207	3	28
Nuclear (PWR)	420-445	6	27
PV (poly) PV (amorph.)	5 350-7 300	240-330	2 040-2 750
Wind (5.5 m/s)	3 700	50	32
Hydro (3.1 MW)	2 400	5	4

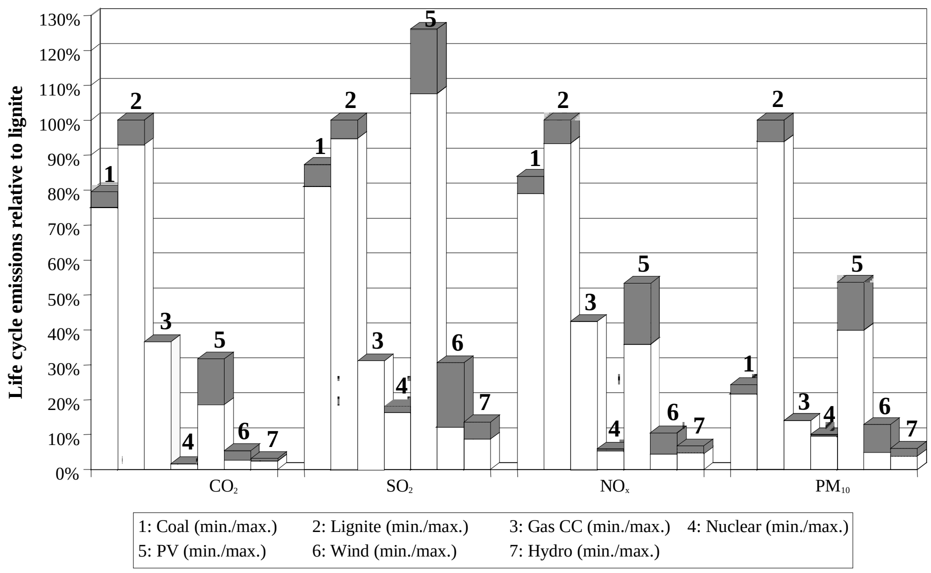
Although there are no direct emissions from the electricity generation stage, the high material requirements for the production of PV panels result in cumulative CO_2 and NO_x emissions of the photovoltaic fuel chain that are close to those of the gas fuel chain and far higher as SO_2 and particulates are concerned.

It might be mentioned that the indirect emissions from material supply and component manufacturing are determined to a great extent by the emissions of the respective energy mix. Due to the high proportion of fossil energy in the German electricity mix, results shown in Figure 1 are not directly applicable to other countries with a different energy mix.

3.4 Human health risks

Electricity generation from fossil fuels, nuclear energy or renewable energy sources leads to an increased level of air pollution, or to an increased exposure of the population to ionising radiation, which in turn might cause an increased risk to the health of the exposed population. Using the emissions from the life cycle assessment as a starting point, health risks resulting from the operation of the energy systems considered here are assessed following a detailed impact pathway approach. For the quantification of health effects from pollutants relevant for fossil energy systems (fine particles, SO_2, Ozone) dose-effect models have been derived from recent epidemiological literature. The risk factors recommended by the International Commission on Radiological Protection (ICRP) are used to estimate effects from ionising radiation.

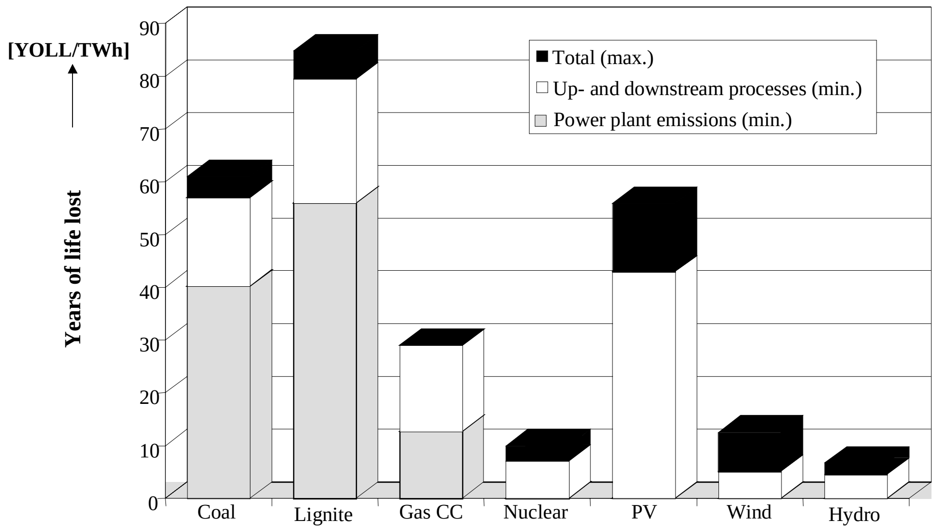
Figure 1. **Total life cycle emissions**



The application of the ICRP risk factors to the very small individual dose resulting from long term and global exposure is, however, a matter of particular uncertainty and might lead to an overestimation of effects. Results of the risk assessment are summarised in the next figure. The increased death risk is presented as the loss of life expectancy in Years of Life Lost (YOLL) per TWh.

Figure 2 shows that electricity generation from coal and lignite lead to the highest health risks of the power generation systems considered, while power generation from nuclear systems, wind and hydro energy is characterised by the lowest risk. Due to the high emissions from the material supply, risks from photovoltaic systems are higher than the risks from natural gas-fired power plant. Results for the nuclear fuel chain include the expected value of risk from beyond design nuclear accidents, which is small compared to the importance of major nuclear accidents in the public discussion. However, the expected value of risk is not necessarily the only parameter determining the acceptability of a technology. Different evaluation schemes that take into account risk aversion or a maximum tolerable impact might lead to a different ranking of technologies.

Figure 2. **Health risks of energy systems**



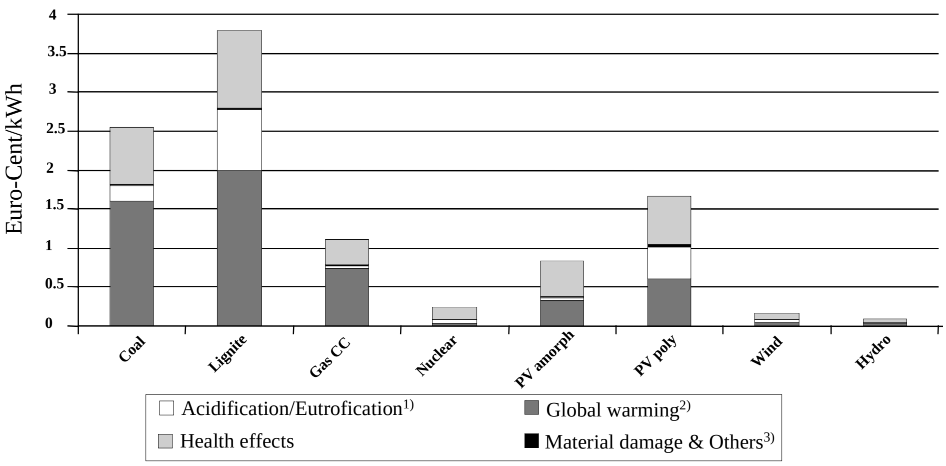
3.5 External costs

It is well accepted now that health impacts and environmental damage due to air pollution cause economic losses which are not accounted for in the electricity price (so called external costs). According to neo-classical welfare economics, external costs have to be internalised, i.e. added to the price of electricity, to achieve a full picture of the consumption of scarce resources.

External costs resulting from impacts on human health, agricultural crops and building materials are considered as quantifiable with a reasonable level of uncertainty, but impacts on ecosystems and in particular potential impacts from global climate change are hardly quantifiable based on current knowledge, so that an economic valuation of the potential impacts is very uncertain. In these cases, marginal abatement costs for achieving policy-based environmental targets (German CO_2-reduction targets in the case of global warming, and SO_2- and NO_x-targets derived from the European Commission's strategy to combat acidification for ecosystem protection) can be used to give a rough indication of the potential damage costs. Using the detailed Life Cycle Inventories as guiding input the marginal external cost estimates are based on applications of the "impact pathway approach", established in the EU ExternE Project. The "impact pathway approach" models the causal relationships from the release of pollutants through their interactions with the environment to a physical measure of impact determined through damage functions and, where possible, a monetary valuation of the resulting welfare losses. Based on the concept of welfare economies, monetary valuation follows the approach of "willingness-to-pay" for improved environmental quality. The valuation of increased mortality risks from air pollution is based on the concept of "Value of Life Year Lost".

External costs calculated for the reference technologies are summarised in Figure 3. For the fossil electricity systems, human health effects, acidification of ecosystems, and the potential global warming impacts are the major source of external costs. Although, the power plants analysed are equipped with efficient abatement technologies, the emission of SO_2 and NO_x due to the subsequent formation of sulphate and nitrate aerosols leads to considerable health effects due to increased "chronic" mortality. A comparison between the fossil systems shows that health and environmental impacts from the natural gas combined cycle plant are much lower than from the coal and the lignite plant.

Figure 3. **External costs from different electricity generation technologies operated in Germany**



1) Acidification/eutrofication: valuation based on marginal abatement costs required to achieve the EU "50%-Gap Closure" target to reduce acidification in Europe.
2) Global warming: valuation based on marginal CO_2-emissions in Germany by 25% in 2010 (19 Euro/tCO_2).
3) Others include noise and crop losses.

External costs arising from the nuclear fuel chain are significantly lower than those estimated for the fossil fuels. Most of the radiological impacts are calculated by integrating very small individual doses over 10 000 years. The application of the ICRP risk factors in this context is at least questionable, and most likely leads to an overestimation of effects. The impact resulting from emissions of "conventional" (i.e. SO_2. NO_x, and particles) air pollutants from the nuclear fuel chain dominate the external costs. The external costs calculated from the expected value of risk from beyond design nuclear accidents are surprisingly small compared to the importance of major nuclear accidents in the public discussion.

External cost of photovoltaic, wind and hydropower mainly result from the use of fossil fuels for material supply and during the construction phase. External costs from current PV application in Germany are higher than those from the nuclear fuel chain and close to those from the gas fired power plant. Impacts from the full wind and hydropower life cycle are lower than those from all other systems, thus leading to the lowest external costs of all the reference technologies considered. While the uncertainties in the quantification of external costs are still relatively large, the ranking of the considered electricity options is quite robust.

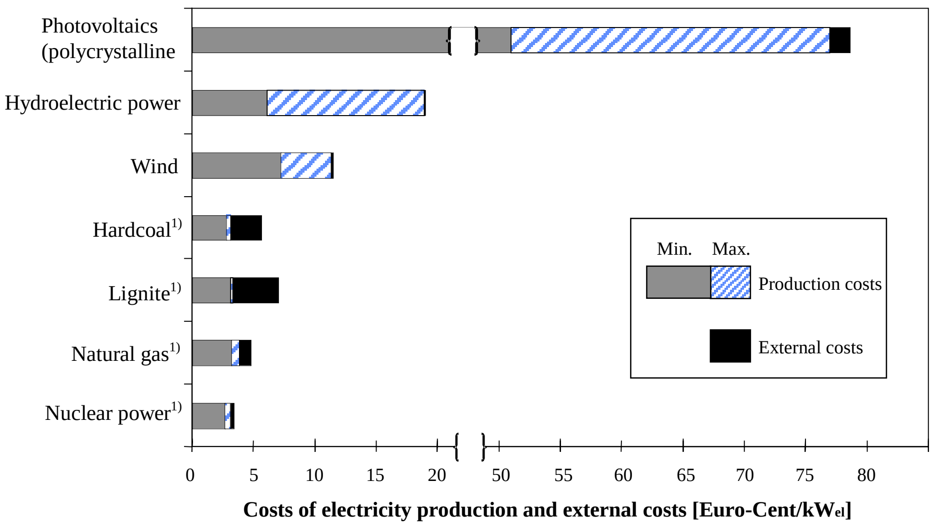
3.6 Power generation costs

Costs in general might be considered as a helpful indicator for measuring the use of sparse resources. It is thus not surprising that a high raw material and energy intensity is reflected in high costs. The power generation costs shown in the next figure indicate that power generation from renewable energies is associated with higher costs – much higher in the case of solar energy – than those resulting from fossil-fired or nuclear power plants. However, as discussed above, the private costs alone do not fully reflect the use of scarce resources. To account for environmental externalities, external costs have to be internalised, i.e. added to the private generation costs. Figure 4 shows that the external costs resulting from the electricity generation of fossil fuels amount from 30% (natural gas) to about 100% (lignite) of the generation costs, while for the other technologies the external costs are only a small proportion of generation costs.

Figure 4 shows that the external costs resulting from the electricity generation of fossil fuels amount from 30% (natural gas) to about 100% (lignite) of the generation costs, while for the other technologies the external costs are only a small proportion of generation costs. The internalisation of external costs might lead to competitiveness of some wind and hydropower sites compared to fossil fuels, but do not affect the cost ratios between the renewable and the nuclear systems. On the other hand it is obvious, that the full internalisation of environmental externalities would improve the competitive advantage of nuclear energy to fossil electricity production.

The results of energy and raw material requirements, life cycle emissions, risks and both external and generation costs discussed so far are based on the characteristics of current technologies. It is expected that technical development will result in a further reduction in costs and in the environmental burdens of power generation. However, this applies to all the power generation technologies considered here. Preliminary results for future systems indicate that the ranking of technologies with respect to total costs is quite robust.

Figure 4. **Total costs of various electricity generation technologies operated in Germany**



1) Base-load.

4. Uncertainties and open problems

In spite of considerable progress that has been made over the last years, especially with regard to the bottom up modelling of the full impact pathways and the monetary valuation of health effects, the life cycle inventory based quantification and valuation of environmental impacts is still linked to partly large uncertainties. Uncertainties related to impact assessment and valuation are certainly larger compared to those of the LCA inventory. Besides the data and model uncertainties the estimation of external cost is faced with systematic uncertainties arising from lack of knowledge, and value choices that influence the results.

Lack of knowledge is the single most important reason for the large uncertainties related to the quantification of climate change damage costs. This suggests using abatement costs based on the standard-price approach to achieve a specific greenhouse gas reduction target as the second-best solution to make this impact visible in the external cost estimates. Large uncertainties are in the exposure-response functions for health impacts and the "Value of Life Year Lost". As in the past improved knowledge, e.g. of the influence of particle composition on the chronic mortality from fine particles could lead to different health related damage costs. But an assessment can always only reflect the current knowledge.

The estimates of external costs are influenced also by the discount rate chosen, to account for damage costs in the future as well as by valuation of damages in different parts of the world. The uncertainties stemming from these assumptions can be best dealt with by sensitivity analysis.

The application of the impact pathway approach and the monetary valuation methods suggest relatively low external costs for both beyond design accidents in a nuclear power plant and radioactive waste deposit. The impact from a single beyond design accident can be very large, but normalised to the electricity generation over the power plants lifetime, the expected value of risk (i.e. probability times consequences) is low, a fact which is even robust against uncertainties in the accident probability.

Some people argue that the use of the expected value of risk to estimate the external cost of a low probability events with large consequences is an open problem. They consider the maximum damage from a single incident as an important key criterion on its own, which has to be included in the impact valuation of technologies. Empirical evidence supporting this kind of reasoning is still missing.

With respect to the uncertainties of external cost estimates it is important to notice, that these uncertainties are of relevance to any other valuation scheme. It is however remarkable, that in spite of these uncertainties and changing background assumptions, external cost estimates at least indicate a robust relative ranking for the key electricity production technologies. However, care has to be taken to acknowledge existing uncertainties and to not take external cost estimates out of the given context when using them in a policy context.

5. LCA and external costs for policy support

Life Cycle Assessment and the external as well as the total cost approach can provide valuable decision support for a wide range of policy relevant issues:

- Assessment of technologies currently used to identify deficiencies and potentials for improvement and corresponding research issues.

- Comparison of current and future energy supply options with respect to their health and environmental impacts, resource requirements and with respect to their compliance with sustainability indicators.

- Cost-benefit-analysis of environmental policy measures.

- Extension of national green-accounting frameworks.

There are several examples of successful applications in these areas, for instance the use of the method for cost-benefit analysis of desulphurisation plants attached to large coal fired power plants in Europe. The comparison of the costs in € per tonne of SO_2 avoided and the avoided damage costs due to the reduced emissions shows that the benefit clearly outweighs the costs. This even remains valid, if the mortality impacts – the impact category considered to involve the highest uncertainty – would be neglected. So, even if the damage cost estimates are varied within the uncertainty range, the conclusion would not change.

In the context of the liberalisation of the European electricity market the concept of internalising external effects by means of technology-specific price adders has been discussed. The idea is to derive science based recommendations on the height of price adders for electricity production by different technologies from LCA and external cost research. This approach is expected to get the prices right in competitive markets and to ensure that the use of the environment is accounted for in the market mechanism.

But the use of simple price adders for each technology appears inappropriate, besides the still existing large uncertainties of the external cost estimates. This is due to the fact that health and environmental impacts depend heavily on the concrete technical design and the location of a specific power plant. For this reason the use of regionally differentiated pollutant-specific damage costs is recommended for the internalisation of external costs due to airborne pollutants. These damage costs should at least be differentiated by country. Table 4 presents damage costs per tonne of pollutant emitted in Germany.

Table 4. **Specific damage costs in € per tonne of pollutant emitted in Germany (reference year 1998)**

	€ per tonne emitted
SO_2	5 650
NO_x	5 030
PM_{10}	8 700
NMVOC	1 770

The advantage of such a pollutant-oriented approach is that it gives a direct incentive for reducing the emissions. This is expected to outweigh the disadvantage of a higher effort for recording the emissions of every pollutant.

In the case of CO_2 and other greenhouse gases, due to the high uncertainties involved in estimating damage costs, it is recommended to base internalisation instruments (such as emission certificates or a greenhouse gas tax) on greenhouse gas reduction targets and the associated marginal abatement cost.The risks of low probability accidents should be integrated into the monetary accounting system by introducing liability insurance obligations.

It can be concluded that LCA and quantification of environmental costs give valuable input to the assessment of the relative sustainability of different electricity production technologies in spite of the current knowledge gaps. These methods thus can contribute to a rational decision support. LCA inventories provide very useful information concerning the evaluation of resource requirements of different electricity production options. External cost estimates represent an aggregated indicator of environmental performance. Together with the private costs, total costs can serve as an integrated indicator for the overall resource consumption and in this respect for relative sustainability of the different energy options.

As far as applicable, the approach of monetary valuation can be considered to be the most appropriate way of weighting and aggregating different impact categories when assessing the environmental impact or the resource intensity of energy systems. Monetary values based on individual and social preferences for a wide range of health and environmental impacts have been derived from empirical work. The use of such values for aggregation has advantages compared to weighting schemes derived from expert or personal judgement, as the weighting is based on “measured” preferences. And last, but not least, monetary values have the advantage of being more illustrative than “utility points” or other artificial measures, although the results might be the same.

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LIFE-CYCLE ANALYSIS AND EXTERNAL COSTS IN TRANSPORTATION

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Overview

- A life cycle analysis of emissions of air pollutants and greenhouse gases from transportation fuels.
- External costs of motor-vehicle use
- Social cost comparison of alternative vehicles and transportation modes

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An LCA of Transportation Fuels

Fuels and electricity life cycle
- End use of fuel
- Dispensing of fuels
- Fuel distribution
- Fuel production
- Feedstock transport
- Feedstock production

Vehicles and infrastructure life cycle
- Materials production
- Vehicle assembly
- Operation and maintenance
- Secondary fuel cycle for transport modes
- Infrastructure construction

Why is LCA important?

Compare CO_2 emissions from end use vs fuel cycle, for motor vehicles (as a percentage of fossil-fuel CO_2):

	End use	Whole fuel-cycle
US	22	30
OECD-Europe	18	24
World	14	19

Upstream emissions from alternative transportation fuel cycles in year 2010 (g/10^6-BTU-fuel)

Fuel-->	**EtOH**	**LPG**	**Diesel**	**CNG**	**RFG**	**MeOH**	**CH_2**	**EtOH**
Feed-->	*wood*	*oil&NG*	*oil*	*NG*	*oil*	*NG*	*NG*	*corn*
CO_2	(9,470)	8,688	14,672	10,167	20,842	29,788	90,662	70,174
NMOCs	32.3	16.2	14.9	6.5	43.9	25.4	9.6	230.7
CH_4	72.0	148.5	209.3	285.9	220.7	342.2	379.2	198.7
CO	185.8	37.8	60.2	33.8	58.6	68.1	58.4	199.1
N_2O	19.8	0.3	0.8	0.3	0.6	1.1	1.5	52.3
NO_x	258.6	51.7	71.3	63.1	80.2	153.7	132.5	412.4
SO_x	10.9	27.1	54.7	15.6	50.4	38.0	52.4	94.9
PM	43.5	8.7	18.6	3.6	18.3	8.2	9.3	122.4
CO_2eq	(1,095)	11,436	18,410	15,787	24,876	36,490	97,953	91,467

Upstream emissions from alternative transportation fuelcycles in year 2010 (% of end-use emissions)

Fuel-->	**EtOH**	**LPG**	**Diesel**	**CNG**	**RFG**	**MeOH**	**CH_2**	**EtOH**
Feed-->	*wood*	*oil&NG*	*oil*	*NG*	*oil*	*NG*	*NG*	*corn*
CO_2	-15%	14%	21%	20%	31%	48%	7430%	108%
NMOCs	26%	39%	21%	42%	28%	34%	77%	187%
CH_4	392%	1270%	4596%	164%	1958%	5592%	6663%	1082%
CO	16%	3%	8%	3%	4%	6%	16%	17%
N_2O	53%	1%	25%	1%	2%	3%	″	140%
NO_x	129%	27%	8%	33%	46%	77%	63%	206%
SO_x	180%	636%	276%	560%	777%	612%	894%	1569%
PM	1426%	475%	51%	249%	261%	269%	547%	4010%
CO_2eq	-1%	15%	25%	23%	29%	46%	4151%	109%

Emissions from the vehicle life cycle, year 2010

Pollutant	Emissions (g/lb)		Emissions (g/mi)		Emissions (% of end use)	
	LDGVs	*HDDVs*	*LDGV*	*HDDV*	*LDGVs*	*HDDVs*
CO_2	3,838	3,285	83	161	30%	5%
NMOCs	1.91	1.85	0.04	0.09	5%	3%
CH_4	8.03	6.93	0.17	0.34	381%	177%
CO	7.19	8.14	0.15	0.40	2%	1%
N_2O	0.09	0.08	0.00	0.00	1%	3%
NO_x	9.51	8.70	0.20	0.43	25%	1%
SO_x	10.18	10.05	0.22	0.49	280%	59%
PM	6.99	7.11	0.15	0.35	532%	23%
CO_2eq	3,865	3,293	83	161	23%	5%

Lifecycle GHG emissions from light-duty vehicles (g/mi and % change)

	Fuel cycle only	**Fuel+materials**
Baseline gasoline vehicle	*455.1*	*554.4*
Diesel (0.032% S)	-34.6%	-30.5%
Natural gas (CNG)	-27.5%	-21.3%
LPG (P95/BU5)	-23.4%	-19.2%
Ethanol from corn	-10.6%	-8.7%
Ethanol wood and grass	-76.5%	-62.7%
Battery EV, coal plants	-16.5%	-6.2%
Battery EV, NG plants	-60.4%	-42.2%
FCEV, Methanol from NG	-46.7%	-38.7%
FCEV, Hydrogen from water	-92.1%	-75.9%
FCEV, Hydrogen from NG	-61.4%	-50.8%

External cost of motor-vehicle emissions in urban areas of the US (10% change in emissions) (1991 $/kg)

	PM10		**NOx**		**SOx**		**CO**		**VOCs, O3**	
	Low	*High*	*Low*	*High*	*Low*	*High*	*Low*	*High*	*Low*	*High*
Health	1.5	187.5	1.5	23.3	4.4	90.9	0.0	0.1	0.1	1.6
Visibility	0.1	5.2	0.2	1.5	0.5	5.3	0.0	0.0	0.0	0.1
Crops	n.e.	n.e.	0.0	0.0	n.e.	n.e.	0.0	0.0	0.2	0.3
Forests, materials	n.e.	n.e.	0.0	0.0	n.e.	n.e.	0.0	0.0	0.2	0.3
Climate change ($/Mg)	0.0	0.000	1.20	20.0	0.0	0.000	0.60	10.0	1.5	25.0
Total MVs	14.3	192.7	1.9	24.9	10.9	96.3	0.0	0.1	0.5	2.3
Total MVs+U	12.8	162.9	1.8	23.5	4.9	37.1	0.0	0.1	0.5	2.0
Total MVs+U+RD	1.6	32.8	1.8	23.5	4.9	37.1	0.0	0.1	0.5	2.0

External cost of oil use in the US, 1991 $/end-use gallon

	Gasoline vehicles	**Diesel vehicles**	**All vehicles**
Strategic Petroleum Reserve - low	0.0004	0.0006	0.0005
Strategic Petroleum Reserve - high	0.0052	0.0064	0.0054
Defense costs - low	0.0045	0.0056	0.0047
Defense costs - high	0.0505	0.0623	0.0529
Pecuniary externality - low	0.0285	0.0350	0.0298
Pecuniary externality - high	0.0596	0.0730	0.0623
Price-shock cost to GNP - low	0.0189	0.0231	0.0198
Price-shock cost to GNP - high	0.1889	0.2314	0.1976
Water pollution - low	0.0023	0.0026	0.0023
Water pollution - high	0.0076	0.0084	0.0078
All costs - low	*0.055*	*0.067*	*0.057*
All costs - high	*0.312*	*0.382*	*0.326*

The marginal cost of noise from a 10% increase in VMT, for different types of vehicles on different types of roads, in urbanized areas (1991$1000-VMT)

Base case	**Interstate**	**Other freeways**	**Principal arterials**	**Minor arterials**	**Collectors**	**Local roads**
LDAs	2.96	4.25	1.18	0.57	0.07	0.00
MDTs	8.50	13.20	7.02	5.37	1.05	0.00
HDTs	16.69	30.80	20.07	29.93	4.93	0.00
Buses	6.36	9.77	7.18	6.42	1.22	0.00
MCs	17.15	27.03	8.71	4.67	0.56	0.00
Low case						
LDAs	0.11	0.18	0.04	0.01	0.00	0.00
MDTs	0.40	0.66	0.32	0.18	0.01	0.00
HDTs	0.81	1.62	1.22	1.77	0.06	0.00
Buses	0.35	0.58	0.38	0.22	0.00	0.00
MCs	0.66	1.13	0.27	0.09	0.00	0.00
High case						
LDAs	40.11	56.02	16.20	9.35	6.04	0.44
MDTs	114.76	173.38	96.05	84.93	78.84	12.13
HDTs	225.61	404.82	269.27	414.17	319.22	92.04
Buses	86.15	128.60	98.66	105.33	108.00	12.84
MCs	232.47	355.73	119.64	76.65	50.08	2.73

Non-monetary externalities of motor-vehicle use in the US (10^9 1991 $)

Cost item	**Low**	**High**	**Q**
Accidental pain, suffering, and death, not accounted for by economically responsible party	10.2	120.0	A3, D
Travel delay, imposed by other drivers (includng accidents) that displaces unpaid activities	30.8	119.5	A2
Air pollution: human mortality and morbidity due to particulate emissions from vehicles[b]	16.7	266.4	A1
Air pollution: human mortality and morbidity due to all other pollutants from vehicles	2.3	17.1	A1
Air pollution: human mortality and morbidity, due to all pollutants from upstream processes	2.3	13.0	A1
Air pollution: human mortality and morbidity, due to road dust	3.0	153.5	A1
Air pollution: loss of visibility, due to all pollutants attributable to motor vehicles	5.1	36.9	A1
Air pollution: damage to agricultural crops, due to ozone attributable to motor vehicles	2.1	3.9	A1
Air pollution: damages to materials, due to all pollutants attributable to motor vehicles	0.4	8.0	B [A1]
Air pollution: damage to forests, due to all pollutants attributable to motor vehicles	0.2	2.0	B [A2]
Global warming due to fuel-cycle emissions of greenhouse gases (U. S. damages only)	0.7	7.4	A1, B
Noise from motor vehicles	0.5	15.0	A1
Water pollution: health and environmental effects of leaking motor-fuel storage tanks	0.1	0.5	D
Water pollution: environmental and economic impacts of large oil spills	2.0	5.0	C [A1]
Water pollution: urban runoff polluted by motor-vehicle oil, and by highway de-icing	0.7	1.7	D
Pain and suffering and other non-monetary costs due to crimes related to motor-vehicle use	1.7	6.1	A3
Nonmonetary costs of injuries and deaths caused by fires related to motor-vehicle use	0.0	0.2	A3

External costs of EVs versus gasoline vehicles (cents/mile)

	Battery EVs			Gasoline ICEVs		
	low	*high*	*best*	*low*	*high*	*best*
Noise	0.00	1.20	0.04	0.00	1.60	0.05
Externalities of oil use	0.02	0.12	0.04	0.22	1.25	0.40
Climate change	0.00	0.13	0.06	0.01	0.19	0.09
Air pollution	0.02	0.21	0.07	0.19	2.32	0.75
TOTAL	0.05	1.67	0.21	0.41	5.36	1.29

Social cost of EVs *vs*. gasoline vehicles (cents/mi)

	Difference in costs		
	low	*high*	*best*
Private lifecycle costs	0.0	30.00	10.00
Noise	0.00	-0.40	-0.01
Externalities of oil use	-0.20	-1.12	-0.36
Climate change	-0.00	-0.06	-0.03
Air pollution	-0.17	-2.11	-0.69
Total externalities	-0.37	-3.69	-1.09
Social cost	-4	30	9

External costs and subsidies for different passenger transport modes

(cents per vehicle mile, except last row is cents per passenger mile)
[Numbers in brackets are my best estimates]

Cost item	Gas auto	Electric auto	Transit bus	Light rail	Heavy rail
Air pollution	[2.0] 0.8 to 13	1.5	[20.0] 5.4 to 123	5?	5?
Oil use, water pollution	[0.8] 0.3 to 1.5	0.4	[4.0] 1.5 to 8.7	1?	1?
Noise	[0.2] 0.01 to 2.0	0.15	[2.0] 0.5 to 10.0	1?	1?
Congestion	4.0	4.0	8.0	n.e.	n.e.
Accidents	2.5	2.5	3.5	2?	2?
Marginal highway and service costs	0.1	0.1	1.5	0	0
Unpriced parking	[1?] 0 to 8	[1?] 0 to 8	0	0	0
Inefficient highway user taxes and fees, meant to cover highway costs	-2.7	0	0 (exempt from fuel taxes)	0	0
Government subsidy: operating costs minus fares, operating+rolling-stock costs minus fares, total operating+capital costs minus fares	0	0	339, 398, 465 [398]	685, 1137, 2800	372, 797, 1177
Extra private costs relative to gas auto	0	0 to 16 [8]	see subsidy	see subsidy	see subsidy
Total cents per vehicle-mile	**[8] 5 to 28**	**[18] 9 to 25**	**359 to 620 [437]**	**694 to 2,809**	**381 to 1,186**
Passengers per vehicle	assume 1.0	assume 1.0	11 (average)	26 (average)	22 (average)
Total cents per passenger-mile	**[8] 5 to 28**	**[18] 9 to 25**	**33 to 57 [40]**	**27 to 108**	**17 to 53**

Conclusions

- LCA is important in transportation, because "upstream" impacts can be significant
- Environmental external costs are dominated by the health costs of particulate air pollution
- In the comparison of the social cost of transportation alternatives, differences in external cost are not trivial, but often are small compared with differences in private costs or in financial subsidies

Round Table

HOW TO USE INTERNALISATION OF EXTERNALITIES IN POLICY MAKING?

ENERGY POLICY AND EXTERNALITIES: THE LIFE CYCLE ANALYSIS APPROACH

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1. Introduction

In the energy sector, getting the prices right is a prerequisite for market mechanisms to work effectively towards sustainable development. However, energy production and use creates "costs" external to traditional accounting practices, such as damages to human health and the environment resulting from residual emissions or risks associated with dependence on foreign suppliers. Energy market prices do not fully reflect those external costs. For example, the costs of climate change are not internalised and, therefore, consumers do not get the right price signals leading them to make choices that are optimised from a societal viewpoint.

Economic theory has developed approaches to assessing and internalising external costs that can be applied to the energy sector and, in principle, provide means to quantify and integrate relevant information in a comprehensive framework. The tools developed for addressing these issues are generally aimed at monetary valuation of impacts and damages and integration of the valued "external costs" in total cost of the product, e.g. electricity.

The approach of Life Cycle Analysis (LCA) provides a conceptual framework for a detailed and comprehensive comparative evaluation of energy supply options. For power generation technologies and transportation fuels, LCA represents a useful tool to assess the sustainability of different energy technologies and, by extension, the electric power and transport sectors of OECD countries.

1. The author would like to thank a number of IEA and NEA colleagues who provided useful comments on earlier drafts of this note, including Jonathan Pershing, Peter Fraser, Evelyne Bertel, Laura Cozzi, Lew Fulton and Giorgio Simbolotti.

This paper offers a summary of the LCA methodology and an overview of some of its limitations. It then illustrates, through a few examples, how the methodology can be used to inform or correct policy making and to orient investment decisions. Difficulties and issues emerging at various stages in the application and use of LCA results are discussed, although in such a short note, it is impossible to address all issues related to LCA. Therefore, as part of the concluding section, some issues are left open – and areas in which further analytical work may be needed are described.

2. What is life cycle analysis?

Life cycle analysis (and assessment) is a process that seeks to identify and assess the environmental, economic and social impacts associated with a product, process or activity. It does so using various specific but continuously evolving methodological tools. The assessment covers the entire life cycle of a product, process or activity from raw material production and transformation to end use and disposal, in a "cradle to grave" approach. For the case of an energy product, process or service, life-cycle analysis encompasses all segments including up-stream and down-stream processes.

Life cycle analysis:

- Identifies the segments of the fuel cycle (or chain) which have the most harmful environmental impacts (and consequently specific points in the cycle that provide opportunities for pollution prevention or control).

- Allows for an overall comparison (in a cost and benefit framework) of short- and long-term economic implications of different energy technologies or strategies factoring in environmental impacts. An aspect of this assessment is the valuation of the so-called external costs (and particularly environmental externalities of the fuel and energy technology cycles) for policy purposes.

- Identifies research and policy areas for further work.

A definition: external costs and externalities

An externality may be defined as:

A cost or benefit that is not included in the market price of a good because it is not included in the supply price or the demand price. An externality is produced when the economic activity of one (or a group of) actor(s) has a positive or negative impact on the welfare function of another actor (or group of actors) and when the former fails to be fully compensated or to fully compensate the latter for that impact. Externality is one type of market failure that causes inefficiency (see Pearce & Turner, 1989).

This definition is most often used in the context of negative environmental externalities such as air pollution which produce damages to human health, crops or materials and in which the polluter may not suffer from the direct or indirect damages. In principle, externalities may also be positive, for example, the case of a bee-farmer whose bees help pollinate the fruit trees of a nearby orchard.

Essential to the definition are both the lack of participation in the decision concerning the economic activity by one or more of the parties affected, and the absence of full compensation of the costs or benefits accruing to the receiving party. It should be noted that under this definition, environmental pollution might conceivably not be an externality if those who suffer from the negative impacts of that pollution are fully compensated.

3. The life-cycle analysis methodology

Undertaking a life-cycle analysis requires performing the following steps:

- Defining the system's boundaries.
- Identifying environmental burdens (inventory) and impacts.
- Quantifying and monetising impacts.

The methods for each of these steps are continuously evolving. Although some of these methodologies have been reviewed and standardised, the methods and assumptions used by different practitioners still vary widely, as do the underlying decision criteria for the analysis. As a consequence, the resulting valuations may show differences of an order of magnitude or more. The process of life-cycle analysis and assessment involves multidisciplinary competencies. For energy technologies and products these include:

- Engineers and technicians who can define various energy technologies and their emissions and identify all elements of the fuel cycle.
- Scientists who can identify pollutant dispersion, and who are versed in issues related to health, ecology, and materials and who can identify and measure the full chain of environmental impacts.
- Economists to monetise the impacts measured and to develop cost-benefit assessments.

3.1 Methods and criteria

3.1.1 Definition of the system's boundaries

The energy system boundaries for the analysis must be clearly defined both in terms of activities and of their geographical location. To do so, the various stages of the fuel/technology cycle must be identified. These normally include exploration, extraction, fuel preparation, transport, conversion, transmission/distribution, use, re-use and final disposal. The analysis should be extended to plant and infrastructure construction, operation and dismantling, as well as to waste product management at all stages of the cycle. A similar path may be developed for technologies that do not use fossil fuels, like wind-power generation or solar photovoltaic conversion, where environmental burdens from the processing and use of materials for the plant or technological device used are more relevant.

In some cases, fuel/energy conversion and use is easily located spatially and may be the main source of environmental burdens. However, activities upstream and downstream from actual energy conversion and use are often located elsewhere. Furthermore, energy commodities and services are normally traded – often internationally. Fossil fuels used at a given power plant may originate from many different supply sources. Fuel extraction activities themselves give rise to flows of wastes and to environmental impacts not necessarily confined to the immediate vicinity of the activity area. Some pollutants are easily transported by air and water and can cross national borders, ending up at distant locations: at each location there may be an effect of the flows of pollutants. Hence the decision concerning the system and the spatial boundaries set for the analysis is far from being a trivial one.

3.1.2 Identification of burdens and impacts

In principle, a life cycle analysis should strive to identify all environmental burdens and associated impacts of a specific energy system that fall within the system boundaries, regardless of whether they are measurable (or even considered relevant). Scientific investigation improves continuously, increasing our capability to identify and measure the real environmental impact of polluting emissions; on occasion, this may lead to a reversal of an early conclusion on the direction of impacts. Producing comprehensive lists (inventories) of burdens and impacts is therefore important not just for consistency and transparency, but also because it allows for an easier updating of the analysis as new information is produced.

This part of the analysis can be performed through the development of an "accounting framework", which identifies the various stages of each fuel/technology cycle and traces the consequences of the associated burdens through the following stages:

Fuel/technology cycle stage → Activity → Burden → Impacts

This accounting framework is easily presented as a matrix like the one in Table 1, having in the columns the four terms above, and in the rows, for each stage of the cycle, the corresponding activities, the burdens produced and the type of impacts.[2] Xs mark indicative priority impacts for further analysis.

The identification of impacts can be done through expert knowledge or through reference to the available literature on the particular segment of the system or on the burden considered. This process requires following all the links of the chain, from the emission of a pollutant (or creation of a burden) all the way to the receptor(s), and the analysis of the receptor's response.

Prioritisation of the importance of different impacts (in terms of their magnitude or of their social, economic or environmental relevance) is then necessary to isolate those impacts that are worth studying in further detail for valuation purposes. One example could be materials such as cadmium, used in the production of photovoltaic systems, whose concentration could lead to significant health impacts.

2. Detailed accounting frameworks for a number of selected fuel cycles have been developed within the ExternE project, a collaborative effort launched by the European Commission and the US Department of Energy to assess the external costs of fuel cycles. The project was implemented with the collaboration of a large number of European and US research institutes.

Table 1. **Schematic example of accounting framework**

			Impacts				
			Health		**Environment**		
Stages of fuel cycle	**Activities**	**Burdens**	Occupational	Public	Natural	Agricultural	Man-made
Fuel extraction	Mine construct.	Dust	X		X		
	Mining		X		X	X	
	Waste water management	Dissolved metals	X	X			
	Solid waste management	Solid waste dumps		X			
	….						
Fuel preparat.	Fuel cleaning		X		X		
	…						
Fuel transport	Construct. of facility		X		X		
	Loading						
	Transport		X				
	Unloading		X	X			
	…		X				
Power generation	Plant construct.		X		X		
	Operation	Primary air pollutants		X	X	X	X
		SO_x					
		NO_x		X	X	X	X
		CO_2		X	X	X	X
		….					
etc…	…..	….					

3.1.3 Impact quantification and monetisation

Monetising impacts is definitely the most complex stage of a life cycle analysis. However, methodologies to quantify and attach a monetary value to the environmental impacts have been well developed in the scientific and economic literature; many methods date back more than 40 years. For a proper assessment, a diverse body of knowledge, which among other disciplines, covers engineering, material sciences, fluid dynamics, climatology, medical and biological sciences, ecology, sociology and economics, must be put together in the impact assessment process. A broadly agreed, standardised methodology for this procedure is the Impact Pathways Methodology.[3] Using this method, the analyst follows the flow of a pollutant or environmental burden from the point of emission to the final impact on the receptors affected, and then assigns an economic value to each impact.

3. The ExternE project has made a significant contribution to the establishment of the Impact Pathway Methodology.

Atmospheric or water dispersion models are used to determine pollutant transport patterns and concentrations at various locations. For the more reactive pollutants (for instance, SO_2), the formation of other pollutants (e.g. H_2SO_4) from the originals is estimated through chemical and photochemical reaction models. Dispersion models vary from very simple to complex depending on the geographic range and the number of pollutants and chemical reactions covered. Dispersion models that simulate transport of inert pollutants are usually reliable, but become less reliable when they involve chemical reactions besides pollutant transport.

Next, the population or the various receptors exposed to the various pollutants must be defined and characterised, and the impact estimated through the use of suitable exposure- (or dose-) response functions. Exposure-response functions, $Y = f(X)$, where f is the impact function, link exposure to pollutant X (e.g. concentration of particulate in the air) to specific physical impacts Y (e.g. the number of cases of respiratory diseases due to concentration of particulate in the air). As impacts are often negative the function is also called *dose-damage function*. These are usually derived from the medical and scientific literature, and based on epidemiological studies, statistical analysis of field or laboratory data.

The specific form of the impact function depends on the precise phenomenon being investigated. Impact functions may be linear or non-linear. Impact functions may have threshold values below which the impact is zero (i.e. below the threshold, the natural response capability of the receptor – biologic organism or ecosystem – may prevent or counteract the impact). In a few cases, the response function has negative values of Y below a positive threshold value of X, indicating that the pollutant may have a positive or "fertiliser effect" at low doses. The existence of thresholds has important practical as well as policy implications. Such levels may be points at which preventative or mitigation action becomes necessary to avoid health damages or acute crises.

Very few of the known impact functions take into account synergies with other pollutants or other variables in the biological or ecological system considered, making it hazardous to transfer results from one specific case to another.

Margins of uncertainty in estimated functions are usually expressed in terms of "confidence intervals", by which the true value of the impact is said to lie with x% confidence within a band of variation centred on the mean estimate. Estimation of these impacts rests on the scientific solidity of the cause-effect relationship and on the goodness of fit of the statistical relationship. As noted above, impact or response functions may or may not include all relevant factors in the equation; both uncertainty in known variables and the fact that important variables may not be known are risks inherent in the evaluation process.

Once the impact of a measured increase in the environmental concentration of a pollutant has been quantified, the next step in the assessment process is that of economic valuation. As with other elements of the life cycle analysis, this element can range from quite simple to extremely complex.

Some impacts lead to a change in the quantity of a good which is traded (and which has a market value or price – the case of damages caused by ozone levels to commercial crops). In such a case, the valuation is straightforward: it is found by multiplying the unit price by the change in yield. Of course, if the impact is large enough to affect the market price, this price change has to be taken into account.

The valuation must include all impacts, whether they affect goods normally traded in the market or not. In fact, when performing a valuation of environmental impacts, one typically has to deal with goods that are not readily traded in the market, but which may still be vital to human welfare, such as health, the preservation of buildings, monuments, wildlife and bio-diversity, and amenities such as an unspoiled landscape.

A number of methods have been developed to value these goods. One is the *control cost method* based on pollution abatement or mitigation costs. This approach considers the value of the environmental good to be correctly approximated by the control or abatement cost sustained to restore it. According to this approach the question posed is what would be the cost to reduce drastically certain polluting emissions, to restore human health or a forest damaged by acid rains, or to clean a building covered by soot. Other useful indicators are the defensive expenditure consumers choose to sustain in order to avoid certain negative impacts (e.g. double windows to reduce noise pollution).

Another approach taken by economists in the valuation of environmental goods such as the ones mentioned, is to try to identify individual preferences through their Willingness to Pay (WTP) for an environmental good or Willingness to Accept (WTA) payment in compensation for an environmental damage suffered. This approach, chosen among others by the ExternE project, covers a variety of methods and techniques:

- One is the *contingent valuation method* (CVM), which seeks to identify consumer preferences through direct questionnaires and interviews, asking consumers how much they would be willing to pay to avoid a damage or to accept in compensation for a damage suffered. The same method can be applied to issues as varied as health risks, water quality or the aesthetic value of an undisturbed landscape.

- Another technique is the *hedonic price method*, which derives the value of some environmental characteristics from the WTP for goods in related markets. A typical case is that of different property values for homes of equivalent size and characteristics but located in neighbourhoods with different noise or air pollution levels.

- A third method is that of *travel costs*, which use expenditures in recreational activities as a proxy for WTP for natural amenities, parks, wildlife, etc.

The WTP/WTA approach can be used both for values arising from the actual use of the environment and for values arising even when no identifiable use is made of the environment, either at present or in the future. The former are called *use values* and can refer both to *direct use* (clean air in one's neighbourhood) or *indirect use* (e.g. clean air for a cousin living in Bhopal): altruistic values are in the latter group. *Non-use values*, on the other hand include cases such as the interest in preserving an unspoiled Antarctic environment, or for the conservation of a threatened wildlife species unlikely to be encountered by any given individual in their lifetime. These latter values are the most difficult to assess.

3.2 Further issues in life-cycle assessment

3.2.1 Discounting

Regardless of the method chosen, one additional step is likely to be necessary in the valuation process, due to the fact that the impacts deriving from the various activities of any fuel cycle stage may take place at a different time from the activity itself. Costs that take place in the future must be translated, through the discounting process, into present values. Discounting is particularly important when the assessment of a fuel cycle or a technology is done for investment purposes. Discounting is, however, controversial. Part of the literature (particularly that which starts from a strong environmental perspective) argues against discounting on ethical grounds, because it places more weight on the welfare of present generations than on the welfare of future ones. Large environmental damages taking place 50 or more years from now as a result of present choice will appear negligible. The bias against future generations becomes more pronounced with higher discount rates.

Even once the decision has been taken in favour of discounting, the choice of an appropriate discount rate is not trivial. It is known that discount/interest rates practised by lending institutions can change substantially depending on the risk profile presented by the borrower or by the enterprise. Furthermore, market rates (faced by individuals and private investors) differ from public discount rates (which are usually lower). The latter would appear to be preferable in assessing environmental goods. But even for governments, discount rates, inasmuch as they represent the opportunity cost of capital, can vary widely depending on whether their country is a very poor one with increasing population or a wealthy one with a stable population. In the current debate, reasonable discount rates for environmental valuation are believed to be in the range from 0% to 10%.

3.2.2 Transferability of estimates on environmental values

Life-cycle assessment should, where possible, be site specific, and consider both the local conditions where an energy activity takes place and where the environmental impacts are produced (even if these are far apart). Ideally site specific analyses should also be undertaken for the monetary valuation of environmental benefits and costs. But it is obvious that doing so *ex-novo* for each assessment would be enormously expensive, if not entirely infeasible. It is therefore becoming more common to apply monetary values from studies made elsewhere to specific cases and locations.

This transfer can lead to problems. The risk of using response functions from the literature in contexts different from the ones where the study was made (noted above) is not the only cause for concern. Problems also arise when monetary values elicited with contingent valuation methods are applied to a very different socio-economic context from the original one – leading to significant distortions in the analysis. Population characteristics such as income, age and lifestyles heavily influence consumer preferences and are critical elements to be taken into account when transferring value estimates from one situation to another.

3.2.3 Valuation of health impacts

Economic valuation of health impacts, and particularly increased mortality impacts, is an especially thorny issue, partly because of the ethical implications of putting a monetary value on human life. It becomes even more of an issue in the context of transferability of values.

For occupational accidents (both fatal and non-fatal) in specific activities pertaining to various stages of the fuel/technology cycle, databases are available on the compensation paid by companies, based on the characteristics of the workforce and the typical occupational hazards to which they are subject. For non-occupational health impacts, a frequently used approach to value mortality impacts and determine the value of a statistical life (VSL), is WTP for a reduction in risk of death or WTA for an increase in risk. Through this methodology a value of

3.1 M€ for a "statistical life" was determined based on accidental deaths (from car accidents). Early studies used to compute the cost of mortality by multiplying this value times the number of additional deaths caused by a given increase in pollution. However, many health impacts caused by exposure to pollutants tend to reduce the life expectancy of the weaker portion of the population (the aged and the ill) rather than kill adults in the 20-40 years age bracket. The ExternE methodology has recently opted in favour of the concept of "years of life lost" (YOLL), i.e. the loss of life expectancy due to risk of death and other health risks, and values the cost of mortality in proportion to the years of life lost (Rabl and Spadaro, 2001).

Actual estimates of WTP can be made in three alternative ways:

1) Using the contingent valuation (CV) method where people are asked to indicate how much they would be willing to pay to reduce by an hypothetical amount their risk of death from certain familiar activities or how much they would be willing to accept in compensation for an increased risk in the same activity.

2) Studying wage differentials (*ceteris-paribus*) for work activities having different health risks or occupational hazards.

3) Studying voluntary expenditures on items that reduce death risks from certain activities (e.g. airbags in cars).

All of the above methods are subject to criticism. CV methods are affected by differences in personal income levels and actual capability to pay and by age differences. Actual risks may widely differ from perceived risks in some activities. The wage-risk method may reflect more conditions prevailing in the labour market rather than actual preferences on risk levels, and therefore tend to be lower than the CVM. Furthermore voluntary risks are valued by the individual much differently from involuntary ones, with the value of the latter being larger for probabilities of death in the same range. Possible sources of bias are therefore plentiful and should be kept in mind when transferring VSL estimates from one country to another or from a probability range to another.

Valuation of morbidity impacts is usually done adding up at least three main value components:

- The cost of the illness in terms of hospitalisation and treatment expenses, or mitigation expenses (at market values).

- The value of time lost or foregone earnings (valued at the after tax wage rate).

- The loss of utility due to pain and suffering, usually determined through CVM methods.

These may or may not cover all the costs (for instance, the value of pain caused to family or people attending the sick person). However, at least in developed countries, one can count on a sufficiently extensive literature on estimates of the value of illness cases.

3.2.4 Valuation of global warming impacts

The impacts of global warming and related external costs are particularly complicated to measure. Few estimates exists, and these are based on models and scenarios which cover different time-horizons, use often very different assumptions on population, income and greenhouse gas emissions growth trends, and try to simulate changes in global climate patterns from limited understanding of the underlying drivers. It is from such scenarios that changes in global and regional climate patterns are translated into impacts on agricultural productivity, frequency of extreme weather effects, epidemiological trends, human mortality, etc. The monetised costs of such impacts are then calculated, discounted and aggregated.

Problematic circularities exist in this process. The timing and intensity of some climatic impacts depends on the rapidity at which certain concentrations of GHG are reached in the atmosphere. Damages from climate change are a function of the capability of a given human population to respond with defensive or mitigation measures: these are in turn a function of income level. On the other hand, as the value of climate change is usually assessed based on willingness to pay principles (also related to income level), the value of the same damage is higher in richer countries. Issues of intergenerational equity are magnified by use of discounting of these costs over time horizons of one century or more. As a result of different choice in the critical parameters, estimates vary often by several orders of magnitude.

Valuation of the global warming impact of a kg of a GHG, hence, requires extensive reviews of available literature on estimates of these impacts and computation of average values. This is the procedure that was followed by the first ExternE project (EC, 1995a).

In alternative, specific emissions scenarios can be chosen (for example those of IPCC) and a range of possible impacts computed from there using other models,[4] as was done in phase two of the ExternE project (EC, 1999b). In that project, marginal damages were calculated based on different discount rate options and for different GHG. The results of the second study indicated marginal values of carbon between 160 and 170 ECU/tC at 1% discount rate, and between 70 and 74 ECU/tC at 3% discount rate. Values for CO_2 were respectively in the range 44–46 ECU/tCO_2 and 19-20 ECU/tCO_2.

4. How can LCA be used in policy making?

Life cycle analysis can potentially provide important inputs both in policy making and in private decision making, and especially in the area of energy and environment policy.

The analysis of impact pathways allows the identification of the segments of the fuel cycle (or chain) which have the most harmful environmental impacts; the full LCA also requires the measurement of these impacts. This in turn can give an indication as to where priority should be placed – or where opportunities may be most attractive – for technology improvement, as well as for policy intervention. LCA and the valuation of external costs can also help identify the extent of the measures needed to reduce the harmful activity.

However, these methodologies and their results must be considered as only one tool among many. Economic theory has some prescriptive indications about the optimal level of the polluting activity: it corresponds to the level where marginal benefits from the activity equals marginal cost (private + external cost). In perfectly competitive markets, with perfect information and no transaction costs, this optimum point would be easy to identify. In such a situation, Pareto-optimality could be restored by simply imposing a Pigouvian tax (per unit of pollutant emitted) equal to the marginal social damage caused by the pollutant (Baumol and Oates, 1988).

In an imperfect world, the quantification of the externalities is a big problem in itself, the optimal tax level cannot be computed and the policy instruments to correct the problem are bound to be "second-best" solutions. This does not mean that efforts to quantify external costs should not be made: LCA and externality valuation attempt to do just that. Neither it means the information thus collected should not be used in policymaking. But designing cost-effective policies to deal with externalities in a situation of imperfect information is big

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4. Two climate change models, Climate Framework for Uncertainty, Negotiation and Distribution (FUND) and Open Framework, were used to gain a better understanding of the issues and obtain estimates of GHG marginal damages. IPCC scenarios IS92a and IS92d were used as a basis for these assessments.

issue in itself. Besides presenting administrative costs of varying magnitude, policies to control environmental damages, may themselves impose further costs to society because of bad design. Furthermore, obtaining more information in order to improve policy design has a cost too. For this reason, policymakers find themselves always in the situation of balancing this type of considerations and exploring trade-offs until satisfactory "second best" solutions are found.

There exists a vast economic literature on the relative efficiency of various policy approaches to address externality problems, and on the merits of price and market instruments (taxes, subsidies and pollution permit trading) versus "command and control" instruments (prohibitions, standards, directives, etc.). Following this debate, governments in the last decade have changed their environmental policy portfolio towards more market-oriented and decentralised approaches. The debate, however, is not over, as in many situations insufficient or asymmetric information, high transaction or policy implementation costs, or the social hazard of some activities leave little practical room for policy choice. Although inefficient in many situations and difficult to enforce, mandatory standards are sometimes the only viable solution. Furthermore, policy making often has to serve and reconcile multiple objectives, not only health and environmental protection.

LCA may be particularly helpful in areas where new technology or developments are underway. In these areas, it can help focus R&D programs in those segments of the cycle where the technology needs to be improved and made more environmentally benign. Carrying out the analysis prior to project development can eliminate costs and prevent damages. A similar use may be made of LCA in more mature technologies (e.g. nuclear power or fossil fuels); LCA can point to a variety of opportunities for improving the sustainability of the full fuel cycle operation, and allow project managers to develop least cost solutions to manage problems.

LCA can be useful in an investment decision framework, as it offers a more complete comparison of different energy technology/fuel alternatives that provide the same energy service, in terms of their short and long-term environmental impacts. Once these elements are correctly included in the evaluation of costs and benefits for each technology and accounted for together with "internal costs", the investor is provided with a clearer view of the options available and can make a better choice. Such value is not limited to an investor. LCA and the dissemination of its results among the general public in a democratic society can help make informed choices among technological alternatives; provide input to the political debate on the sustainability of energy systems; build consensus around specific projects or help reject those that pose severe environmental threats.

The following section of this paper provides a few practical examples to illustrate these issues, using information on external costs produced through the application of LCA and externality valuation methods. The first deals with the transport sector, the second with power generation. Both examples not only provide further insight into the LCA methodology, but also illustrate how current policy approaches might be changed were such LCA results to be factored into investment and decision making.

4.1 Case 1 – Transport

4.1.1 Results from LCA analysis

One of the most thorough evaluations of the LCA approach was that developed in the ExternE project. In its second phase, the ExternE project applied the "impact pathway" to the transport sector (Bickel *et al*., 1997). LCA was undertaken in a number of case studies in France, Germany, Greece, Italy, the Netherlands and the United Kingdom. The results provide an overview of the variation within the European Union, among several site-specific results.

The study covers passengers and freight transport by road, rail and waterways, but not air transport. Within each type of transport, different technologies are examined:

- For passenger transport: petrol cars with and without three-way catalysts (TWC), diesel cars, LPG cars, diesel coaches and buses, various types of small buses (diesel, LPG, bio-methanol, bio-ethanol, bio-diesel, electric) electric buses, trams, metros, and trains (electric: local, inter-city, high speed).

- For freight transport: light goods vehicles (petrol, with and without TWC, and diesel), heavy goods vehicles (diesel), and trains (electric and diesel).

For passengers transport, travels in different urban agglomerations (Paris, Athens, Milan), urban areas (Stuttgart, Amsterdam, Barnsley) and extra-urban areas (motorway drive or inter-city train, Stuttgart-Mannheim) are considered, and the environmental impacts of each transport option are quantified and monetised.

The analysis starts from quantification of air emissions for each pollutant by multiplying emission factors by transport distance covered. Emission factors in turn depend on the technology used (vehicle type), traffic conditions, and driving cycles. Total emissions are the sum of emissions in each road segment. Then marginal increases in pollutant concentration are estimated both locally (in a band 35 km wide on each side of the road) and regionally (the entire European continent, except former Soviet Union) through the use of dispersion models. Physical impacts on health (mortality and morbidity), crops and materials are then calculated through exposure-response functions. Finally, economic valuation of the estimated impacts is performed.

While the authors recognise that substantial uncertainties arise in the estimate of environmental cost of air pollution, they suggest that the largest uncertainties lie in the last stage and are particularly related to political and ethical issues. They also provide uncertainty labels for each impact value, based on geometric standard deviations (labels A, B, C, in decreasing order of confidence in the estimate).

The evaluation covers various types of impacts. Health impacts include, besides mortality, acute and chronic health effects from pollutants emitted in the transportation activity: particulate, SO_2, NO_x, CO, ozone, as well as aldehydes, ethene and ethylene oxide, MTBE (Methyl tertiary-butyl ether), volatile organic compounds, benzene, butadiene, polycyclic aromatic hydrocarbons, diesel exhaust particulate and lead. Impacts on the natural (soils and crops) and man-made environment (materials, buildings), as well as on global warming were also taken into account.[5] Furthermore, in two cases the impacts of the up- and down-stream processes, based on the detailed life-cycle analyses (fuel production; production maintenance and disposal of vehicles and infrastructure) available from ExternE or in the existing literature, were added.

The results for 1995 indicate that health impacts dominate the damages quantified in that study, in particular mortality due to primary and secondary particulate, while carcinogens (highly toxic but usually emitted in small quantities) have a much smaller size. Furthermore, researchers found that the magnitude of the health impacts, particularly for diesel vehicles, depends on the population density around a road. This is mostly due to the importance of primary particles in total damage.

Estimated damages for a diesel car range from 560 mECU/vkm in Paris, to 65-107 mECU/vkm in urban areas and 30-38 mECU/vkm in extra-urban areas.[6] Table 2 gives some additional information on costs.

It is possible to compare these costs against prices experienced by consumers in 1995 (year the evaluation was performed). For comparison we take

5. Global warming impacts in this study were determined on the basis of IPCC results, but using two climate damage models. Resulting values, when catastrophic or higher order damages are neglected, are in the range of those reviewed by the IPCC, i.e. \$5-\$125/tC. When higher order damages are included the range of values is greatly enlarged. Hence the IPCC spread of values was used in the ExternE project.
6. These are the damages stemming from vehicle use only.

as reference the 1995 prices (including fuel tax and VAT) for diesel oil and unleaded gasoline, and average fuel consumption per 100 km, to compute direct fuel costs in Table 3.

It shows that quantified damage due to airborne pollutants and greenhouse gases are at least as large as the direct fuel costs for a diesel vehicle, and may be up to 13 times as large in an urban agglomeration like Paris. On the other hand for gasoline in a three-way catalyst car, the direct fuel costs are higher than pollution damages; in fact, damages are more than covered by the fuel tax, which in the countries considered represents over 70% of the direct fuel cost.

Table 2. **Damage estimates for diesel and TWC petrol cars in different locations given as "best estimate" in mECU/vkm**

Impacts	**Agglomerations**	**Urban areas**			**Extra urban areas** (motorway drive)	
	Paris FR	Stuttgart DE	Amsterdam NL	Barnsley UK	Stuttgart-Mannheim DE	Tiel drive NL
Diesel car						
Particles (PM2.5)	534.090	50.430	78.600	97.400	18.770	29.500
Other primary pollutants [a)]	4.970	1.663	1.283	2.055	0.781	0.540
Secondary pollutants [b)]	20.060	10.920	4.900	4.380	8.700	6.100
Global warming	2.970	2.280	2.700	3.450	1.990	2.300
Up- and down-stream [c)]		7.100			7.100	
Total	562.090	72.393	87.483	107.285	37.341	38.440
TWC petrol car						
Particles (PM2.5)	53.410	3.730	1.960	4.170	1.100	0.740
Other primary pollutants	1.440	0.320	0.180	1.115	0.094	0.131
Secondary pollutants	18.040	5.210	2.320	4.440	6.600	3.640
Global warming	3.580	2.980	3.200	3.480	2.380	2.490
Up- and down-stream		9.000			9.000	
Total	76.470	21.240	7.660	13.205	19.174	7.001

Source: Bickel P., S. Schmid, W. Krewitt, and R, Fredrich (eds.): External Costs of Transport in ExternE. Publishable report – Contract JOS3-CT95-0004, 1997.

a) SO_2, CO, carcinogens.
b) Sulphates, nitrates, ozone.
c) Up- and down-stream costs were only computed for Germany: this estimate could be transferred to the other country cases, with due caution.

The ExternE report also provides information on other technologies and modes and shows that vehicles with internal combustion engines have higher environmental impacts than electric vehicles. Damages per passenger kilometre for trains range from 1.1 to 6.6 mECU/pkm, with the higher value corresponding to a diesel train. To these values one must add up- and down-stream costs ranging from 2 to 5.5 mECU/pkm, mostly occurring in infrastructure production, maintenance and disposal. The advantage of electric train over road transport for goods is even larger than for passenger transport: 0.8-8.8 mECU/tkm for trains against 33-400 mECU/tkm for gasoline or diesel vans, with heavy goods road vehicles posting costs in the middle of the range.

Table 3. **Average fuel consumption, fuel price and direct fuel costs for France, Germany, the Netherlands and the United Kingdom, in 1995**

	Fuel	France	Germany	Netherlands	UK
Average fuel consumption, L/100km[a]	Gasoline	8.5	9.3	8.3	9.2
	Diesel	6.7	7.6	6.9	7.5
After tax fuel price ECU/litre[b]	Gasoline	0.8641	0.8142	0.9156	0.6779
	Diesel	0.59	0.598	0.7374	0.655
Direct fuel cost mECU/vkm	Gasoline	73.4	75.3	76.3	62.4
	Diesel	39.8	45.6	50.9	48.8

a) *Source: OECD/IEA (2000): Energy Prices and Taxes, Paris.*

b) Gasoline prices are an arithmetic average of two different grade and octane n. unleaded gasoline. Data have been converted in 1995 ECU.

4.1.2 Existing policies on transport

Governments have adopted a wide variety of approaches to deal with the environmental and social externalities associated with transport. These range from imposing standards on automobile emissions to taxing vehicles, to taxing gasoline. It is interesting to compare the costs associated with the externalities (as described by the ExternE study above) to those experienced by the consumer. In the following analysis, we look specifically at taxation.

Historically, the main role of taxes on energy products (particularly oil products, gas and electricity) has been to raise revenues for governments. Demand for oil products is rather inelastic with respect to price, which makes them an excellent candidate for high taxes.

In OECD countries, these taxes represent, on average, slightly less than 6% of total tax revenues[7] (6.5% for the EU15)[8]. While definitely less productive than

7. OECD/ENV – Environmentally-Related Tax Database. http://www.oecd.org/env/policies/taxes/index.htm. Note that the OECD definition of environmentally-related taxes includes “any compulsory, unrequited payment to general government levied on tax bases deemed to be of particular environmental relevance.” This definition includes all energy taxes (which represent around 90%

income taxes as a source of fiscal revenues, energy taxes still significant. About 90% of total energy taxes come from motor fuel taxes. Countries dependent on imported energy have also used taxation as a way to decrease this dependency, reinforcing the fiscal justification.

In the last decade, a third motivation has appeared with increasing frequency: the environment. Environmental taxes reflect some of the environmental costs of using fossil fuels, discouraging the use of the resources taxed. Ideal application requires that these taxes fully reflect the magnitude of the environmental externalities associated with the use of the fuel in question.

The definition of environmental externalities used by the OECD implies that *any* tax on energy is environmental in nature. A stricter definition, in which taxes are counted only if levied specifically to help attain environmental goals, sharply limits the list of environmentally related energy taxes. Under this definition, environmental taxes on fossil fuels would include CO_2 or sulphur emission taxes. A number of other taxes levied on vehicle ownership based on engine type and horsepower or on use of highways may also be considered as related to environmental goals. Currently, however, only 1% of total energy taxes is environmentally based, in the sense that it is proportional to polluting emissions.

Ultimately, a tax of a given amount per litre or tonne of fuel will have the same impact on the consumption of that fuel regardless of the specific purpose for which it was created.[9] The impact of a tax depends on the absolute level of all taxes on a specific fuel, as well as the relative tax on that fuel compared to the tax on other fuels.

While the purpose of an environmental tax is to discriminate against fuels that damage the environment or public health, when it is superimposed to a tax structure designed to promote other objectives, the end result may not support the environmental goal.

In this sense, transport fuel taxation can present some striking anomalies. One is the preferential fiscal treatment accorded to automotive diesel fuel in certain countries. This in part has been supported by the fact that diesel engines are more energy-efficient than conventional gasoline-powered engines. However, in many cases (France, Spain, Italy and Belgium) this difference was

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of the total) as well as vehicle taxes, but the database also covers fees and charges for environmental services provided by the government.

8. Commission of the European Communities: Green Paper – Towards a European Strategy for the Security of Energy Supply. COM(2000)769. Brussels, 29 November 2000, p. 57.

9. Jean-Philippe Barde (OECD/ENV): *Taxes environnementales et réformes fiscales vertes dans les pays de l'OCDE*. Paper presented at the Seminar "Les Réformes Fiscales Vertes en Europe", Paris, 10-11 October 2000.

simply designed to discriminate in favour of commercial road transport, especially of goods (EC: Green paper-2000).[10] Already, diesel vehicles represented about a quarter of all new cars sold in 2000 in Western Europe. The preferential tax treatment may in fact have resulted in increased driving as a "rebound effect", thus erasing much of the potential diesel fuel savings.[11]

Figure 1 on gasoline and diesel oil price and taxes gives an idea of the variation in tax rates in European countries compared with countries in North America or the Pacific. Figure 2 shows tax differentials for diesel oil compared to gasoline.

The ExternE – Transport study can be used to give an indication on the discrepancy between the current structure and magnitude in transport fuel taxes in European Union countries and the estimated level of health and environmental externalities their use produce. To perform this simple calculation we have used average fuel efficiencies for gasoline and diesel passenger cars in EU countries in 1995. Then we compared the range of damages with the range of unleaded gasoline and diesel tax in EU countries in the third quarter of 2001. Results are shown in Table 4.

While present day gasoline taxes in the EU are high enough to cover the minimum estimated damages from its use, only in one case (that of the United Kingdom) do they cover the maximum value. The same is not true for diesel oil. Taxes on diesel fuel are not only lower than for gasoline (counter-intuitively with information on health costs), but the highest tax applied in Europe on this product is only sufficient to cover the lower end of the damage range.

10. Commission of the European Communities: Green Paper – Towards a European Strategy for the Security of Energy Supply. COM(2000)769. Brussels, 29 November 2000, p. 111.
11. This effect may represent in fact up to half of the fuel savings offered by diesel cars. See OECD/IEA (2001), *Saving Oil and Reducing CO_2 Emissions in Transport – Options and Strategies*, Paris (France).

Figure 1. **Prices and taxes for automotive diesel and gasoline, 3Q2001 (US dollars/litre)**

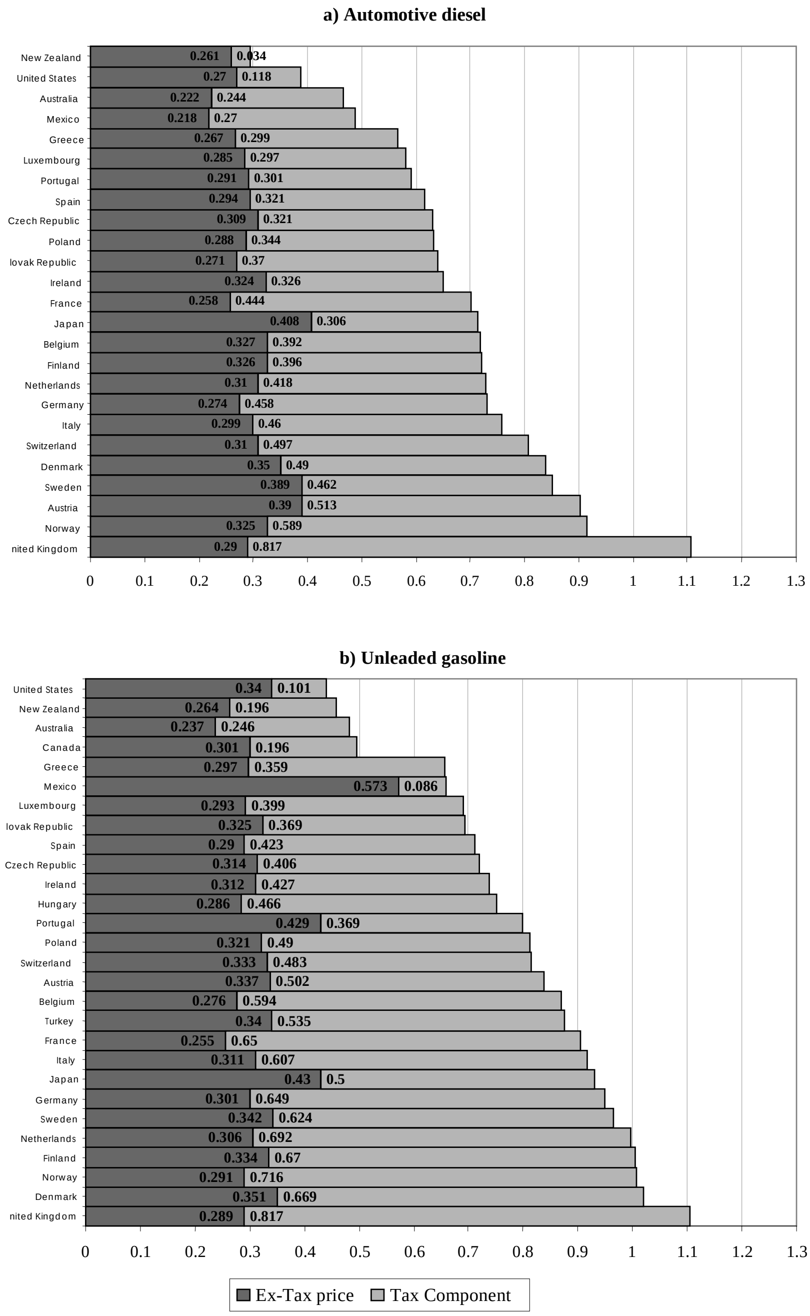
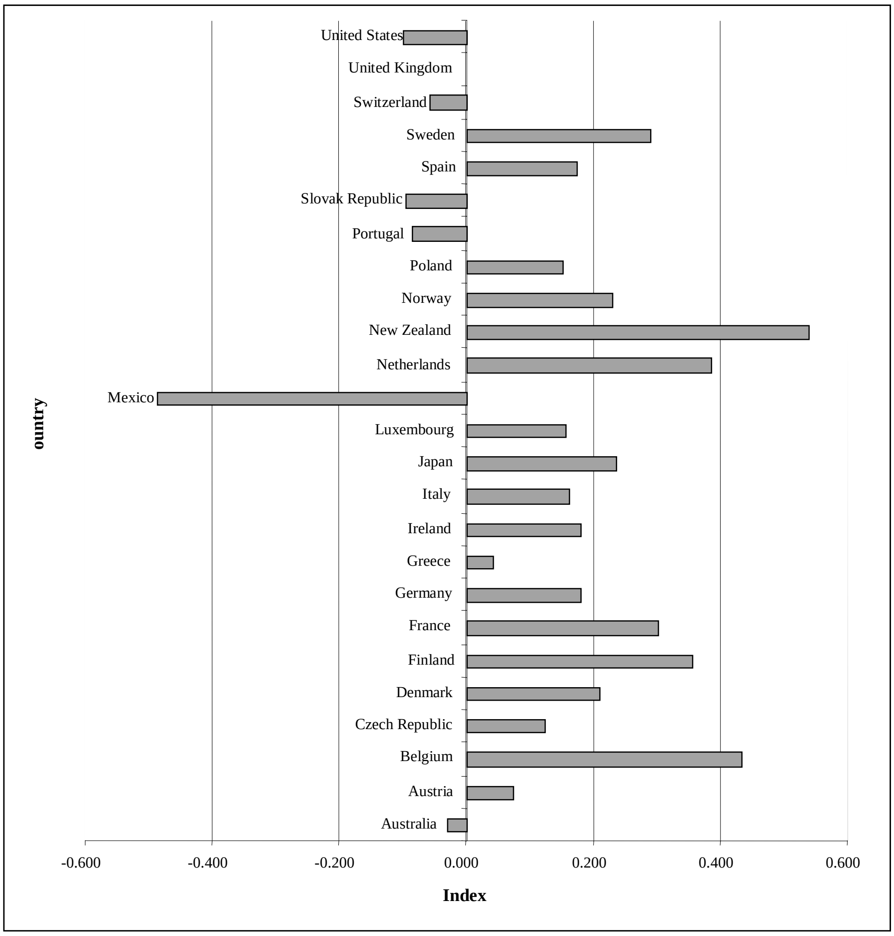


Figure 2. **Tax differential between unleaded gasoline and automotive diesel in IEA countries – Index values based on prices and taxes of 3Q2001***



Source: The index is computed based on data from OECD/IEA (2001): Energy Prices and Taxes- Third Quarter 2001. Paris.

*The tax differential index TDI has been computed according to the following formula:

$$TDI = \{[(ETP_g + T_g)/ETP_g]/ [(ETP_d + T_d)/ETP_d]\}-1$$

where ETP_g = ex-tax price of gasoline; ETP_d = ex-tax price of diesel fuel;

T_g = tax price on gasoline; T_d = tax price on diesel fuel.

Countries showing negative values (to the left of the vertical axis) apply higher overall (Excise + VAT) tax rates on diesel oil than on gasoline; countries showing positive values (to the right of the vertical axis) apply higher tax rates on gasoline than on diesel oil.

Table 4. **Comparison of estimated external costs and fuel taxes for unleaded gasoline and diesel fuel**

	Unleaded gasoline		Automotive diesel	
Average fuel efficiency in km/litre (1995)	11.331		13.937	
	Minimum	Maximum	Minimum	Maximum
External cost in Euro-cents/vkm[a)]	0.700	7.647	3.734	56.209
External cost in Euro-cents /litre	7.933	86.651	52.043	783.402
External cost in Euro/litre	0.079	0.867	0.520	7.834
Tax range in 3Q2001 in EU-16[b)]	0.411	0.935	0.340	0.935

a) Source: P. Bickel, S. Schmid, W. Krewitt, and R. Fredrich (eds.), External Costs of Transport in ExternE, Publishable report – Contract JOS3-CT95-0004, 1997.

b) Source: OECD/IEA (2001): Energy Prices and Taxes – Third Quarter 2001, Paris (France).

In fact, to cover the high end of this range, diesel taxes should be in the neighbourhood of 7.85 Euro/litre. The calculation is rather crude and does not take into account progress in fuel efficiency and environmental standards made by passenger cars since 1995. Best fuel efficiencies in Europe (excluding sport utility vehicles and mini-vans) are currently in the range 4.4-6.6 liters/100 km for diesel cars and 5.8-8.8 for gasoline cars.[12] In many cases product specifications on the sulphur content of the diesel fuel have been tightened, as have the standards on abatement of particulate emissions. Newest technologies allow for a significant abatement level for larger particles (PM_{10}). However, capturing the smaller particles ($PM_{2.5}$), which are the most dangerous ones for human health, is rather more difficult.

These results do not suggest, for example, that a threefold increase in diesel taxes (to remain in the middle of the external cost range) would be desirable. Not only would such an extreme taxation level be highly unlikely, but it would also be an inefficient way to deal with the environmental problems caused by diesel cars. However, the results of life cycle analysis and logic of the valuation of external costs do suggest that the fiscal advantage (in fact a true subsidy) granted to diesel oil should be eliminated, if not reversed, at least until the technological improvement in diesel engine and pollution-capture systems (especially finer

12. OECD/IEA (2001), Saving Oil and Reducing CO_2 Emissions in Transport – Options and Strategies, Paris (France).

particulate) improve to the point that the health impacts are more in line with those of gasoline cars.

Ideally, one should be taxing particulate emissions by each car, but as emissions depend (among other issues) on the type of driving and on car technology, and there is no end-of-pipe way to meter emissions, this is hardly feasible. Raising and enforcing emission standards for diesel cars or raising vehicle taxes would be another approach, as would be restricting use of diesel cars in large urban areas like Paris. In the meantime investment in R&D on cleaner diesel engines could be increased. But it is clear that something could be done to strike a better balance between the environment (and our own health) and our desire for mobility.

A different approach might be required to deal with pollution from diesel trucks for freight transport. Memories are still fresh of the oil price increase of 2000, which was followed by widespread protests and roadblocks from truck drivers, and prompted effective tax reductions on transport fuels. Increasing taxes on diesel fuel for this category of users may need a strong political resolve. And the strength of the truckers rests on the very simple fact that over 80% (in some countries over 95%) of all goods in Europe are moved by road transport. This is a reality that is impossible to change overnight, especially if no measure is taken to counteract existing trends.

4.2 Case 2: Electric power generation

4.2.1 Results from LCA analysis on cross fuel comparisons

A number of interesting applications of life cycle analysis to power generation can be found in the recent literature: we shall illustrate a few examples and then discuss how they can be used in public policy as well as in a private investment context.

A. Voss at the University of Stuttgart undertook an LCA for a cross-fuel assessment of electric power generation options in Germany (Voss, 2000).

In that study he used the following seven energy technology options to represent the current and near future portfolio in German electricity generation:

- Pulverised hard coal fired power plant, equipped with flue gas desulphurization (FGD), selective catalytic NO_x reduction and with a net capacity of 508 MW and 43% thermal efficiency.

- Lignite fired power plant, with FGD and selective NO_x reduction, net capacity 935 MW, and 40.1% efficiency.
- Gas combined-cycle power plant, with 778 MW net capacity and 57.6% thermal efficiency.
- Pressurised water nuclear reactor, 1 375 MW capacity.
- Photovoltaic home application, with amorphous Silicon cells, 5kW peak capacity.
- Wind-converter, with 1 MW capacity, 5.5 m/s average wind speed.
- Run-off hydropower plant, with 3.1 MW capacity.

The study, carried out from a sustainability perspective, took into account some of the key impact categories, including cumulative energy requirements, raw material requirements, emissions, health risks, external costs and power generation costs.

The results indicated that cumulative energy requirements, on a kWh of output basis, for plant construction and decommissioning plus production and supply of the main fuel used, were highest for the PV installation, followed by the nuclear plant and then by lignite, coal, wind, gas and finally hydro. A similar situation was found for materials requirement in plant construction and operation and in fuel supply: PV had the highest requirements per kWh in such minerals and metals as bauxite, iron and copper, and was second only to lignite for limestone requirements. The relatively small energy density of solar radiation explains this comparatively high material demand. By contrast, on a kWh of output basis, nuclear or hydro are relatively frugal in their use of materials with respect to all other technologies.

A slightly different story emerges when total life cycle emissions of CO_2, SO_2, NO_x and fine particles are considered. The situation is as shown by Table 5.

Emissions from coal and lignite are higher in this case, but the PV still does not compare favourably with respect to natural gas, for instance, while nuclear, wind and hydro show similar orders of magnitude in their overall emissions. The table above, however, although considering some of the pollutants with higher impacts on human health, does not include other emissions (e.g. in the case of nuclear, like ionising radiation).

Once all these were taken into account, both in up-stream and in downstream processes and in plant operation, the total health risks to the population were estimated, using the impact pathway approach, in terms of years of life lost (YOLL)[13] per unit of electric power produced (TWh). Results for the nuclear fuel chain include the expected value of risks from nuclear accidents beyond design, rated in this study as negligible.

Table 5. **Total life-cycle emissions for various power generation technologies**

	CO_2 equivalent (g/kWh)	**SO_2 (g/kWh)**	**NO_x (g/kWh)**	**Fine particles (g/kWh)**
Coal	951	351	696	64
Lignite	1 072	402	830	263
Nat. Gas CC	410	125	351	37
Nuclear	20	73	48	25
PV	216	433	321	107
Wind	41	68	49	18
Hydro	31	42	45	12

Source: A. Voss, (2000), Sustainable Energy Provision: a Comparative Assessment of the Various Electricity Supply Options. In Proceedings of SFEN Conference "What Energy for Tomorrow?", Hemicycle of the Council of Europe, Strasbourg, 27-29.11.2000. French Nuclear Society (SFEN), pp. 19-27.

Lignite and coal plants, with 73 and respectively 54 YOLL/TWh, top the risk scale, mostly due to health impacts from emissions in the plant operation stage. Third comes the PV plant, with 33 YOLL, entirely due to emissions occurring in the up- and down-stream processes, followed by the gas CC plant with about 25 YOLL/ TWh, three-fifths of which due to up- and down-stream processes. Nuclear compares favourably with about 10 YOLL/TWh, almost entirely resulting from emissions in the up- and down-stream processes. The best performers, according to this criterion, are wind and hydro with respectively 4 and 3 YOLL/TWh due to plant operation.

Once these health impacts and other environmental damages due to air pollution are taken into account and assigned a monetary value, an estimate is obtained of the external costs not accounted for in the electricity price. Table 6 shows the breakdown of these costs by power generation technology.

For each technology the author of the study then adds these costs to those for power generation (or private costs), sustained to pay for construction

13. Years of life lost represent the loss of life expectancy due to risks of death and other health risks.

materials, plant and equipment, fuels, and operations and management costs. The result gives an estimate of the overall social costs of electricity produced with different technologies and some information on the weight of external costs in the total (generation + external) cost of each technology. Knowledge of the magnitude of these two elements is extremely valuable when choosing among generation options.

Table 6. **Quantifiable external costs of energy systems (in Euro-cent/kWh)**

Impact	Coal	Lignite	Gas CC	Nuclear	PV	Wind	Hydro
Health effects	0.8	1.0	0.3	0.2	0.4	0.05	0.04
Crop losses	-0.03	-0.03	-0.01	0.0008	-0.003	0.0005	0.0004
Material damage	0.02	0.02	0.007	0.002	0.01	0.001	0.0007
Noise nuisance						0.006	
Acidification/ Eutrophication[a)]	0.2	0.8	0.04	0	0.04	0	0
Global warming[b)]	1.6	2	0.8	0.03	0.3	0.03	0.03
Sub-total	**2.6**	**3.8**	**1.1**	**0.2**	**0.8**	**0.09**	**0.07**

Source: A. Voss, (2000), Sustainable Energy Provision: a Comparative Assessment of the Various Electricity Supply Options. In Proceedings of SFEN Conference "What Energy for Tomorrow?", Hemicycle of the Council of Europe, Strasbourg, 27-29.11.2000. French Nuclear Society (SFEN), pp. 19-27.

a) Valuation based on marginal abatement costs required to achieve the EU "50% – Gap Closure" target to reduce acidification in Europe.
b) Valuation based on marginal CO_2-abatement costs required to reduce CO_2 emissions in Germany by 25% in 2010 (19 Euro/t CO_2).

For the specific case examined, the sum of maximum[14] generation costs and external costs equals:

- 60 €-cents/kWh for solar photo-voltaic power.
- 10 €-cents/kWh for hydro-power.
- 7.5 €-cents/kWh for lignite.
- 7 €-cents/kWh for wind-power.
- 6 €-cents/kWh for coal.
- 5 €-cents/kWh for gas CC.
- 3 €-cents/kWh for nuclear power.

14. A. Voss indicates minimum and maximum values for each technology generation costs.

It is important to underline that external costs represent a significant portion of the total cost for fossil fuel power generation (from about 1/3 for gas CC to ½ for lignite), and a very low or trivial portion of the costs for hydro, wind and nuclear power. On the other hand, while external costs are non-trivial in absolute terms for PV, they almost disappear compared to the high generation cost of this new technology.

A paper by Rabl and Spadaro (Rabl and Spadaro, 2001), reports some figures on the external costs of various fossil, nuclear and renewable power generation cycles, computed in the framework of the ExternE project and with an impact pathways methodology. The results are reported in the Figure 3. In both analyses, fossil fuel based generation presents higher external costs than generation from renewables. This is mainly due to the costs of global warming effects and of health impacts.

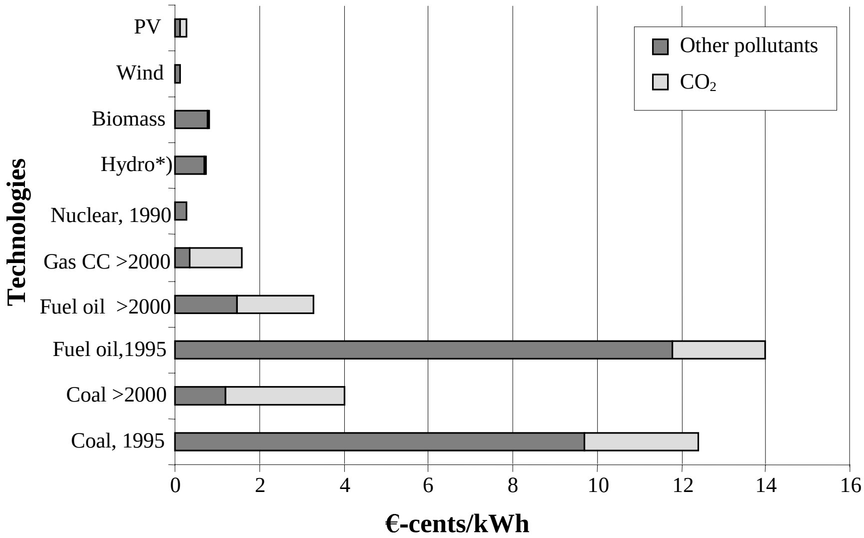
Rabl and Spadaro warn the reader about the variability and uncertainties connected to these estimates, which represent average values. For fossil fuels and nuclear plants, estimates correspond to typical existing plants in France and Europe or to new planned ones responding to EU environmental standards. The nuclear fuel cycle also features small external costs, although it is not so clear whether all the significant impacts of this fuel cycle have been duly quantified.[15]

For renewables, estimates represent averages for the case studies analysed in Europe within the ExternE project. The range of variability, however, is large enough that estimates could vary by a factor of four. And in fact the differences with respect to the situation depicted in Table 6 by Voss are rather significant. Figures for fossil-fuelled plants in the German example seem to correspond to those for the newest planned installations considered by Rabl and Spadaro.

Further estimates on the external costs from newer power generation technologies have been produced by the ExternE project (EC, 1999c). Fuel cells, for example, present non-trivial values for external costs, especially when they are fuelled with natural gas.

15. See the Externe website: http://externe.jrc.es/All-EU+Conclusions.htm.

Figure 3. **Cost of environmental damages for various power generation cycles. €-cents/kWh**



*) Due to the extreme variability of costs depending on the site, the value reported corresponds to the highest value of the estimate, ranging between 0.04 and 0.74 €-cents/kWh.

Source: A. Rabl, J. Spadaro (2001): "Les coûts externes de l'électricité", Revue de l'Énergie, No. 525, Mars-Avril, pp. 151-163.

Results from the ExternE study on two CHP plants with a phosphoric acid (PAFC) and respectively a molten carbonate fuel cell (MCFC), indicate the following costs:

- For the PAFC system: 0.07-0.13 €-cents/MJ heat produced and 1.6-3 €-cents/kWh electricity produced.
- For the MCFC system: 0.05-0.1 €-cents/MJ heat produced and 1.1-2.3 €-cents/kWh electricity produced.[16]

This is due to the fact that, while in operation SO_2, NO_x and particulate emissions from fuel cells are very small or zero, the production of the plant itself and the gas fuel cycle contribute considerably to SO_2, NO_x and particulate emissions and thus to increased mortality. The production of the platinum and

16. See p. 174, European Commission, DGXII-Science, Research and Development (1999), *ExternE, Externalities of Energy. Vol. 9:Fuel Cycles for Emerging and End-Use Technologies, Transport and Waste,* EUR 18887. Office for Official Publications of the European Communities, L-2985 Luxembourg.

nickel used in the fuel cell stack are very energy intensive. Nontrivial also is the global warming damage caused by this technology.

4.3 How LCA results can be used

4.3.1 Subsidies for renewables?

A press release of the European Commission – DG Research issued on 20 July 2001, reports on some recent results of the ExternE project concerning the external costs of electricity production in Europe. Estimates have been performed according to the impact pathway methodology. This study suggests that aggregated estimated external costs from electricity generation, according to the report, would amount to 1-2% of the European Union's Gross Domestic Product (GDP),[17] not including the cost of global warming.

Table 7 summarises the findings for nine fuel/technology groups for power generation in 15 EU countries. As shown, wind, hydro and nuclear power present the lowest external costs, below 1 €-cents/kWh, followed by biomass. Fossil fuels present different ranges of costs: gas is between 1 and 2 €-cents/kWh, while oil, peat, coal and lignite are more in the range 3-10 €-cents/kWh. These figures should be compared with an average power generation cost around 4 €-cents/kWh.

To take into account the costs of environmental and health damage, the European Union report mentions two possible types of policy:

1) Factor the external cost of damaging fuels and technologies in their electricity prices, via a tax.
2) Encourage or subsidise cleaner technologies that cause much lower environmental costs.

Considering the enormous difficulty of introducing taxation on an EU level, the European Commission is opting for the second solution. Community guidelines on state aid for environmental protection, published by the EC in February of 2001 in fact foresee that "Member States may grant operating aid to new plants producing renewable energy that will be calculated on the basis of the external costs avoided".[18] Based on that principle, the maximum amount of aid granted to renewable energy technologies must not exceed 5 €-cents/kWh.

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17. Preliminary work within ExternE has shown that the aggregated external costs of road transport would amount to another 1-2% of the combined GDP of the EU.
18. "Community guidelines on State aid for environmental protection" (2001/C 37/03). E.3.3.3, Option 3, §63, Official Journal of the European Communities of 3.2.2001.

Table 7. **External costs for electricity production in the EU in Euro-cent/kWh (ranges)***

Country	Coal & lignite	Peat	Oil	Gas	Nuclear	Biomass	Hydro	PV	Wind
Austria				1.1-2.6		2.4-2.5	0.1		
Belgium	3.7-15.0			1.1-2.2	0.4				
Germany	3.0-5.5		5.1-7.8	1.2-2.3	0.4-0.7	2.8-2.9		0.1-0.3	0.05
Denmark	3.5-6.5			1.5-3.0		1.2-1.4			0.1-0.2
Spain	4.8-7.7			1.1-2.2		2.9-5.2			0.2
Finland	2.0-4.4	2.3-5.1				0.8-1.1			
France	6.9-9.9		8.4-10.9	2.4-3.5	0.3	0.6-0.7	0.6		
Greece	4.6-8.4		2.6-4.8	0.7-1.3		0.1-0.8	0.5		0.25
Ireland	5.9-8.4	3.3-3.8							
Italy			3.4-5.6	1.5-2.7			0.3		
Netherlands	2.8-4.2			0.5-1.9	0.7	0.4-0.5			
Norway				0.8-1.9		0.2	0.2		0-0.25
Portugal	4.2-6.7			0.8-2.1		1.4-1.8	0.03		
Sweden	1.8-4.2					0.3	0.04-0.7		
United Kingdom	4.2-6.7		2.9-4.7	1.1-2.2	0.25	0.5-0.6			0.15

* Sub-total of quantifiable externalities.

Source: European Commission, DGXII-Science, Research and Development (1999): ExternE, Externalities of Energy. Vol. 10: National Implementation. EUR 18528, Office for Official Publications of the European Communities, L-2985-Luxembourg, p. 6.

However, another critical element to be taken into account in generation portfolio choice, besides the variability of estimates concerning externalities, is the spread of generation cost estimates found for some technologies, particularly for those not yet well established in the market, as is the case for wind and solar PV. Table 8 illustrates some of the uncertainties in the ranges of costs that may confront the decision maker when considering investments using different technologies for centralised power generation. Costs are expressed in €-cents per kWh produced (including investment, fuel, O&M costs) and reflect average costs for new power plants projects in EU member countries and 2000 fuel prices.

Table 8. **Electricity generating costs for various alternatives – 1990Euro-cents/kWh**

	Coal & lignite	Oil	Gas	Nuclear	Biomass	PV	Wind
Minimum	3.2	4.9	2.6	3.4	3.4	51.2	6.7
Maximum	5.0	5.2	3.5	5.9	4.3	85.3	7.2

Source: European Commission – Green Paper: Towards a European Strategy for Energy Supply (2000).

Note: Production costs are for power generation at 7 000 hours overall availability (equipment availability for PV) and exclude excise taxes and subsidies.

As can be seen, even for some mature technologies (like nuclear, coal and gas) the range of variation in generation costs is rather large. Costs become a lot more variable when considering new technologies such as power from biomass, and solar PV.

The European Commission attitude is very much in line with what governments in the EU (and in IEA countries) are already doing. Information collected by the IEA on energy policies and measures implemented in the years 1999 and 2000 to deal with climate change, shows that most recently adopted fiscal measures are aimed at encouraging technology improvement and diffusion (OECD/IEA, 2001b). In 2000, more than two-thirds of the fiscal policies adopted are unambiguous price support measures in favour of renewable energy or of electric power produced through more environmentally benign technologies. The emphasis is on fostering the deployment of commercially available technologies and fuels that have very low greenhouse gas emissions but are not yet competitive with conventional fuel sources. Direct subsidies for construction of renewable energy power plants, minimum guaranteed price schemes for power generation from renewables or tax relief measures are frequent forms of support. Energy tax reductions or waivers, income-tax reductions, or tax credits are often granted to power generation from renewable energy sources.

Green pricing schemes and price support programmes are based on the consideration that in the near term, GHG reductions will be brought about through the enhanced use and improvement of already existing technologies rather than through technologies now at the laboratory stage. Support measures to speed deployment of existing technologies exploit the reduction of cost that follows increased output[19] – known as the "technology experience curve".

A correct application of price support mechanisms requires accounting – and assuming an economic value – for a number of factors. These include the cost of other technologies/fuel sources; climate and non-climate related external costs (local air or water pollution, damages to human health as well as to the natural environment, and to the materials, noise, siting issues, etc.); and the long-term expected changes in energy activities that could emerge as a result of such programs. Such factors are likely to differ from country to country, but assessments like the ones performed under the ExternE project can help set benchmarks and avoid from the beginning the most obvious distortions.

The information collected by the IEA indicates that new taxes or increases in existing taxes are the least frequently applied measure. Very few taxes that have recently been adopted seek to discourage technologies and fuels with high

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19. Experience Curves for Energy Technology Policy, OECD/IEA Paris, 2000.

CO_2 or other pollutant emissions, thus seeking to "internalise" environmental costs.[20] Tax increases are sometimes applied to final consumption of electric power when it is fossil fuel-based.

The prevailing policy attitude, rewarding environment-friendly technologies rather than forcing polluting fuels and technologies to bear the full costs of environmental damages, encounters much less resistance from the public, and from the fossil fuel lobbies, but does have some potential drawbacks. Subsidies tend to stay in place beyond their "useful" lifetime, and they can discourage further, even more desirable technology development.

A more consistent approach, with respect to the results of the ExternE project would be the elimination of remaining coal subsidies, which still exist, with different political justifications, in a number of European (France, Spain and Germany) and IEA countries. The IEA does not consider there to be a realistic security of supply justification for such assistance to continue

4.3.2 A tool in investment decisions

Life cycle analysis and the valuation of the external costs stemming from a new energy project certainly adds important information to the process of assessing different investment options. The potential investor, be it a state monopoly or a privately owned company, would be well advised to take it into account in its strategic planning. As an actor operating in a given socio-economic context, it must pay due attention to public attitudes towards health and the environment, and follow a code of social responsibility, besides complying with existing law. In fact a company should be, to the extent possible, anticipating change in public opinion and legislation, and the best time to do so is when deciding on a new investment.

Ultimately, however, especially in liberalised and deregulated markets, the decisive element for a company willing to incorporate environmental concerns in its decision process is the institutional framework. If a company faces a regulatory environment (environmental taxes, environmental liability with or without mandatory insurance schemes, standards, or subsidies for environmentally benign technologies) which encourages it to pay attention to environmental externalities, it will do so to the extent possible. Otherwise, a green image with the public will loose to price competition in the market.

20. The "regulatory energy tax", implemented by the Dutch government for consumption of non-renewable energy by small consumers, the "environment tax" imposed by Denmark on fossil fuel and electricity consumption and the CO_2 taxes imposed in Norway and Sweden are a notable exception.

The need to factor LCA in the decision, clarifies some of the economic/environmental trade-offs although investors must still take risks and face often large uncertainties and constraints including technological uncertainty, infrastructure constraints, price variability or changing regulatory frameworks. Table 8 shown earlier illustrates this problem.

Certainly a utility interested in accommodating a specific growth in power demand does not usually face the full range of technology options or uncertainties shown in that table. It has a much more constrained decision framework when it comes to supply routes for fuels, plant siting options and land availability, infrastructure availability, environmental regulations, access to capital financing, etc., which may basically rule out many options. Information such as the one presented in Table 8 represents average costs, based on average assumptions on discount rates, plant costs, load or availability factors, construction lead times, fuel costs and so on. But when evaluating a specific set of options, the investor may, for example:

- Have to use very different discount rates for a gas combined cycle plant or for a nuclear one, or for different segments of the same cycle.
- Face a set of delivered prices for the fuel totally different from international prices.
- Need a specialised plant design (with significantly different investment costs) due to the specific characteristics of the fuel used.
- Consider different construction lead times due to uncertainties in the licensing process.
- Consider different sale price hypotheses.

The list could get much longer, but it is clear that all elements of the investment decision must be carefully evaluated and that sometimes purely subjective elements, may make the difference between a good investment and a poor one.

If the external cost estimates (e.g. those of the ExternE project (Table 7) are added to this already complex picture, generation costs for coal or oil-fuelled plants could increase by 3 to 15 €-cents/kWh. At least in part there would be a trade-off between investment costs and external costs. As a plant becomes more complex and uses more sophisticated pollution abatement devices, its cost goes up, while environmental costs produced during plant operation should be reduced.

With gas, the range of variability of external costs is smaller than that for coal and the externalities are also smaller, which could favour this fuel *vis à vis* coal in future investments. A similar consideration may be made for the nuclear technology, at present mature enough to have a smaller range of variability in its production costs, and showing relatively low external costs. Biomass-based generation[21] presents a fairly large range of variability, both for internal costs and for external ones. On the other hand wind power and PV have fairly low environmental impacts and external costs, but a large range of variation for investment and other internal costs.

But again, the ExternE estimates give a range of values, which come from specific case studies and which should be applied with much caution to different specific cases: all the caveats in the methodology section about the transferability of estimates become very important. An in depth assessment of the life cycle costs (including external costs) of the specific project may yield rather different figures with respect to the value ranges supplied by ExternE. This could depend on such elements as siting of the plant (population density, prevailing climatic conditions), origin of the fuel, infrastructure needs, process and fuels used in plant manufacturing, etc.

While life cycle analysis and an evaluation of external costs should be included in the decision making process for a new investment, it is unclear to what extent an energy company can do it and actually controls the life cycle of the product it is selling. More likely, a company will try to improve the segments of the cycle it can control, based on LCA results. This is another reason why using ready-made parameters and values may not be a good idea, especially for large projects. Uncertainties in available estimates of externalities could make decisions much less robust. Considering the range of variation of externality valuation, this would not impact just decisions on projects with very similar total (private + external) costs. Hence, an ad-hoc study, using more realistic parameters, may be safer. On the other hand, for small generation projects, it may be unnecessary or unjustified to perform a full-scale life cycle analysis. In that case it could make a lot of sense to use results as the ones available in the literature.

A different situation could be the one in which an investor is facing not just a generic social responsibility, but also specific policies that impose taxes or grant subsidies. These policies become a part of the regulatory framework and must be taken into account even if the costs and benefits they impose do not reflect the true costs and benefits of the project.

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21. See Table 1.3, p. 6 of the volume: European Commission, DGXII-Science, Research and Development (1999), *ExternE, Externalities of Energy. Vol. 10: National Implementation.* EUR 18528. Office for Official Publications of the European Communities, L-2985 Luxembourg.

5. Conclusions and issues for further analysis

In spite of the many limitations and uncertainties underlying life-cycle analysis and particularly the valuation of external costs, it allows the integration of environmental impacts and externality considerations in policy making. The methodology has a wide range of possible applications, covering all sectors of human activity. Concern for the impact of energy production and use on both local pollution and global climate makes it a particularly valuable tool in energy and environment policy.

The possibility LCA and externality valuation methods offer to quantify those impacts and attach monetary values to them, gives policy makers useful indicators on which to gauge policies to mitigate environmental and health damages, or monitor and correct past policies according to rationality and effectiveness criteria. However, in order to get the prices right, straightforward application to policy design of LCA and externality valuation results is not sufficient: policy design needs to become a lot smarter and forward looking and take into account different factors.

Most examples of LCA cases examined in this paper suggest that the largest share of total external damages assessed, for most energy technologies studied, are due to health impacts. Damages due to global climate change, to crop, wildlife or eco-systems loss, or even to the built environment, are apparently less important. While this result gives a sound motivation to pollution mitigation efforts, it seems also to indicate possible sources of bias. Models to predict climate change patterns and their future impacts on human life and activity still have important limitations and present enormous uncertainties, as do methodologies to value impacts that might take place decades into the future. Here, the process of discounting and its rate could be a critical issue. On the other hand, it has to be acknowledged that these evaluations necessarily reflect subjective, time-related and species-related values.

This paper has illustrated some examples in which the results of life-cycle analysis can be (or have been) used to introduce new policies or change their orientations. However, policymaking encompasses social objectives of which health, environmental protection and energy supply are only a subset. Within this subset, LCA can provide useful insights. On the other hand, when criteria such as income or employment growth, maximisation of tax revenues appear in the same objective function, as is often the case, LCA cannot be expected alone to provide the answer.

To date – at least in the examples described above, governments do not seem to be employing LCA as a deciding factor in policy making. It would, however, be interesting to know to what extent LCA results are informing government policies on sustainable development, and to what extent they can be useful in deriving “sustainability indicators”.

Similarly, from the point of view of investors, it is not entirely clear what impact the results of studies like ExternE are having on investment decisions and whether they represent useful quantitative tools to help orient their choices towards more environmentally benign technologies.

Future work might seek to evaluate these questions – both as a way of refining the LCA methodology, and as a way of evaluating how such methods might be more usefully applied.

This paper has highlighted some of the risks connected to unqualified use of some of the results of this methodology and to their transfer across different social, environmental and technological contexts. It has also noted the significant variability of some of the estimates of energy cycle externalities. What is less clear is whether a consensus is emerging, in the LCA community, on the relative ranking of technologies with respect to the magnitude of their external costs. This too thus represents a possible area for further work.

It has been noted that ample room exists for further refinement of the methodology, particularly concerning the swift incorporation of new scientific results from epidemiological studies, global climate research, and ecosystem analysis. Continuous progress in the technologies assessed should be also factored in, especially when using LCA results for policy design. Furthermore, inasmuch as consumer preferences remain a key element in the economic valuation of impacts, this methodology should closely reflect and incorporate change in value systems, both at the societal and at the individual level.

A number of other issues remain open, and would need to be tackled, both with a solid theoretical framework and with the help of some applied analysis. One such issue, which we have on purpose avoided, is that of energy supply risk, its evaluation and the evaluation of security of supply as a positive externality, with a value attached.

Notwithstanding these caveats – and the consequent need for future work – the LCA methodology does provide clear value-added in the decision-making process. It is therefore strongly recommended that such analyses continue to be developed, and where possible applied to policymaking in the energy sector.

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Printed in France.