THE COST OF POWER:

EXTERNALITIES IN SOUTH AFRICA'S ENERGY SECTOR

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Abstract

The long-awaited birth of political democracy in South Africa in 1994 has led to a fundamental re-assessment of policy in most sectors of society. Although the energy sector has witnessed a clear shift away from the self-sufficiency concerns of the apartheid era, to more universal goals of economic efficiency, social equity and environmental sustainability, there has, as yet, been very little analysis of problems at the energy-environment interface.

In this context, this thesis investigates environmental externalities arising in South Africa's energy sector. Two questions are posed: first and foremost, which environmental problems give rise to the most significant social costs? Secondarily, how helpful is an environmental economic analysis in this context? With respect to the first question, it is hypothesised that the external costs arising from two sectors are significant: the electricity generation sector, and the low-income, unelectrified household sector. Of these two, it is suggested that externalities in the latter are most serious.

After reviewing the literature on externalities and environmental valuation, the thesis undertakes an empirical investigation of external costs in both energy sub-sectors. A classification system is developed and used to select those externalities in each sector which are potentially serious and regarding which there is sufficient information for quantification purposes. After reviewing a larger number of impacts, data are collected from both published and unpublished sources for four environmental externalities in the electricity sector, and six in the household sector.

The results include quantitative estimates of total external costs, average external costs and marginal external costs for each external effect. These empirical results show that the external costs in the household sector are far in excess of those in the electricity sector, especially if damages attributed to greenhouse gas emissions are excluded.

With respect to the second question posed, a number of limitations inherent in an analysis of this kind are pointed out. It is suggested that an environmental economic approach can make an important contribution to sound policy in South Africa (and developing countries, more generally), but that limitations of the approach need to be taken explicitly into account.

An important conclusion is that strategies which are aimed at mitigating external costs in the household sector may bring about significant improvements, measured against both economic efficiency and social equity objectives. This convergence is especially pertinent in the context of South Africa's new development imperatives.

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Two people assisted in specific ways in this thesis: Tim James helped with the production of two graphics using CorelDraw (Figures 3.1 and 3.2), and Paul Hatfield assisted with the collection of data and literature related to the incidence of paraffin poisoning and burns - this was used in Chapter Five. I wish to acknowledge their assistance, and thank them for it.

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Finally, it is obvious to economists that few things in life come without a cost; unfortunately, a thesis is no exception. It is perhaps ironic - given the subject of this thesis - that some of the costs of my own efforts were borne also by others, most notably by Michelle. For her, the word `externality' took on a less-than-welcome, personal connotation at times during the completion of this thesis. So, as some measure of compensation, I wish to dedicate this thesis to her.

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List of abbreviations and acronyms

ADRAS	atmospheric deposition risk advisory system
AEC	Atomic Energy Corporation
ANC	African National Congress
APDC	Asian and Pacific Development Centre
ARI	acute respiratory infection
CBA	cost-benefit analysis
CMA	Cape Metropolitan Area
CNS	Council for Nuclear Safety
CO_2	carbon dioxide
COI	cost of illness
COSATU	Congress of South African Trade Unions
CSERGE	Centre for Social and Economic Research on the Global Environment
CSIR	Council for Scientific and Industrial Research
CSS	Central Statistical Services
CVM	contingent valuation method
DEAT	Department of Environmental Affairs and Tourism
DMEA	Department of Mineral and Energy Affairs
DR	dose-response
DSM	demand-side management
DWAF	Department of Water Affairs and Forestry
EDRC	Energy and Development Research Centre
EIA	environmental impact assessment
EMF	electro-magnetic field
EPA	Environmental Protection Agency (United States)
ESI	electricity supply industry
ESP	electro-static precipitator
EU	European Union
FCCC	Framework Convention on Climate Change
FEV_1	forced expiratory volume in one second
GDP	gross domestic product
GEF	Global Environment Facility
GHG	greenhouse gas
GME	Government Mining Engineer
GNP	gross national product
GP	general practitioner
GWh	Gigawatt-hours (one million watt-hours)
GWP	gross world product

HEAL	Human Exposure Assessment Location
ICU	intensive care unit
IDRC	International Development Research Centre
IEA	International Energy Association
IEP	integrated energy planning
IPCC	Intergovernmental Panel on Climate Change
kV	kilovolt
kWh	kilowatt-hour
LA	local authority
LPG	liquefied petroleum gas
LRI	lower respiratory infection
MAB	marginal abatement benefit
MAC	marginal abatement cost
MEC	marginal external cost
MEP	marketable emission permit
mill	one-thousandth of a Rand
MJ	megajoule (one million joules)
MNPB	marginal net private benefit
MPC	marginal private cost
MRC	Medical Research Council
MSC	marginal social cost
Mt	megatons (one billion kilograms)
MW	megawatt (one million watts)
NDP	net domestic product
NNP	net national product
NEC	National Energy Council
NER	National Electricity Regulator
NO_x	nitrogen oxide
Nucor	Nuclear Development Corporation
OECD	Organisation for Economic Co-operation and Development
ORNL	Oak Ridge National Laboratory
PM10	particulate matter with a diameter of less than 10 microns
RDP	Reconstruction and Development Programme
RFF	Resources for the Future
Rm	million Rand
SACP	South African Communist Party
SAIRR	South African Institute of Race Relations
SALDRU	Southern African Labour and Development Research Unit
SANCO	South African National Civics Organisation
SNA	system of national accounts

SO_2	sulphur dioxide
TBVC	Transkei, Bophuthatswana, Venda and Ciskei
TEA	total exposure assessment
TEV	total economic value
TSP	total suspended particulate
UCT	University of Cape Town
UNDP	United Nations Development Programme
URI	upper respiratory infection
VAPS	Vaal Triangle air pollution study
VAT	value added tax
WHO	World Health Organisation
WTA	willingness to accept (compensation)
WTP	willingness to pay
μg	microgram (one-thousandth of a gram)

Chapter One

Introduction

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1. The changing policy context in South Africa

The political transition from apartheid to democratic government in South Africa has been accompanied by wide-ranging reviews of policy, across sectors and at all tiers of government. The shift in policy orientation in the energy sector has been significant, yet considerable uncertainty remains in numerous areas. One such area is at the energy-environment interface, where there has been no systematic prioritisation of problems, nor of possible abatement options. This question - of the relative significance of environmental problems in the energy sector - lies at the heart of this thesis.

The energy sector during the apartheid era was characterised by a policy environment in which the state was highly interventionist, and in which the objectives of energy security or national self-sufficiency were paramount. This policy led to the investment of considerable public resources in large-scale energy supply infrastructure: beginning in the 1950s with the Sasol synthetic fuel processes which produced petroleum from coal, followed by an indigenous nuclear fuel (and weapon) production cycle which provided the feedstock for the Koeberg nuclear power station, and including most notoriously, the Mossgas oil-from-gas facility. Arguably, the over-investment during the 1980s in coal-fired electricity generation capacity could be attributed not only to over-estimates of the economic growth rate, as conventional wisdom would have it, but should be seen in the context of the dominant policy view of ensuring the country's energy security.

These massive investments which were made over a period of more than three decades, were undertaken directly, or under the influence of, the government of the day, with scant regard for the economics underpinning them. The economic consequences were (and indeed still are) significant: firstly, in the case of the synthetic fuel industry, a complex regulatory system was put in place to protect Sasol and, more recently, Mossgas, effectively guaranteeing them a market for their synfuels products. The net effect is that every litre of petrol, diesel or other petroleum-based product sold in South Africa is subject to a levy (3.4 cents in the case of petrol in 1992) which is granted to the synfuels industry (McGregor 1994: 10). Secondly, in the case of the nuclear industry, enormous subsidies have been granted to the industry since the 1970s, averaging about two-thirds of the annual budget of the Department of Mineral and Energy Affairs (DMEA). This public subsidy has delivered highly questionable results: nuclear fuel for Koeberg power station, which could always have been imported at a lower price, and, indeed, a number of nuclear bombs. Thirdly, the effect of over-investment in large coal-fired power stations was seen in higher electricity prices throughout the economy, particularly since the public utility financed these investments through commercial borrowings at high real interest rates.

Three points need to be made, therefore, regarding the energy policy environment during the apartheid era. Firstly, as suggested already, the driving force in the sector was the state's desire to reduce the country's dependence for its energy needs on an international community which was becoming increasingly intolerant of its apartheid policies. The economic performance of the resulting energy investments featured poorly in decision-making. Secondly, the state was highly-interventionist in these sectors, notwithstanding rhetoric to the contrary. Indeed, it is most improbable that the synfuels and nuclear industries would have developed without this state support. Thirdly, the direct involvement of the government in increasing national energy self-sufficiency stands juxtaposed against the neglect by the state and institutions under its control, of the energy needs of the majority of the population. In other words, equity concerns did not feature on its agenda. This is in contrast to many other developing countries at the same time where governments were actively assisting in the extension of national infrastructure (such as the electricity grid) to towns and rural areas. One measure of the opportunity cost of the mega-investments in energy security, was the low level of household access to preferred energy carriers such as electricity.

It was against this background that the post-1990 political transition took place. In some respects, the policy shift has been fundamental, yet in other ways, it remains constrained by inertia and vested interests. Energy self-sufficiency is no longer the dominant force it was; instead, other forces are manifesting themselves at the policy level. In terms of stated energy policy, at least, three goals have been presented as the new imperatives:

- social equity: in other words, improving the access of the poor to adequate and affordable energy services;
- economic efficiency: improving the efficiency and competitiveness of the economy by providing cheap and reliable energy services; and
- environmental sustainability: reducing the negative environmental impacts arising from energy use and, in the long term, moving away from the country's non-renewable resource base to ecologically sustainable options (DMEA 1995: 7).

These three goals - equity, efficiency and sustainability - are sometimes presented as the basic pillars of sustainable development. In the government's energy policy development process, which commenced with the writing of an Energy Policy Discussion Document (or

`Green Paper') in mid-1995, and which was followed by a White Paper in early-1996, these three goals are highlighted as the basic goals of the country's energy policy (ibid).

Whilst it is not difficult to find these goals agreeable, they are relatively general and hide the fact that there may be significant trade-offs to be made between them. It is therefore worth considering briefly how each of these objectives has been manifested in the South African political-economy.

Firstly, the aspirations of the poor have clearly moved up the policy agenda, encapsulated at the most basic level by the African National Congress' (and now the Government of National Unity's) Reconstruction and Development Programme, or RDP (ANC 1994). The RDP suggests that one of the government's key challenges is to meet the basic needs of the poor, and outlines a number of strategies to achieve this. It is interesting to note how the RDP very rapidly became the dominant social and economic statement of intent in many sectors of South African society subsequent to the April 1994 elections. Indeed, RDP-jargon became dominant in the political and economic discourse in the post-election period, to the extent that terms such as `redistribution' virtually disappeared from the public discourse.[1]

In the energy sector, the shift in policy objectives towards the basic needs of the poor was anticipated by Eskom,[2] and some local authorities, who launched an accelerated electrification programme in 1991. By the end of 1995, more than a million new connections had been made to low-income households in both urban townships and rural villages (Davis 1996). Likewise, in government, the Department of Mineral and Energy Affairs re-organised itself and established an `Energy for Development' directorate, whose aim was to develop policy for low-income households, an area which had received negligible attention prior to the 1990s.

Thus `equity' or redistributive objectives now occupy a rung high on the ladder of competing policy objectives. At the same time, the forces of economic competition and efficiency have become more dominant. It might have been expected that these forces would have injected themselves more quickly and forcefully into the policy arena, especially as South Africa re-engaged with the global economy. For instance, competitive forces might have been expected to expose and unravel the intricate protective measures developed to cushion (inter alia) the synfuels and nuclear industries from the harsh realities of a

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[1] This is not to say that the RDP is without its critics: during 1995, a number of critical views began to appear in the press and to be voiced in key forums (for example, Nattrass 1995).

[2] Eskom is the main public electricity utility in South Africa; it is described in more detail in Chapter Three.

competitive economy. This, however, is an area in which the policy shift has been less than radical or dramatic. Reasons for this include the strong vested interests which have been built up over many years by these protective measures, as well as the constraints inherent in a coalition government intent on maintaining goodwill and not threatening `national reconciliation'. The latter factor is particularly relevant in the energy sector, since its ministry was allocated after the April 1994 elections to the National Party (which governed South Africa from 1948 to 1994). Thus old alliances and relationships have not been directly challenged in this sector. To take just two examples, there were no major structural changes in the first two years of democratic government with respect to the subsidy to the nuclear industry; secondly, it took almost two years for government to take the decision to reduce tariff protection on Sasol's synfuels. The forces of economic efficiency and competitiveness, therefore, have been relatively slow to manifest themselves, although the scope for ignoring them is smaller than ever before.

In the area of environmental sustainability, much remains to be seen as to how environmental objectives will fare in relation to equity and efficiency goals. Environmental concerns feature relatively prominently in most policy pronouncements of the new government. The RDP base document gives prominence to environmental criteria (ANC 1994: 38), and the chapter on human rights in the draft constitution places up-front `the right to a clean and healthy environment'. As noted above, the Energy Policy Discussion Document places environmental quality and sustainability alongside the other fundamental goals of equity and efficiency. One of the document's cross-cutting themes was `environment, health and safety' which received an equivalent amount of space in the document to the gas supply sector, or the transport demand sector (DMEA 1995: 169-177). Thus, on paper at least, `environment' is an important priority.

In practice, however, the first two years of democracy have seen few occasions - in the energy sector, at least - on which environmental goals have been put to the test.[3] This is not to say, though, that energy-environment challenges do not exist or have not received much attention. Indeed, there are several issues which have received considerable attention from the public, policy-makers and analysts: for instance, the air pollution impacts of power stations on the one hand (for example, Clark 1991: 15), and township combustion of coal on

[3] Outside of the energy sector, however, there have been at least two major tests: first the long-standing question over the possible extension of Richards Bay Minerals' titanium mining area in St Lucia, and secondly, the possible construction of a steel mill at Saldanha Bay. Although neither of these issues have been fully resolved, indications are that both will involve a compromise agreement between environmental and development interests.

the other hand (Terblanche et al 1992a); the possible introduction of environmental control technology, such as scrubbers, in coal power stations (IDRC 1994: 56); and the role of the energy sector as a major source of greenhouse gas emissions (Scholes & van der Merwe 1995). As noted already, assessing the relative importance of some of these problems is one of the key themes of this thesis.

Before concluding this broad survey of the policy context at the energy-environment interface, it is worth describing briefly an initiative of the Department of Environmental Affairs and Tourism (DEAT) to promote the use of environmental economics in dealing with environmental problems.[4] The DEAT has established a programme, under the rather clumsy title of Environmental Resource Economics, in which it seeks to identify a range of market-based approaches to solve environmental problems (DEAT 1994). Considerable resources have been directed by the DEAT to the production of a number of base documents surveying the discipline, and outlining the possible application of its analytical and policy tools in various sectors of the South African economy. To-date, however, there have been few demonstrable results from this programme, and it remains a relatively academic exercise. Unless the DEAT is promoted from its historical status of `poor cousin' in government, it is difficult to imagine it having the strength or authority to introduce strategies (for example, tradable emissions permits) as done by environmental regulators in industrialised countries. What is more likely from a policy perspective, is that the DEAT will be able, at best, to support initiatives originating in other sectors or line departments. Thus, for instance, the introduction of pollution taxes on industrial emissions is most unlikely to be driven by the DEAT in the current institutional scenario; if anything, they would be introduced by the ministries overseeing energy and trade and industry. The analysis in this thesis is perhaps an example of such an exercise: one which is firmly located within the latter sectors, and which might represent a practical analysis of what the DEAT has proposed for some time but not yet accomplished.

[4] Significantly, the Ministry of Environmental Affairs and Tourism is also held by the Nationalist Party in terms of the negotiated coalition Government of National Unity. The allocation of ministries amongst political parties reflects many factors, not least of which the priority attached to that sector for purposes of serving each party's constituency. It is interesting in this context, that the energy and environment ministries were not amongst the ANC's highest priorities.

2. The research questions and hypotheses

The questions addressed in this thesis are posed against the background of an energy policy which has already undergone a fundamental shift away from energy self-sufficiency to more universal goals of efficiency, equity and sustainability, and in which, at the same time, there remains considerable uncertainty over the appropriate place of environmental goals *vis-à-vis* other national objectives. This thesis sets out to address two key questions, the first of which is of primary significance:

- Which are the most serious energy-environment problems in South Africa? Put differently, which are the problems with the most significant costs to society?
- How helpful is the discipline of environmental economics in addressing this question in the contexts of South Africa specifically and developing countries more generally?

The first of these questions has direct relevance for South Africa's energy and environmental policy insofar as resources available for intervention and abatement options can then be targeted to those problems currently imposing the highest costs on society. The question is posed from the perspective of a national policy-maker, who has limited resources at his or her disposal, and who wishes to effect the greatest possible improvement in welfare. The second question is more relevant from a methodological or analytical point of view, and involves a critical consideration of the responses to the first question. The principal focus in this thesis is on the first question; the process of addressing that question raises a number of methodological issues which, in turn, are relevant to the second question.

In considering which of the energy-environment problems are most important, it is clearly necessary to narrow the range of options to allow for meaningful analysis of each. Given the pervasive nature of energy in the economy, it is necessary to adopt a systematic approach to identifying and screening the range of energy-environment issues. One such approach is to distinguish between four scales of impacts, corresponding to household, national, regional and global impacts (see, for example, Pearson 1993: 100). Using this distinction, the following impacts can be identified:

1. *Household energy-environment impacts*: these problems include air pollution emissions from large numbers of low-income consumers who use coal and wood to meet their basic energy needs. Also at this scale, are health and safety hazards such as paraffin poisoning, burns and damages from accidental fires, and the social costs of increasing fuelwood scarcity.
2. *National energy-environment impacts*: at this scale, a key energy-related environmental problem concerns the impacts of electricity generation in terms of air pollution and other effects. Secondly, the effects of transportation - particularly the extensive use of private

modes of transport - are not insignificant in terms of air pollution emissions, opportunity costs of time spent commuting and resource depletion to support transportation activities.

3. *Regional energy-environment impacts*: at present, problems may exist in respect of trans-boundary transport of air pollution from South Africa to its neighbours, although the extent of this problem is not clear. In the medium-term, however, important policy questions will arise when decisions have to be taken regarding the next source of bulk energy supply for South Africa. In the new political environment prevailing in Southern Africa, the range of supply options is wider than ever: hydro-electric power from Mozambique, Zambia, Zimbabwe, Zaire and Angola; natural gas from the Pande fields in Mozambique and the Kudu fields in Namibia; and coal power in South Africa and Zimbabwe. Each of these options will have their own environmental consequences and it is most likely that these will play some role in future decisions.

4. *Global energy-environment impacts*: although there is considerable uncertainty surrounding the issue of global climate change, there is little doubt that it will remain a feature of the international political economy for the foreseeable future. The energy sector in South Africa and internationally is one of the major sources of greenhouse gas emissions, and it is widely believed that these could have a major impact on global climate patterns.

All of these issues have policy-relevance, although their urgency and scales differ. Of all the issues mentioned, this thesis focuses on two main environmental issues:

- the impacts of electricity generation - that is, a national scale impact, and
- the impacts of household energy consumption - a household scale impact.

It appears that significant social costs may exist at these two levels; moreover, the policy choices and trade-offs are particularly acute within and between these levels. Focusing on these two issues, however, does not mean that others are irrelevant or unimportant. For instance, significant environmental questions arise in the context of regional energy trade and investment, since the resource endowments of countries in the sub-continent span the range of supply options, each with their attendant environmental impacts: hydro, coal, gas and nuclear. This notwithstanding, the policy debate in this area lacks the immediacy of decisions to be made about household energy and national electricity policies. Likewise, South Africa's policy choices regarding global environmental impacts have received relatively little attention in their own right. Since global issues can generally be traced back to their causes at the local and national scales, they will be partially addressed in an analysis of the latter micro-scale problems. The focus of the remainder of the thesis is therefore on environmental problems arising in the electricity generation and household energy sectors.

Importantly, energy-environment issues at these two scales are not unrelated. In many respects, both are a result of the political economy of apartheid. On the one hand, apartheid government ensured that the interests of white households were served by supplying them with clean and safe energy sources. Likewise, the economic interests of mines, industries and the commercial sector were catered for with an abundant supply of affordable electricity. These interests were met through the efforts of the state to build up a large electricity supply industry, based as it now is, on the production of electricity by Eskom from coal and nuclear resources.[5] On the other hand, apartheid policies meant that the interests of black households were, at best, ignored and, at worst, actively opposed; one effect of this is that these households were, by and large, left to consume energy sources which generate serious environmental problems. Thus, the inheritance of South Africa's first democratic government includes the environmental burdens arising from these sectors: an electricity supply industry which was supported by its predecessor and which served the interests of the enfranchised minority, at the expense of the disenfranchised majority. The environmental costs in these two sectors are, in many respects, two sides of the same coin, a coin which was struck by the apartheid government.

At this stage, two hypotheses can be offered corresponding to the research questions posed earlier. First, it is postulated that the potential economic gains from mitigating energy-environment problems at the household scale are highly significant in aggregate, and that this area demands the highest priority for policy-making. This runs somewhat counter to the greater preoccupation of policy-makers and analysts in the past two decades with environmental problems in the power generation sector. Significantly, if this hypothesis is correct, then not only will effective mitigation strategies yield economic efficiency gains, but they will also deliver equity benefits by improving the welfare of low-income households. Thus the potential exists for convergence between efficiency and equity objectives, both of which enjoy a high priority at a political level in South Africa.

The second hypothesis is concerned with methodology: it is suggested that an environmental economic analysis of externalities can make an important contribution to the development of sound energy and environment policy, and can assist in the re-orientation of South Africa's development path along more efficient lines. At the same time, however, it is suggested that

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[5] Christie (1984) offers a comprehensive analysis of the ways in which the electricity supply industry developed in response to the interests of capital and the white minority. See also Horwitz (1994).

there are likely to be serious limitations inherent in such an analysis, both at a technical level - due mainly to data constraints - and at a fundamental level, particularly with respect to questions of income distribution and equity, which are of central concern in South Africa's policy discourse. Thus it is postulated that a cautious deployment of an environmental economic framework is warranted.

3. Originality and methodology

There are two respects in which this thesis contributes to the body of economic and development literature. Firstly, there have been very few environmental economic studies of externalities in South Africa's energy sector nor, indeed, in other developing countries. This is the first time externalities in South Africa's household energy sector have been analysed in any depth; apart from one prior analysis,[6] the same applies to the electricity generation sector. There have certainly been no comparative studies to-date of externalities in these energy sub-sectors. It is also the first time air pollution modelling has been undertaken in South Africa with a view to performing an economic evaluation of air quality impacts.[7] As such, this study is therefore unique in South Africa and, at the least, unusual amongst developing countries.

Secondly, almost the entire body of economic literature on externalities and related issues originated outside of developing countries, mostly in North America and Europe. Social and economic conditions differ considerably in these contexts and this, inevitably, has important implications for the application of the theory to developing country contexts. This thesis, in its attempt to apply the theory to South Africa's energy sector, offers perspectives which have not been particularly well-documented in the environmental economics literature.

The methodology followed in the thesis is drawn, in the first instance, from mainstream environmental economics thinking. The body of the thesis comprises two empirical applications of environmental economic techniques to the electricity generation and household consumption sub-sectors. Subject to constraints of data availability, the analysis follows relatively conventional approaches for externality evaluation. In addition to this, however, the thesis also considers the weaknesses and limitations of adopting such an approach in the particular political-economic context of South Africa, as a typical middle-income developing country.

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[6] Dutkiewicz & de Villiers (1993) made a comparison of the social costs of various generation options; this study, however, was very brief and did not involve serious economic analysis.

[7] This study uses an air quality model, contained in a larger computer model, called EXMOD - this is described in Appendix 3, and applied in Chapters Three and Four.

An important methodological consideration relates to the question of the breadth and depth of the analysis. Given that the research questions involve a comparative study of externalities in two sectors, the tendency in the thesis is unavoidably towards greater breadth. In making this trade-off, clearly the depth of analysis is less than it would have been had only one sector been analysed. Nonetheless, both sectors are analysed in sufficient (but not exhaustive) depth to permit meaningful comparisons to be made and conclusions to be drawn.

The subsequent chapters contain significant amounts of data regarding both sectors; the research methodology reflects a balance between primary data collection and the use of existing data sources. The breadth of the study precludes the collection of large amounts of primary data for any single external effect: beyond a certain minimum amount of data, the diminishing returns resulting from in-depth investigation in any single area militate against this approach. In order to make explicit the criteria for inclusion and exclusion of externalities in each sector, a classification methodology has been developed and is described in more detail in Chapter Three.

With regard to environmental valuation, there have been very few (if any) efforts to value environmental elements of any kind in South Africa. In North America, by contrast, there have been hundreds (even thousands) of valuation studies, with the result that externality studies there can draw upon a large existing data set. The paucity of such studies in South Africa poses particular challenges for externality quantification exercises such as the present one; this theme is addressed again in the final chapter when methodological issues are considered.

Finally, it should also be noted that the definition of `environment' employed in this analysis is one which explicitly includes humans and their surroundings. Thus a human-centred conception of the environment is adopted, not to the exclusion of the natural environment (birds, bees, trees and the like) but explicitly including the human living environment. Indeed, as will become apparent, this anthropocentric view informs much of the subsequent analysis. This is seen to be appropriate for the specific context in which South Africa finds itself at present.

4. Outline of the thesis

This thesis sets out to analyse the external costs of energy production and consumption in South Africa through a critical application of environmental economics theory to the energy sector. To begin with, Chapter Two reviews the literature, focusing on the large body of writing which has emerged in recent years on the theory and application of environmental economic approaches to externalities. Studies of energy externalities reported in the literature are also reviewed.

The focus of analysis then narrows, with Chapter Three identifying and describing the main externalities arising in South Africa's electricity generation sector. The following chapter quantifies, in economic terms, the main externalities for which sufficient information exists, producing a range of monetary estimates.

In the following two chapters (Chapters Five and Six), the focus shifts away from electricity generation and addresses externalities arising from the consumption of energy by low-income households. In Chapter Five, the most significant environmental impacts are described and analysed, and in the next chapter, they are quantified in economic terms.

Having assessed and made monetary estimates of externalities in the electricity generation and household consumption sectors, the discussion broadens again in Chapter Seven to consider the implications of the preceding analyses: firstly, in terms of energy-environment policy, and secondly, in terms of limitations and criticisms of the methodologies adopted. This critique is informed in many respects by a political economic perspective. The final chapter also draws together the preceding analysis, making explicit the implications of the analysis for the two key questions which were presented in this chapter.

Chapter Two

Externalities and environmental valuation: a literature review

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1. Introduction

There are many ways of taking account of environmental considerations in development policy, perhaps as many ways as there are intellectual traditions. This chapter reviews the conceptual approach suggested by the environmental economics literature, with particular reference to the issue of externalities. The review addresses three aspects of the literature: firstly, the theory of externalities and environmental regulation, secondly, the valuation of environmental services and damages, and thirdly, practical studies of energy sector externalities.

Although it is only in the past two to three decades that environmental issues have been a central concern of economic analysis, the concern with the environment and natural resources is not new. In many respects, environmental and resource issues have been a major concern of economists since the emergence of the modern discipline in the 1700s. The fundamental problem in economics involves the satisfaction of large or infinite demands in the face of limited resources. This concern with scarce resources has always been at the heart of economic thinking and can be traced back to the work of the Classical economists in the late eighteenth and nineteenth centuries. From the inquiries of Malthus (1798), Ricardo (1817) and Mill (1865), who concerned themselves with absolute and relative resource scarcity, to the interest of twentieth century economists in natural resources and their optimal rates of extraction, resource issues have been at the centre of economic thinking.

Notwithstanding this long history, it was only with the birth of an environmental consciousness in industrialised countries in the 1960s, that many economists discovered an interest in environmental problems. In contrast with the focus of the Classical economists, however, the problem began to be identified increasingly with `the environment' *per se*, as opposed simply to the question of resource scarcity.

The effect of the increased intellectual activity among economists concerned with environmental problems was the emergence of a new sub-discipline devoted to addressing these issues, namely resource and environmental economics. This branch of economics been an area of rapid intellectual advance and practical application, with the result that it has now developed a clear identity as a separate sub-discipline within the economics tradition. Whilst much of the writing in this discipline has its roots in the neo-classical tradition, many economists have presented strong critiques of existing modes of analysis and production, and some have suggested alternatives. Significant amongst the critiques of existing models of economic activity, are economists such as Daly (1973, 1990) and Georgescu-Roegen (1966, 1971) who, taking the second law of thermodynamics as their point of departure, have argued that the finite nature of the earth's resources means that `sustainable growth' is not possible, and that it is necessary to accept limits to the continued growth of the global economy in its present form.

In similar vein, some have taken a `materials balance approach' to the analysis of economic activity (for example, Kneese et al 1970). In this view, waste products and externalities are seen not as `freakish anomalies' but as inherent and normal parts of production and consumption activities, which should be dealt with not only through improved measures of their external costs, but also through more systematic and comprehensive analysis of these stocks and flows in economic activity (1970: 14).

Although there has been considerable intellectual activity since the 1970s around issues such as these, all under the broad title of `resource and environmental economics', the mainstream of environmental economics activity has been concerned with the analysis, in a neo-classical framework, of more conventional issues such as air and water pollution (for example, Baumol & Oates 1975, Kneese 1984), utilisation of renewable and non-renewable resources (for example, Barnett & Morse 1963, Dasgupta & Heal 1979, following early classics such as Hotelling 1931) and environmental valuation (for example, Freeman 1993, Johansson 1987, Pearce 1993). Although almost all of these areas are relevant to the energy sector, in view of its resource-intensive base, its pollution impacts and materials flows, and its pervasiveness in the economy, it is necessary, for present purposes, to focus on the literature which has greatest relevance to the questions addressed in this thesis. Consequently, this chapter reviews the literature as it pertains to environmental externalities in terms of the following divisions:

- the theory of externalities and environmental regulation;
- the valuation of environmental services and damages;
- practical examples of externality studies in the energy sector.

Each of these areas represents a huge body of knowledge and experience which cannot be comprehensively reviewed here; instead, a brief overview of the main concepts and approaches in each area is provided. Specific aspects of the literature are explored in more detail in later chapters, alongside the relevant externality analyses.

2. The theory of externalities and environmental regulation

2.1 Externalities, private and social costs

A central concept in the economic analysis of environmental impacts is that of an externality. A considerable body of literature has accumulated on this subject. Externalities were first analysed in depth by Pigou (1920), but have received much more attention in the latter half of the century (Mishan 1971, Baumol & Oates 1975, Fisher & Peterson 1976, Cornes & Sandler 1986, Johansson 1987, Cropper & Oates 1992).

An externality or external effect can be either positive or negative, although policy is most frequently concerned with the latter because of the implied welfare loss. In its earlier usage, the term was sometimes defined so broadly as to include most sources of market failure (Mishan 1971: 6, Baumol & Oates 1975: 16), although in its more contemporary usage, it generally refers to a situation where two conditions are met:

- One individual's (or entity's) utility is directly affected by variables determined by another agent, who acts without particular attention to the effects on the other's welfare; more formally, the first entity's production function includes variables whose values are determined by the second entity.
- Also, the decision-maker causing this change in welfare does not compensate or appropriate this change in the utility of those experiencing it (Baumol & Oates 1975: 17).

Thus an externality arises, for example, where a productive facility causes the emission of pollutants or waste products which, in turn, impact upon human health or environmental elements which have value for humans (such as agricultural crops), where the costs of those impacts are not captured in the market relationship between the producer and its customers, and those who bear the costs are not compensated in any way.

Several types of externalities may be identified. One distinction is between private (depletable) and public (undepletable) externalities (Baumol & Oates 1975: 19, Hartwick & Olewiler 1986: 386).[1] In principle, private externalities are those in which only two or a small number of agents are involved, and the external diseconomies are fully absorbed or appropriated by the party or parties. However, examples of negative private externalities are not common in practice since they are easily internalised through bargaining between the agents.

Public externalities, by contrast, are characterised by their non-excludability or non-appropriability: in other words, consumption of the good (or bad) by one individual does

[1] Baumol & Oates prefer the terminology given in parentheses because of the multiple and ambiguous interpretations accorded the terms `public' and `private goods' (1975: 19).

not reduce its availability to others. For example, a view which is impaired by pollution from power stations is a public externality, since one person's consumption of the view does not decrease the consumption possibilities of any other person. In practice, most externalities are of a public nature since negative effects usually impact upon a large number of individuals simultaneously. The relevance of this is that private negotiation (between affected parties) is not effective, or is very costly, and therefore some kind of government action is called for to internalise the external effects.

A special case of a public good (or bad) is known as a Tiebout good (after Tiebout 1956): in this case, the effect is not shared evenly between people (as in the case of a pure public good - such as a view). Rather, the good (or bad) is unevenly distributed, so that its quality may change, for example, according to distance from a source of emissions. Moreover, Tiebout goods can generally be more easily valued with reference to secondary markets, such as property prices which reflect the degree of exposure to a source of pollution.

A further distinction which is sometimes made in the literature, is between technological and pecuniary externalities (Baumol & Oates 1975: 28). The former are considered \`true' externalities which directly affect the allocation of resources and which usually cause a shift in a victim's production function - and a consequent misallocation of resources if the externality is not internalised. Pecuniary externalities, on the other hand, refer to the changed financial circumstances of one individual as a result of changes in prices of inputs or outputs caused by another agent in the economy. In many respects, these effects reflect the normal competitive mechanism through which resources are reallocated in response to changing prices (ibid: 30). Because pecuniary externalities do not cause a misallocation of resources, they are generally not analysed in detail; attention is usually focused on conventional (technological) externalities.

In addition to the above distinctions, an externality is said to be \`Pareto-relevant' if it prevents the necessary conditions for Pareto-optimality from being achieved, the latter being defined as an economic state in which an increase in utility cannot be achieved without decreasing someone else's utility (Baumol & Oates 1975: 18). Thus Pareto-relevant externalities may lead to outcomes which are less than ideal: the presence of an externality represents the failure of the pricing system to reflect all the costs of producing a given item, and therefore consumption decisions will be based on sub-optimal prices. Consequently, it is possible that resource allocations could be inefficient, that is, less will be produced than is possible with given inputs.

Pareto-irrelevant externalities are less common, but do exist. If the transaction costs of internalising an externality are greater than the external costs themselves, the continuation of an \`uncorrected' externality may be consistent with a Pareto-optimum. Using an old example from Baumol & Oates (1975: 23), in the case of coal which fell from trains travelling through the country-side, the costs of collecting it or of preventing individuals from doing so for their own purposes, may have been higher than the value of the coal itself (the external cost of its loss) and therefore it would be efficient to ignore that loss - in other words, the externality is Pareto-irrelevant.[2] A more relevant example for present purposes would arise where the costs of monitoring and controlling emissions from multiple non-point sources exceed the costs imposed by those emissions.

A further distinction which has been made in the literature, is between stock externalities and flow externalities (Pearce 1976: 101). The latter refers to conventional external effects where the relevant variable is the flow of pollution into the environment. In the case of a stock externality, by contrast, the relevant variable is the stock of the pollutant in the environment: for instance, cadmium is hazardous to human health, but only after it has accumulated to a particular threshold level in the food and water chain. Thus, marginal damages from additional flows of cadmium into the environment will depend upon whether they occur below or above the threshold level. This presents difficulties for cost-benefit analyses, since the marginal damage function will intersect the horizontal axis at a non-zero level and, moreover, where the environment's assimilative capacity is zero, it is difficult or impossible to reduce the stock of cadmium. Abatement strategies are therefore seriously constrained.[3]

An important way of defining externalities for purposes of this thesis, is with reference to the divergence between private and social costs (Pigou 1920, Pearce & Turner 1990: 66). Private costs are those costs which are borne by the producer of the good, whereas social costs go further than this to include the full costs of producing or consuming a commodity, which may be borne not only by the producer but also by other groups in society at large.

The difference between private and social costs, then, represents the external cost or the externality which is borne by society at large. This means that the market-based relationship between the producer and consumer, as reflected in the price, does not reflect external costs

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[2] This example was also used by Baumol & Oates as an illustration of a private (depletable) externality: clearly, collection of the coal by one individual reduces the amount available for other individuals (1975: 20).

[3] In some respects, greenhouse gases can be considered to be stock externalities, since it is less the flow of gases into the atmosphere which is relevant, than the total stock which has accumulated over time. Because the assimilative capacity of the environment (in particular, forests and oceans) is low in relation to the current rate of emissions, the existing stock of gases can be taken as given for the foreseeable future. Of course, from an ecological perspective, the optimum requires the stock of greenhouse gases to be reduced.

since these are borne by other members of society. Frequently, the distribution of these costs is such that they are not borne equitably by society as a whole, but fall more heavily on some groups or classes than others.

The principles of external costs are illustrated graphically in Figure 2.1. An individual producer in a competitive product market faces a horizontal marginal revenue curve (MR) equivalent to the price of the commodity, and a marginal private cost curve as shown by MPC. If the producer seeks to maximise its surplus, it will clearly produce at the point where its marginal revenues and costs are equal: point B, that is, at a level of output equal to Q_1. With the assumption that it seeks to maximise profits, it makes little sense to deviate to either side of that point. The marginal external cost at any level of output is given by the vertical difference between the MSC (marginal social cost) and MPC lines, and so the total external cost at the individual's optimum (Q_1) is equal to the triangle ODB.

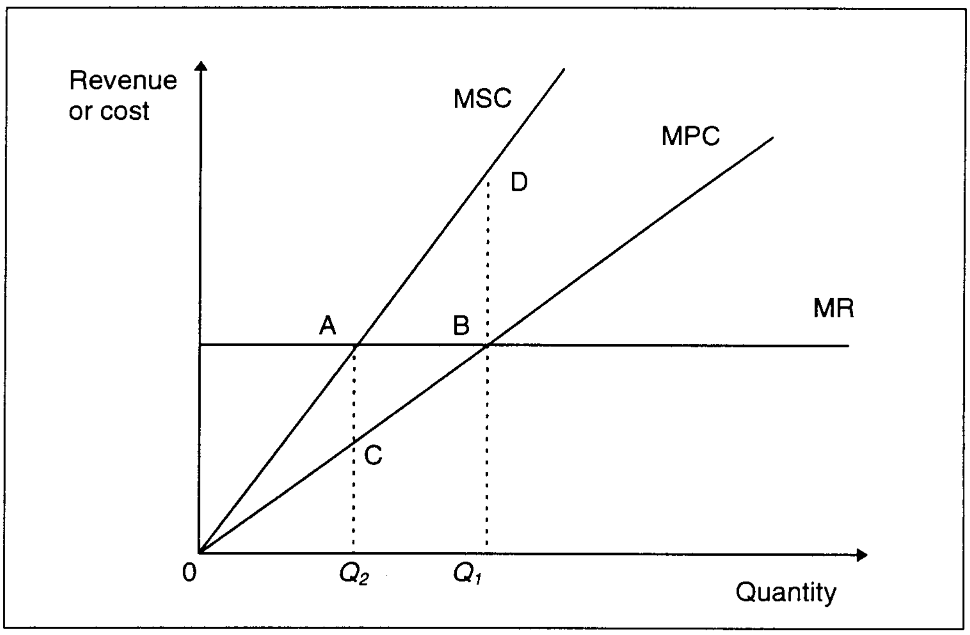

Figure 2.1 Illustrative marginal revenue (MR), marginal private cost (MPC) and marginal social cost (MSC) curves in a competitive market[4]

Whilst this may be optimal from the individual producer's perspective, it is not ideal for society as a whole. The socially optimum level of output will be at a lower point, that is,

[4] The figure illustrates the less common situation of a competitive market. In the case of non-perfect competition, the marginal revenue curve would slope downwards - exactly the same principles would apply in that case.

point A, where the marginal benefit equals the marginal social cost, equivalent to a level of output of Q_2. At this point, the benefits to society are exactly equal to the costs to society of producing the commodity, and so, if the objective is to maximise social welfare - rather than individuals' surpluses - then it does not make sense to deviate to either side of Q_2.

Whilst this represents a simple application of marginal analysis using conventional economic logic, it is significant that the economic optimum will still be associated with a non-zero level of pollution, the external costs of which are shown by the triangle OAC in the figure. This economic optimum will frequently not coincide with an environmental optimum. The latter would generally occur at, or close to, a point of zero pollution (corresponding with zero activity), if the criterion for environmental quality is related in any way to a natural or pristine (that is, uninhabited) environment. These alternative points of departure may account, to some extent, for the different stereotypical approaches of economists and environmentalists to the issues of the `limits to growth' and steady-state economies.[5]

The theory as outlined above, therefore suggests that externalities constitute an important source of market failure: in the first instance, because of their potential to hinder a Pareto-efficient outcome. Secondly, although generalisations on this score are risky, the burdens of external costs are seldom spread equitably across society and often fall disproportionately on social groups which are least able to afford them. Thus the distribution of income may be further skewed in the presence of externalities.

Thirdly, a higher rate of productive activity usually translates into more rapid consumption of resources, including non-renewable ones, and this undermines goals of environmental sustainability. Collectively, the effect of externalities may therefore be in conflict with goals of economic efficiency, social equity and environmental sustainability - the three criteria which are often taken to constitute the foundations of `sustainable development'. Consequently, some policy response is usually warranted if these goals are to be pursued.

2.2 Achieving the optimal rate of pollution

A range of possible responses exists to deal with externalities, and environmental economic analyses are frequently used to assess the relative merits of these. For present purposes, four

[5] At a slightly more sophisticated level, if the dynamic nature of the environment is accommodated by recognising its assimilative capacity and adaptive abilities, then it may be that the `environmentally optimal' level of pollution will be above zero and thus not as widely divergent from the economically optimal level of pollution (Pearce & Turner 1990: 65).

such broad approaches can be identified: *laissez-faire* bargaining options, environmental standards, pollution taxes and subsidies, and marketable emissions permits.

2.2.1 *Coasian bargaining options*

The first and most extreme pollution control approach entails an entirely free market response and stems from the analysis of Coase (1960). The Coase Theorem suggests that if a polluter and pollutee could bargain over the level of pollution and output, then the socially optimal level of output will be achieved without any need for government intervention. This occurs regardless of whether property rights over the environment vest in the polluter or pollutee, because, in theory, each party would be able to compensate the other for any increase in environmental costs suffered or revenue lost by them, as the case may be.

While the theoretical basis of the Coase Theorem has received some minor criticism,[6] its *laissez-faire* approach has been more seriously challenged from a practical perspective. Although a full critique of the Theorem need not be restated here, the following problems have been documented in the literature:

- First, it cannot apply where there are large numbers of polluters and pollution victims, let alone where the pollution emitters cannot easily be identified, because of the high transaction costs involved in bringing parties together to make their bargains. This case has also been referred to as one involving 'many receiver diseconomies' (Zeckhauser & Fisher 1995: 119). If these costs are so high as to offset a party's possible bargaining gains, then it would be 'optimal' for that party to withdraw from the process and accept its environmental costs. This represents an 'unfalsifiable theory' since an optimal situation results whether a bargain takes place or not (Pearce & Turner 1990: 75) and makes little intuitive sense.
- Secondly, the Theorem is based on an assumption that markets operate efficiently, and specifically that price equals marginal revenue (P = MR). The theoretically optimum level of output occurs where marginal external costs (MEC, borne by the pollutee) equal marginal net private benefits (MNPB, accruing to the polluter); the latter, in turn, equals the difference between price and marginal cost (MNPB = P - MC) and represents the polluter's bargaining curve, determining the amount it should pay or accept as compensation. However, under imperfect market conditions price and marginal revenue

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[6] The Theorem implicitly assumes zero wealth effects: in other words, it ignores the fact that conferring property rights to one or other party represents a transfer of wealth to that party, and could therefore shift his or her demand curve to the right, thereby affecting the bargaining outcome (Tietenberg 1994: 47).

diverge, so that the requirement at the optimum level (for MNPB to equal MEC) will no longer be valid (ibid: 73).

- Thirdly, it is impossible for parties to bargain with each other where some of those affected by pollution are from future generations, who cannot possibly be represented themselves and on whose behalf present generations may not act ideally (Helm & Pearce 1990: 6).
- Where a pollution victim compensates the polluter for reducing its levels of output, the situation is opened for threat-making, whereby other potential polluters can also extract compensation by threatening to increase their pollution. Apart from questions of distributive justice, this cannot represent a Pareto-efficient use of scarce resources (Tietenberg 1994: 48).
- Finally, in contexts where property rights are not clearly defined, and especially where there are common property resources, it may benefit some polluters to pollute beyond the `socially optimal' level if they can get away with it - the `free rider problem' (Helm & Pearce 1990: 6).

In summary then, the Coasian market-based bargaining approach to reaching the socially optimal level of pollution has little applicability to pollution problems because its theoretical requirements are seldom matched by conditions in practice. This explains the relative scarcity of case studies of the bargaining approach compared to other approaches such as those outlined below.

2.2.2 *Environmental standards*

A second response to pollution problems, occupying the opposite end of the scale to the Coasian option, entails the setting of standards, which are usually established on the basis of health-related criteria (Baumol & Oates 1975: 134, Tietenberg 1994: 219, Pearce & Turner 1990: 102). These are the most commonly applied form of pollution control in practice, and may take the form, for example, of a maximum concentration of pollutants to be emitted in a power station's flue gases (often measured in parts per billion, ppb, or in micrograms per cubic metre, μgm^{-3}). Alternatively, the standards might relate to the maximum permissible ambient concentrations of pollutants, based on estimates of pollutant concentrations which are damaging to human health.

Environmental standards have one major advantage, which probably accounts for their widespread application in practice, namely their relative simplicity and ease of implementation. Standards are usually employed by government regulators in three circumstances: firstly, where the marginal external costs of pollution are relatively high and abatement costs relatively low (Cropper & Oates 192: 730); in other words, where the risk of over-control is low. Secondly, standards are frequently used where markets are weak or non-existent so bargaining solutions are not feasible, and where little institutional capacity exists in governments and regulatory agencies to devise, implement and operate systems of pollution taxes. This is an important reason for their use in practice, especially where other more complex and perhaps more efficient pollution control regimes cannot be implemented. Thirdly, standards may be the socially optimal approach where the pollutant concerned is so dangerous as to require an outright ban on its use.

However, strong theoretical arguments exist to show that standards seldom result in an economically efficient solution, except perhaps by chance (Baumol & Oates 1975: 135, Tietenberg 1994: 414, Pearce & Turner 1990: 102-106). This is largely because the standards cannot easily be set at a level which encourages polluters to produce at the economically optimum level (Q_2 in Figure 2.1), due to the lack of information on the part of regulators. Secondly, the imposition of standards coupled with penalties for transgressors will be less than effective if the enforcement function is poorly performed. In other words, polluters will assess the probability of being caught and having to pay the penalty, and base their production decisions on that assessment. In most cases where standards are applied in practice, the monitoring and enforcement of those rules and related penalties suffers from severe resource constraints, which further undermines the efficiency of the system.[7] A third disadvantage of environmental standards is that they provide polluters with no incentive to reduce pollution at levels of production below the acceptable limit, even though the marginal damage costs are usually positive below the point at which a penalty becomes payable.

On the whole, standards are not favoured by environmental economists because of the significant additional abatement costs which often result from their use as compared to approaches which make explicit reference to the costs and benefits of abatement. However, the advantages of standards-based approaches, which were alluded to above, are also acknowledged in the literature, leading to a recognition of the need for a mixture of policy responses depending on the specific circumstances (Cropper & Oates 1992: 699, 730).

2.2.3 *Pollution taxes and subsidies*

A third category of responses to environmental pollution, which falls between the extremes of the *laissez-faire* bargaining option and the standard-setting option, centres around the use

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[7] In South Africa, for instance, there are only seven pollution control officers who are responsible for monitoring and enforcing pollution control standards across the entire country - thus the probability of being caught transgressing standards is extremely low (not to mention that the costs of being so caught are also low).

of taxes and subsidies to correct for social (or external) costs not accounted for in private decision-making (Baumol & Oates 1975: 172, Helm & Pearce 1990: 5). This intellectual approach, developed from the work of Pigou (1920), entails the imposition of a tax to bring a polluter's cost function into line with the full social costs of production, including any external costs, and thus encouraging the producer to adjust output to the socially desirable level (Pigou 1920). This is the logic underlying the more contemporary term, the `polluter pays principle'.[8] Clearly, government agencies play a central role in the imposition and collection of taxes, and this represents a more interventionist response than that suggested by the Coase Theorem.

Similar logic was, at one time, applied to the use of subsidies whereby a polluter is compensated for its efforts to reduce polluting activities. At face value, the incentive inherent in a subsidy of R1 per kilogram of reduced pollution emission is equal to the incentive provided by a tax of R1 per kilogram of pollution actually emitted. However, scholars soon observed that taxes and subsidies are not equivalent in practice (Baumol & Oates 1975: 172, Hartwick & Olewiler 1986: 413, Cropper & Oates 1992: 681). In particular, the use of subsidies increases profits in the industry which will act as a signal to other firms to enter the industry, causing an increase in the supply of the pollutant. Also, a subsidy may allow firms that would otherwise be unprofitable to remain in the industry.

While this Pigovian approach has more relevance to practical pollution problems than the Coasian bargaining option, it also suffers from several limitations (Helm & Pearce 1990: 5, Baumol & Oates 1975: 154):

- Firstly, the approach has stringent information requirements; specifically, government regulators have to base the taxes on estimates of the marginal benefit functions of the polluter and the damage cost functions of the pollutees, which obviously represents a very onerous requirement; in other words, there may be an asymmetry of information between government officials and private polluters.
- The aim of the tax is to ensure that polluters face the true marginal cost of production, which therefore assumes that there are no other market failures which might distort these cost functions. This assumption is often not valid since environmental externalities

[8] This term is somewhat misleading, since the producer itself seldom bears the final cost: this is usually borne by consumers of its products to the extent that price increases are passed on (which obviously depends on price elasticity), and possibly by its upstream suppliers where the quantity and price of inputs is affected.

frequently occur in the presence of numerous other market distortions, such as oligopolistic conditions and taxes.

- Thirdly, it is usually assumed that regulatory agencies have only one goal, namely to achieve the economically efficient situation. However, this ignores the possibility that regulators may have other goals in practice (whether illicit or not), and also neglects other possible goals such as equity considerations.
- Fourthly, Pigovian taxes have also been criticised on the basis that they accept as a *fait accompli* that pollution occurs and that it is not bad in itself. This criticism is mostly identifiable with more radical environmental views.
- A more technical problem with the implementation of Pigovian taxes arises where there are non-convexities in the production sets, for example, where the damage functions reflect remedial action taken by pollutees at some level of pollution (Fisher 1981: 176, Burrows 1995: 243). This might lead to multiple equilibrium points, which poses practical difficulties in attempting to set taxes equal to the marginal external cost at the optimal level.
- Finally, pollution charges have heavy information requirements and a not insignificant administrative burden, which makes them inappropriate for dealing with short-term problems and crisis situations. In this context, environmental standards may be more appropriate since pollution taxes are unlikely to achieve the optimal level of pollution in a short time.

Notwithstanding these problems, pollution taxes are generally held to offer advantages over the use of standards, both in theory and in practice. The most important relates to the point made previously, namely that taxes are more likely to lead to an economically efficient solution (Baumol & Oates 1975: 134, Pearce & Turner 1990: 102). A second argument in favour of the use of taxes is that they will deliver positive social benefits, even in the context of uncertainty and inadequate information on behalf of the environmental regulator, provided the amount of the tax brings the polluter's private cost function closer to the social cost function. Thus, full information is not necessary for pollution taxes to deliver net benefits to society and, indeed, they can be revised incrementally and iteratively as better information becomes available regarding private and social cost functions.

2.2.4 *Marketable emissions permits*

The fourth and last category of pollution control responses suggested by the environmental economics literature also makes use of market-based mechanisms, namely marketable emissions permits (MEPs). As in the case of standard-setting, the environmental regulator specifies a maximum allowable level of emissions, and issues permits for that amount. In this case, however, the permits can be traded in a market.

A number of advantages to this option have been described in the literature (Tietenberg 1980, 1994: 240, Pearce & Turner 1990: 111-115, Cropper & Oates 1992: 688). First, MEPs will encourage a least-cost solution, because producers with low abatement costs will prefer to sell their permits and reduce pollution, while producers with high abatement costs will prefer to buy permits rather than invest in expensive abatement controls. Secondly, the system is flexible enough to allow the regulatory authority to change the desired amount of pollution, either by buying up permits itself if it wishes to reduce emission levels, or by issuing new permits if the limit is to be increased. Thirdly, MEP systems do not require adjustment in inflationary environments because they are not denominated in monetary terms, unlike taxes which have to be adjusted to stay constant in real terms. Fourth, tradable permits can theoretically allow interested third parties to participate in the market if they desire. For example, environmental organisations might buy up permits if they prefer for total emission levels to be reduced. In practice, however, there are few such organisations with the resources to compete with private producers in MEP markets. A fifth advantage quoted in favour of MEPs is that the response to changing demand for permits, due to lumpy investment profiles, changing goals of regulators, or uncertainty on the part of polluters, is simply for their prices to adjust, while the environmental standard remains fixed.

Tradable permits have been employed in several countries, most notably the US, where the 1990 amendments to the Clean Air Act established a national cap on sulphur dioxide emissions and an allowance trading system for these emissions (Cropper & Oates 1992: 689). Whilst practical experience with these systems is still relatively limited, albeit growing, indications are that the tradable permit approach can be less cost-effective than promised, because of the existence of high transaction costs (Stavins 1995: 9). These costs are associated with creating and managing a market in which prospective buyers and sellers of the permits can meet and trade, and with ensuring that permit holders do not exceed their allowable quotas.

A second disadvantage with MEP systems is that in a stable market, they do not contain any in-built incentive for holders to innovate or introduce less-polluting technologies; taxes on the other hand, always contain an incentive for producers to reduce their pollution emissions. Thirdly, although the MEP market is supposed to be self-regulating through the price mechanism, the system usually still has high administration costs because of the need to manage the market in the permits and to ensure producers do not exceed their permitted quotas (in the same way that this is required for conventional regulation or for collection of pollution charges). Fourth, MEPs are likely to be less effective in monopolistic and oligopolistic markets because of the limited number of producers trading in the permits. Furthermore, where trading in permits is `thin', change will happen very slowly and the least-cost situation may be slow to materialise. Finally, in an imperfect market, a monopoly or oligopoly might seek to monopolise the permits, not on the basis of its marginal abatement costs relative to those of other producers, but simply as a means of raising entry costs for new firms and increasing its control over its industry. This may lead to an economically inefficient situation and, moreover, to an inequitable concentration of resources in the larger producers to the detriment of smaller producers.

Having reviewed four main categories of responses to pollution problems, it is clear that each has its own strengths and weaknesses, with only the Coasian bargaining approach offering little value to situations where pollution problems arise in practice. As suggested by Cropper and Oates (1992: 699) it may be misleading to lump together in `cavalier fashion' command-and-control methods of regulatory control and to contrast them with least-cost approaches usually associated with economic incentive systems: rather, a more selective and critical consideration of the merits of all approaches is urged. When policy responses to externalities in South Africa's energy sector are considered in the final chapter of this thesis, it will become apparent that this mixed regulatory approach offers the greatest potential in practice.

3. The valuation of environmental services and damages

A central requirement in the analysis of externalities and the development of policy responses, is that they be valued in economic terms. This area, the valuation of environmental goods and services, has been a major area of work in environmental economics over the past two decades or more. Numerous methods and techniques have been developed in the literature, and are applicable to a range of different circumstances. Before describing the most important valuation methods, however, it is relevant to consider briefly the debate around whether valuation of environmental factors is possible and desirable. This is relevant particularly in view of the criticisms often directed at environmental valuation, mostly by non-economists.

3.1 The debate over whether environmental valuation is appropriate

Environmental valuation has attracted much critical attention, from scholars both within and outside the economics profession. These criticisms include (mainly from within the profession) technical problems with the valuation methods, many of which will become apparent when the main methods are described shortly. The second important category of criticisms, coming usually from non-economists, is more fundamental, and revolves around the question of whether monetary values can be assigned to the environment in any meaningful way at all (Redclift 1987: 38, Amin 1992, Jacobs 1991: 204, Knill 1992). Some scholars adopt an ethical or moral opposition to environmental valuation because they find it difficult to reduce environmental components to money values. One of the most contentious areas is around the valuation of human lives, since finite economic values do not sit comfortably with most ethical and philosophical views of the value of human life.

The most important and simplest reason for undertaking valuation exercises in environmental economics is because the unit of analysis in economics is expressed in monetary terms; thus for environmental issues to be incorporated into neo-classical analysis, they need to be expressed in monetary terms.

More specific reasons for attempting to make valuations of environmental factors include the following:

- Firstly, environmental services or damages are usually taken into account in decision-making processes through some implicit evaluation on the part of those involved. Where environmental impacts of decisions are not given explicit attention, they are either ignored altogether (reflecting a very low valuation), or accounted for in some qualitative way which can lead to ambiguous or normative recommendations. In this context, economists usually suggest that it is better to at least make explicit the value judgement which is being made (Freeman 1993: 10, Pearce et al 1991: 5, Jones-Lee 1994: 293).
- In order to devise economically efficient pollution control strategies such as taxes, it is necessary to have an idea of the marginal external costs (MEC) resulting from such pollution; in other words, it is necessary to place an economic value on the environmental damage caused by pollution (Pearce & Turner 1990: 120). This represents a microeconomic argument for valuation.
- Thirdly, by attaching economic values to environmental policy issues, the importance of those issues in the wider macroeconomic context becomes highlighted. For instance, estimates of environmental damage in West Germany, in the form of air, water and noise pollution, amounted to about 6% of the country's GNP for 1985 - clearly not insignificant (ibid: 124). By placing some economic value on the environment, it can be argued that a better representation is provided of peoples' personal valuations of the environment, as evidenced, for example, by the growth in environmental consciousness internationally. This becomes particularly important for the purpose of environmental accounting.
- Fourthly, economic valuation of environmental changes is essential for the performance of a cost-benefit analysis, a standard and widely-used tool in decision-making (Layard & Glaister 1994). If the environmental impacts of an investment decision are not reduced to monetary terms, then they cannot be included in the stream of costs and benefits associated with the project and, instead, have to be accounted for in some other way in

the decision-making process - frequently through cost-effectiveness analysis in which only the cost side of the equation is quantified (Bojo et al 1992: 81).

- Another argument often raised by economists is that the valuation process does not entail placing an economic value on the environment itself, but rather reflects individual preferences for or against an environmental service or problem, or of trade-offs between different risk-scenarios (Pearce et al 1991: 4, Freeman 1993). Peoples' preferences for many different goods and services are expressed every day in monetary terms, and environmental valuation attempts to introduce a level of consistency and comparability with respect to human preferences for other items.

Against this background, it is suggested that the pursuit of sound environmental policy is not furthered by absolutist approaches which either attempt to place values on every environmental problem, or preclude any valuation efforts at all. Rather, it is suggested environmental valuation can serve a useful purpose in specific instances, provided all assumptions and limitations are made explicit. This is the position taken in this thesis; the following sections describe the concepts and methods used in environmental valuation.

3.2 Environmental valuation: theoretical measures of utility and welfare

The economic valuation of environmental goods and services follows the basic tenets of neo-classical welfare economics in which each individual is believed to be the best judge of his or her welfare, and in which each person's welfare depends upon his or her consumption of goods and services (Freeman 1993: 6). These goods and services include not only those produced by firms or governments, but also non-market goods and those produced by the natural environment: for example, breathable air, potable water and visual amenity. Further, it is assumed that people have some set of preferences for alternative combinations of goods and services, and that it is possible to substitute one element for another, without changing the overall level of utility for that person. Thus, from individuals' willingness to make trade-offs between various elements in their preference sets, some of which may be marketed goods and other not, something is revealed about the values people place on those goods and services.

These trade-offs can be expressed in terms of a person's willingness to pay (WTP) or willingness to accept compensation (WTA). WTP represents the maximum amount a person would be prepared to pay to enjoy an increase in some (environmental) good or to prevent a decrease in that good,[9] whereas WTA is the minimum amount a person would require to

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[9] This `good' can be conceived of very widely and could include, for instance, the welfare of the poorest people in society (as is the case with people whose collective or social interests are significant).

forego an improvement in welfare that would otherwise have occurred, or to tolerate a reduction in welfare (ibid: 8). In principle, WTP and WTA need not be exactly equal, since the former is, by definition, dependent upon an individual's income, whereas the latter could have a very much higher value, much less constrained by income than WTP.

Several approaches exist, in theory, by which individual welfare changes can be measured. The first is the ordinary consumer surplus: the difference between the amount a person is willing to pay for something and the amount they actually pay; in other words it is equal to the area below a simple demand curve and above the horizontal price line (Freeman 1993: 46, after Marshall 1920: 124). Given that it reflects an individual's expression of the utility attached to consumption of the relevant good or service, it forms a part of the economic value, in addition to the amount actually paid.

Two other measures of welfare change, which are refinements of the ordinary consumer surplus (after Hick 1943), are the compensatory variation (CV) and the equivalent variation (EV) approaches. Both are based on a Hicks-compensated demand curve - this refers to a demand curve in which only the substitution effect of a change in relative prices is observed, with a compensating adjustment being made to the individual's income to leave him or her at the original level of utility (Freeman 1993: 52). The CV calculates WTP or WTA based on peoples' ex-ante or pre-project utility, in other words, it asks what compensating payment (or change in income) is required to make a person indifferent between their original situation and a new set of prices or conditions (Freeman 1993: 48, Johansson 1990: 35). On the other hand, the EV measure is based on the expected ex-post utility, or in other words, it asks what payment (or change in income) is required to lead to the same utility as the proposed project or change in prices (ibid).[10] In theory, each of these measures results in a different valuation of welfare change; given the practical difficulties of observing the CV and EV measures - because they are based on an unobservable Hicks-compensated demand function - the most frequently-used measure is the ordinary consumer surplus (Freeman 1993: 61). The use of the ordinary consumer surplus was given impetus by Willig (1976) who demonstrated that the approximation error for the CV or EV measures was, in most cases, negligible; however, as Freeman (1993: 61-70) subsequently showed, this conclusion also required qualification.

[10] Two other measures also exist, namely compensating surplus and equivalent surplus; these are not described here as they refer to a restrictive situation in which an individual is restricted to consuming a specific quantity of a good even though its price changes (Freeman 1993: 49).

3.3 Environmental valuation in practice

Aside from the theoretical complexities of measuring welfare change (and therefore environmental values), significant difficulties arise in practice. Environmental valuation is obviously not simple because of the different nature of the benefits and costs associated with environmental goods and services. For instance, the cost to a poor rural household of a woman's labours in collecting fuelwood, is of a very different nature to the benefit derived by an affluent household in Europe from the knowledge that parts of the Amazon forest have been saved from logging. Consequently, environmental economists have put forward the concept of total economic value, and offered a disaggregation.

Put simply, total economic value (TEV) is seen to comprise a number of components (Pearce & Turner 1990: 131):

TEV = actual use value + option value + existence value.

Use value is considered to be the value a person derives from actually using part of the environment, for example, by an angler fishing from a river. Option value is more complex and represents the value of potential use of the environment; for example, it might relate to the value placed by a person on the option of being able to fish in a river at some future date. Existence value is even more difficult to define accurately, but is a value which resides intrinsically in something, quite apart from human use of that thing. For instance, a person may never see a particular species of fish or make any use of such a fish, but might attach value to knowing that it exists in a river system. While the idea of existence value presents practical measurement difficulties, it is significant that it is conceptually useful in accommodating many environmentalist concerns which are often unrelated to current or future use of resources.

Environmental valuation methods can be categorised as being either direct or indirect;[11] various methods described in the literature are mentioned briefly below.

3.3.1 *Direct valuation approaches*

The first of three categories of direct valuation approaches is termed the contingent valuation method (CVM), in which a hypothetical market is constructed in order that individuals may reveal their preferences for a given environmental `good' or `bad'. Typically, CVM studies attempt to gauge one of several possible values: willingness to pay

[11] The terminology used to describe these valuation methods varies in the literature; other categories sometimes used include `stated preference' measures (direct valuation approaches such as contingent valuation and ranking), and `revealed preference' measures (indirect valuation approaches such as the travel cost and hedonic pricing methods).

(WTP) for receiving a benefit, or for avoiding a cost, and willingness to accept compensation (WTA) in respect of a cost incurred or a benefit not received (Johansson 1990: 35). People are asked directly about their WTP or WTA through questionnaires or surveys in an attempt to reveal their individual preferences, which are then aggregated or extrapolated to yield a total value of the environmental factor.

A large body of literature exists around the use of contingent valuation methods, and their strengths and weaknesses are well-documented (Freeman 1993: 165, Pearce & Turner 1990: 148-153, Johansson 1990). A technical attraction of CVM is that it is theoretically applicable to any environmental question since it elicits `valuation bids' from people as if they were participating in an active market for the environmental element concerned.

Problems with the CVM approach are, however, not easily dismissed in practice (Cropper & Oates 1992: 710, Pearce & Turner 1990: 149-153). Firstly, it is notoriously difficult to ensure that the responses given by people are truthful or accurate, especially where they might benefit by giving a response which adjusts the aggregate valuation of a public good from their personal valuation. This is known as the free rider problem and is classified as a form of strategic bias. Secondly, survey approaches can suffer from design bias, insofar as the starting point in the bidding process may influence a respondent's valuation, or as people may have different sensitivities to the `vehicle' chosen as an instrument of payment (for instance, entrance fees, property rates or price increases). A third problem is referred to as hypothetical bias, which results from the fact that peoples' bids will differ from the prices they would have been prepared to pay in an actual market. This is primarily because participation in hypothetical markets carries no risks or real costs of making poor decisions. A fourth problem area relates to operational bias, in other words, differences in operating conditions between real and fictitious markets. For instance, people may not know enough about the issue being surveyed, might have had a prior experience which differs from that being evaluated, or might have little or no experience of the bidding processes adopted in market transactions.

A fifth problem with CVM approaches which has been extensively discussed in the literature, is the divergence between WTP and WTA (Freeman 1993: 177, Knetsch & Sinden 1984, Cropper & Oates 1992: 710, Pearce & Turner 1990: 156). Many CVM studies have found a wide divergence between valuations yielded by WTP and WTA, with the latter consistently exceeding the former, sometimes by several factors (Freeman 1993: 177). A simple example can illustrate the divergence between WTP and WTA:[12] if the question aims

[12] This example is pertinent to the valuation of externalities, such as those in subsequent chapters, where human health outcomes are significant.

to establish the amount an individual would be willing to pay (WTP) to avoid his or her own death, the answer is most likely to approximate `all of his or her wealth'. On the other hand, if the question aims to establish the willingness to accept compensation (WTA) for a certain death, the response is most likely to be that no amount is sufficient to compensate for death - in other words, an infinite valuation. Clearly, there is a wide divergence between WTP and WTA in this example.

Several explanations for these differences have been offered, including suggestions that problems lie with the format of hypothetical bidding questions which cause disparities between revealed (as distinct from true) WTP and WTA (Freeman 1993: 178). Other explanations focus on the existence of an `endowment effect', which suggests that individuals' valuations of gains or losses relate to some reference point, usually the status quo (Thaler 1980, Kahneman & Tversky 1979). Negative deviations from this original endowment (of environmental quality) will be given higher values than equivalent positive deviations. This is consistent with the findings in actual CVM studies. Further, this view suggests that the manner in which gains and losses are secured is important, in that losses which are `imposed' will attract a higher value than an equivalent loss which is voluntarily incurred (Pearce & Turner 1990: 157).

In addition to these criticisms which are well-documented in the literature, CVM methods face difficulties in the aggregation process where the distribution of income across society is unequal to begin with. For instance, poor rural inhabitants of an area proposed as a conservation area may be asked to make a bid of their WTA compensation for loss of land due to the establishment of a nature reserve. On the other hand, wealthy urban residents' WTP for the right of use of the area might be a much higher amount, leading to the simple cost-benefit conclusion that the proposed conservation area is worth more than the currently inhabited area. The project would therefore be deemed to deliver a net benefit to society, and would be pursued. However, because the marginal utility of income of a poor rural person will be much higher than that of a wealthy person, the exercise of comparing the monetary valuations expressed in the respective WTA and WTP may not maximise societal welfare or satisfy equity goals. In principle, this refers to the need to make adjustments to individual valuations according to a social welfare function, taking explicit account of varying marginal utilities of income: in other words, a weighting factor may be applied to give greater weight to valuations of people lower down the income distribution (Layard & Walters 1994: 196). In practice, however this presents serious problems, and although weighting factors were sometimes used in the 1970s, this is infrequently done in contemporary studies (Little & Mirrlees 1994: 208). In this context, the problem remains consequential.

A second kind of direct valuation approach is the hedonic pricing method (refer Freeman 1993: 367). Most commonly applied to land and property cases, this method is based on the assumption that property prices reflect the stream of benefits flowing from ownership of the land, and that differing property prices are attributable, at least in part, to owners' willingness to pay for the relevant environmental characteristics related to the piece of land. By using appropriate statistical techniques it is possible to identify the price differential attributable to a particular environmental feature and obtain a valuation for it in this manner. Comparisons can be made on a time-series basis, or cross-cutting basis, with the former having been found to be more convincing (Rees 1992: 344).

A common variation of this approach is known as hedonic wage pricing, in which wage differentials are assumed to capture workers' valuations of environmental and other conditions of their jobs (Freeman 1993: 421). The logic behind this is that workers would not accept work under dangerous or unhygenic conditions, unless they were adequately compensated for those risks. All other things equal, in an efficient market all environmental risks would be fully internalised into workers' wages, so wage differentials could be isolated statistically, to reveal individual preferences for relevant environmental conditions.

In spite of the large amount of literature on this approach, especially in the North American context, the hedonic pricing approach is not easily applicable in many circumstances, such as where property markets do not operate efficiently, or where unemployment rates are high. The latter conditions are commonly found in developing countries, which consequently pose serious problems for the application of hedonic wage methods.

A third direct valuation technique is the travel cost approach (refer Freeman 1993: 443). Developed in the 1960s (see Knetsch 1963) in the context of valuing demand for recreational activities in North America, this approach values environmental services or commodities (such as nature reserves) by calculating the total of visitors' expenditure on transportation to and from the reserve, as well as other costs such as entry fees and the opportunity cost of their time devoted to this activity. These valuations are used to derive demand curves for the environmental service, which can be aggregated or extrapolated to determine society's total WTP for the environmental feature.

The travel cost approach, like the hedonic pricing approach has the advantage of using actual rather than hypothetical markets to derive environmental values (unlike the contingent valuation method). However, the travel cost approach is also of limited applicability in practice because it relates to only a few environmental issues where travel and time costs can be observed (for example, visits to nature reserves). Moreover it also has large informational requirements for implementation in practice.

3.3.2 *Indirect valuation methods*

Indirect valuation approaches differ from those described above, insofar as they do not attempt to derive a valuation based on the revealed preferences of individuals (as if a market existed for the environmental commodity in question). Instead, they derive valuations on the basis of opportunity costs or indirect costs which can be attributed to a given impact (Cooper 1981: 51). These costs are usually incurred in already-existing markets, and are related in some way or another to the environmental element of concern, particularly where these concern health effects. Thus, for example, the social cost (value) of a case of pneumonia resulting from a worker's exposure to air pollution in the workplace might be calculated to include as many of the following as apply:

- foregone income of that person while he or she is absent from work due to illness, and of any care-givers who are unable to work whilst attending to that person;
- expenditure on private health care - medicines, visits to doctors and hospitals and other forms of treatment;
- expenditure by public health authorities on treatment of that person;
- transport expenditure by that person and any others in respect of visits to hospitals, doctors and so on;
- any other costs incurred as a result of the illness.

The sum of these items, discounted to a present value, can be used as the value of such an externality. This valuation method is also called the cost-of-illness (COI) approach (Freeman 1993: 343); it has some elements in common with the `averting behaviour' valuation method described by Cropper and Oates (1992: 703). The latter relies on the fact that environmental damage or amenity can be valued with reference to inputs (or services) which are purchased to mitigate the negative effects. In the example given by these authors, the value of additional air pollution can be measured by the value of additional expenditure on medication to relieve itchy eyes and runny noses (ibid). In effect, this method assumes that other inputs can be substituted for environmental quality.

Although most environmental economists tend to prefer revealed preference methods because they reflect individuals' valuations based on their underlying utility functions, the opportunity cost method has important advantages, particularly in a developing country context where WTP valuations are prone to the problems outlined earlier. Moreover, the data requirements of opportunity cost approaches are usually more modest than those of hedonic pricing or CVM techniques, which make them more practicable in a context such as that of South Africa. Opportunity cost measures, such as those based on the costs of illness, are generally viewed as a lower bound estimate of willingness to pay for the change in health status (Cropper & Oates 1992: 715).

The choice of valuation method for purposes of valuing externalities in the South African energy sector is discussed in further detail in Chapter Four, in the specific context of the electricity generation and household consumption sectors. For the present, it can be noted that the opportunity cost method is used: the rationale for this choice is described in more detail in Chapter Four.

3.4 Internalising environmental costs into accounting systems

A final aspect of relevance in the literature on environmental valuation, is environmental accounting. This is an area in which the identification and valuation of externalities is given practical effect. In essence, it is concerned with `getting the values right' for purposes of decision-making.[13]

Environmental accounting can occur, broadly speaking, at two levels. The first is the micro-scale, and essentially involves devising mechanisms or systems to incorporate external environmental costs into cost functions. Usually, taxes or levies of one form or another are applied to bring private production costs into line with true social costs. The theory of this issue was discussed earlier in the chapter; its practical application forms the basis of Chapters Three and Four of this thesis. The only point to be made here is that the internalisation of external costs can sometimes lead to a re-ranking of supply options (for example, coal, nuclear, hydro, demand-side management) and can therefore be highly consequential at a micro-economic level.

The second area in which environmental accounting is the subject of attention, is in relation to the macro-scale and specifically national environment accounting. The concern in this case is also with `getting the prices (values) right'. The motivation for this stems not so much from a need for accuracy and precision *per se*, but from the signals which are provided to planners and other users of national accounts through the incorporation of environmental values in the account.

It is widely accepted, at least in the development economics literature, that standard measures of economic growth such as gross national product (GNP) do not necessarily reflect changes in welfare over time, nor particularly, changes in environmental quality or the stock of natural capital (Adger & Grohs 1994, Pearce et al 1989, Munasinghe 1993a: 11,

[13] Similar logic, but in another context, is applied to electricity utilities in developing countries, for example, by advocates such as the World Bank which typically advise them to `get their prices right' - equal to the long-run marginal costs of supply.

Pezzey 1992: 3, Dasgupta 1990: 60, Pearce & Atkinson 1995, Pearce & Warford 1993: 83). This is because conventional measures of national income do not indicate whether the economic activity they are based upon is sustainable, however that may be defined. Further, many kinds of income or expenditure which make a positive contribution to national income may in fact have resulted in depletion of non-renewable resources or deteriorating environmental quality.

Notably, there is some disagreement in the literature on the adjustments which can legitimately be made to national accounts, particularly over the treatment of defensive expenditures. Proponents of national environmental accounting systems have proposed (inter alia) that defensive expenditures should not be included in final demand - but as an intermediate good (Ahmad et al 1989). Others, however, have noted that defensive expenditure should be treated just as any other item of expenditure since it contributes to improved well-being (Dasgupta & Maler 1994: 325, Bojo et al 1992: 49). These differences are perhaps attributable to the fact that the former view takes as its starting point an environmental state which is clean or `natural', whereas the latter starts with actually prevailing conditions (usually a deteriorating environment).

In recent years, there has been much activity by environmental economists in identifying measurement and accounting systems which more appropriately reflect the impact of economic activity on environmental quality and sustainability. In particular, efforts have been directed at defining and measuring `sustainable development'. Pearce and Atkinson (1995: 167) trace the thinking around sustainable growth from Hartwick (1977), Solow (1986) and Page (1977) to their formulation that sustainability depends on a non-declining capital stock over time (see also Pearce et al 1993: 15). The capital stock can be thought of as comprising three kinds of capital: reproducible (human-made) capital K_r, human capital K_h (the stock of knowledge and skills held by humanity) and natural capital K_n - the stock of renewable and non-renewable environmental resources and services. By acknowledging the role of savings in the creation of capital, and assuming that the stock of human capital cannot depreciate, a condition for sustainability can be derived:

$$S/Y - \delta_r/Y - \delta_n/Y \geq 0$$

where S is savings, Y is income, δ_r and δ_n are the depreciation of reproducible and natural capital respectively (ibid: 168).

This condition is termed the `weak sustainability' rule, since it assumes that substitutability can occur between various components of the capital stock: in this case between reproducible and natural capital. In other words, the natural environment can be degraded in the process of generating wealth, provided some of those rents are reinvested in capital which will maintain human consumption at constant levels over time - effectively the

Hartwick rule (1977). However, the assumption of substitutability between these forms of capital clearly does not always hold, since many environmental losses will be irreversible. Pearce and Atkinson therefore also describe a `strong sustainability' rule, which would require that K_n not be declining, whilst total K is constant or increasing. A second condition defining the state of sustainability is that the rate of technical change should exceed the rate of population increase.

Not considering the latter condition, based on studies of 22 countries for which the sustainability test was done, Pearce and Atkinson found that 8 of the 22 failed the weak sustainability test, while all 22 failed to meet the condition defining a state of strong sustainability (1995: 173). Whilst the authors acknowledge that their analysis is approximate and does not fully consider factors such technical change, the results are nevertheless interesting and suggest further work is required in the area.

Recent developments include adjustments to the United Nations System of National Accounts (SNA) to include items recording the depreciation of natural capital (El Serafy 1994). Deducting depreciation of capital from GNP yields Net National Product (NNP), which is frequently some percentage points lower than the unadjusted figure for GNP. The rationale for these adjustments can be illustrated by means of an extreme-case example. If two economies have equal economic results measured in terms of GDP and GDP growth rates, it would be assumed at first glance, that they are equally well-off. If, however, economy A produces its income from a sustainable eco-tourism industry, whilst economy B produces its income by mining gold and diamonds, it no longer follows that they are necessarily equally well-off. Economy B is producing its income by depleting a non-renewable resource (K_n), and its overall welfare is therefore dependent upon how it uses its income: whether its saving rate and consequent investment is sufficiently high to substitute for the depreciation of its natural wealth.

The effect of the adjustments to the SNA is to make explicit the depreciation of natural capital in the calculation of Net National Product (in the same way as depreciation of reproducible capital is deducted).[14] The significance of these kinds of adjustments is obvious, especially insofar as planners and policy-makers have historically found it difficult to convert rather vague goals of sustainable development into practice, and then to measure whether progress has indeed occurred. Some examples may illustrate this. An analysis by

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[14] By the same token, if an improvement were to occur to the state of natural capital, as an unintended consequence of other activities, this benefit should be offset against any depreciation of natural capital. In practice, however, attention of analysts has been on the cost side of the equation.

Adger and Grohs of Zimbabwe's national accounts, for example, suggested that by ignoring the value of soil erosion and forest depletion, the country's agricultural output was overstated by some 10% in 1989 (1994: 1). Another analysis of Zimbabwe's soil erosion problem, sponsored by the UN's Food and Agriculture Organisation (FAO) found that the cost of soil erosion could exceed 16% of agricultural GDP, equivalent to 3% of the country's total GDP (Norse & Saigal 1993: 230).

In another study, conducted in Mexico in 1990/91 and supported partly by the United Nations and World Bank, two sets of environmentally-adjusted net domestic product estimates for the country were produced (Van Tongeren et al 1993: 257). The first estimate deducted from net domestic product the environmental costs related to deforestation and the depletion of oil, timber and ground water resources, and yielded a national income amounting to 94% of traditional NDP. The second estimate deducted, in addition to the above, the costs of environmental degradation through air and water pollution, waste disposal and soil erosion. The adjusted national income thus calculated was equivalent to 87% of conventional NDP. Again, even though the authors of this report emphasised that their objective in the exercise was to highlight conceptual and methodological issues, rather than to quantify adjustments with high levels of precision, the order of magnitude of their results suggests that environmental adjustments to national accounts may produce dramatically different results. This is an ongoing area of applied research in environmental economics, which is dependent to a large extent upon micro-scale or sector-based externality valuation studies, such as those attempted in this thesis. The latter represent the building blocks for macro-scale environmental accounting exercises.

4. International studies of externalities in the energy sector

Having reviewed the literature as it pertains to the theory of externalities environmental valuation, it is important to consider how the theory has been applied to practical externality problems arising from the energy sectors of developing and industrialised countries. At the broader level, there is a notable body of literature which has emerged in the past decade or so concerned with various aspects of energy-environment policy (not all of it specific to externalities *per se*): see for example, Munasinghe (1990, 1991, 1993a, 1993b), Pearson (1989, 1993, 1994), Pearson and Stevens (1992), Fisher and Smith (1982), Pearce (1995a) and Burtraw et al (1995).

Of relevance for present purposes, are studies of energy externalities which have been reported in the literature or which have been documented in the public domain. In reviewing the literature, particular emphasis has been directed towards studies undertaken in developing countries; however, as will become apparent, the bulk of externality studies have been undertaken in industrialised countries. Moreover, there have been many more studies of externalities in the electricity generation sector than in the household energy consumption sector.

4.1 Externality studies in the electricity generation sector

There have been several major initiatives in the past decade to account for external costs arising in the power sectors of industrialised countries. In addition to these major externality studies, there have also been numerous smaller studies in which external costs have been estimated for specific classes of environmental impacts. Many of these micro studies have been utilised in the valuation components of the larger externality studies. This section of the report reviews the major externality projects, taking account of the smaller scale studies only to the extent that their results have been incorporated into the former.

Externalities studies employ widely differing methodologies, which account in large measure for the different numerical valuations they yield (Lee 1995:2). Methodological approaches can be usefully categorised as being either `top-down' or `bottom-up' (ETSU 1995:6). Top-down approaches generally use aggregated data, for example, of national air pollution emissions or health impacts, to which costs are ascribed, and which are then apportioned to the energy sector. The result is an estimate of average costs (as opposed to marginal costs of, say, new generation facilities). Advantages of this approach are that the data requirements are relatively modest and so results can be produced in a wider range of situations. Its main disadvantages include the lack of attention to site-specific environmental impacts and conditions, and the lack of marginal analysis which is required when making optimal investment decisions.

Bottom-up methodologies generally entail analyses of environmental costs which are more specific to given technologies, sites, demographic and environmental conditions. Data are required from the main stages in the `impact pathway': emissions of pollutants, their dispersion and deposition, the sensitivity of receptors to pollution, dose-response relationships, and finally economic valuation of impacts. This category of approaches therefore attempts to construct some kind of damage function. It allows for marginal analysis of new projects, and has the advantage of being more specific to particular power plants and locations. The most serious disadvantage of the damage function approach is that it is very data-intensive, which imposes severe constraints where data gaps are significant.

In the sections which follow, a number of significant externality studies which have been documented in the literature, are summarised briefly. The reason for presenting them is less to make comparisons between them or with South Africa, than to identify methodological approaches which can inform a South Africa-specific analysis.

4.1.1 *Externality studies in industrialised countries*

4.1.1.1 The Hohmeyer study in Germany, 1988

Hohmeyer (1988) was responsible for one of the first major attempts at valuing environmental costs in the energy sector, using West German data. This study used a `top-down' approach: it utilised other studies' estimates of the country's total damage costs attributable to air pollution, and then apportioned this to various sources of emissions, including the fossil fuel-based power sector and the nuclear power sector. The study was performed under contract for the Commission for European Communities and was undertaken in the context of governments taking corrective action against market barriers to renewable energy technologies. It concluded that accounting for external effects would improve the competitive position of renewable sources of energy.

The valuation estimates produced by the Hohmeyer study are summarised at the end of this section, but it is worth noting that they were highly significant: of the same order of magnitude as private electricity production costs. As a result, the Hohmeyer study received considerable amounts of attention, particularly in Germany, and was heavily criticised on the grounds of its methodology and results (Friedrich & Voss 1993:114).

In subsequent work, Hohmeyer has extended the scope of the externalities for which valuations have been calculated, and has produced valuations for the emissions of greenhouse gases (Hohmeyer & Gartner 1992). The latter estimates were very significant in relation to other external costs.

4.1.1.2 The Pace University study in the United States, 1991

Ottinger et al (1991) were contracted by the United States Department of Energy and the New York State regulatory authority to review the international literature on the methodologies used to assign monetary costs to environmental externalities, and to present the results of those studies (1991:13). The intention of the study was to assist utilities, regulatory bodies, legislators, policy analysts and public interest groups in estimating the environmental costs of electricity supply, in the context of integrated resource planning decisions. The Pace study utilised a bottom-up, damage function approach, drawing on numerous studies which had addressed specific aspects of the damage pathway; it undertook no primary data collection of its own.

The environmental costs produced by Ottinger et al were derived from a spreadsheet-based model which included data on pollution emissions, dispersion, dose-response relationships, and monetary valuation of impacts. Default values for all of these variables were selected on the basis of the existing body of literature.

The result of their efforts was an extremely comprehensive report, over 750 pages in length, with estimates of environmental costs of the major impacts for each of seven resource options: coal, oil, natural gas, nuclear, renewable technologies (hydro, solar, wind and biomass facilities), waste-to-energy facilities and demand-side management options. The most important of these results are summarised at the end of this section.

4.1.1.3 The Tellus Institute study in Wisconsin, United States, 1991

Bernow et al (1991) performed a study of the environmental costs of a group of pollutants associated with electricity utilities in Wisconsin state. The context was one of resource planning for utility investment, and estimates of environmental costs were produced for air pollution emissions from several coal-based generation technologies.

This study differed from those described thus far insofar as its methodology was not based on the damage function approach, but on the regulators' revealed preference approach. The motivation for this was that information requirements for the damage cost approach were excessive, and instead, the cost of meeting targets for environmental quality as prescribed by regulatory agencies was taken to reflect society's willingness to pay to avoid the risk of those damages (ibid: 21). The marginal control costs were thus used as a surrogate for marginal damage costs.

Whilst the regulators' revealed preference approach avoided the information gathering problems faced by damage function studies, it has been criticised for making the assumption that regulators know what marginal damage costs are and that they make the optimal regulatory decision (ETSU 1995:7). This logic is argued to be self-referencing, and would automatically equate marginal abatement costs with marginal abatement benefits, since the latter are equal to avoided environmental costs, which in turn are valued at the cost of achieving the regulator's target level of pollution.

4.1.1.4 The Pearce et al study in the United Kingdom, 1992

Pearce and colleagues were contracted by the UK Department of Trade and Industry to survey the social costs of energy production and use (Pearce et al 1992). This study drew on Ottinger et al (1991) and other literature, which was used as the basis for making its own estimates of external costs. The approach used was similar to that of Ottinger et al, although it adopted a fuel cycle approach, which meant therefore that it addressed a more complete picture (ETSU 1995:7).

As with the previous studies, Pearce and colleagues did not collect any primary site-specific data and therefore the externality valuations produced are prone to uncertainty and inaccuracy when considering specific sites. More recently, Pearce revised the earlier estimates (1995b).

4.1.1.5 The Lockwood study in the United Kingdom, 1992

The Parliamentary Office of Science and Technology in the UK commissioned a study undertaken by Lockwood (1992), which was a review, synthesis and extension of the literature on external costs, giving specific attention to the UK. The main sources of literature which were reviewed were the studies of Hohmeyer and Ottinger et al. Their methodologies were assessed as being `broadly correct' and their results in line with those in the literature at the time, although they were criticised on two grounds: their estimates of dose-response relationships (for example, in the case of the impacts of acidic deposition) were not easily generalisable since these are highly site-specific in practice, and secondly, the wide range of estimates attributable to nuclear power was due to extreme uncertainty over the probability of catastrophic accidents. The report did not offer any revised estimates of social costs.

An important element in this study was the consideration given to `fiscal externalities', in addition to the usual environmental externalities. Fiscal externalities include (primarily) taxes and subsidies which often affect different fuel cycles to varying degrees. The report concluded that coal-based electricity in the UK is taxed by up to 0.95p/kWh (about 5 c/kWh), while nuclear-generated electricity is effectively subsidised by about 0.15 to 0.45 p/kWh (or 0.8 to 2.5 c/kWh) (ibid: i). The report also concluded that there was some economic justification, on the basis of external cost estimates, for the incremental installation of flue-gas desulphurisation equipment on coal-fired plant.

4.1.1.6 The Friedrich & Voss study in Germany, 1993

The attention which the 1988 Hohmeyer study brought to the issue of externalities precipitated many other studies elsewhere, but also prompted other analysts in Germany to analyse the issue. In one such case, Friedrich and Voss scrutinised the methods and data used in the Hohmeyer study, found them to be `unsuitable' and the corresponding estimates of external costs `too high' (1993:114). As a result, they did their own analysis of Germany's external costs in various fuel cycles, with a view also to establishing whether the relative ranking of resource options would change.

Their conclusions were that while the internalisation of externalities could avoid or reduce misallocations of scare economic resources, it would not necessarily lead to changes in the competitive rankings of coal, nuclear, wind, solar and so on (ibid: 122).

4.1.1.7 The US-EC external costs of fuel cycles project, 1991-1993

Following the earlier studies described above, a major project was initiated jointly in 1991 between the US Department of Energy and the European Commission, with the aim of developing an accounting methodology to systematically evaluate external costs of various fuel cycles. The study used a damage function approach and provided an accounting framework within which external costs could be operationalised.

A number of research groups from both sides of the Atlantic Ocean contributed to this project, which developed a conceptual and accounting framework for assessing the external costs of various fuel cycles. The first phase of the project was completed in June 1993, and thereafter work continued to be done in the US and Europe, although more independently, since this involved the application of the methodologies in particular contexts. These projects are described separately below.

4.1.1.8 The ExternE study for the European Commission, 1993 and ongoing

The European Commission supported subsequent phases of the joint US-EC study described above, in the so-called `ExternE project', covering the period from July 1993 to 1998 (ETSU 1995:11). The main components of ExternE included the following:

- The methodology which had been developed in the first phase for the coal and nuclear cycles was extended to several other cycles: oil, gas, lignite, hydro, biomass and wind.
- The methodologies were implemented in a number of European states, including the UK, Germany, France, Netherlands, Greece, Portugal, Italy, Spain and Norway (ibid: 12).
- Software tools for application of the methodology were developed with a view to allowing application of the approach in local circumstances, including facilities to perform sensitivity analyses.

Through the process of implementation of the methodology and the development of software tools, the approach has been reviewed and improved.

The principal focus in ExternE has been on the development of a methodology, rather than the calculation of values of external costs. The aim of the methodology is to allow for the calculation of marginal external costs and benefits for a specific power plant, at a specific site and using specified technologies. It adopts a cradle-to-grave view of environmental impacts and thus approximates a life cycle analysis in cases where upstream impacts (such as coal mining) are included.

4.1.1.9 The US Department of Energy study, 1993-1995

The US portion of the joint US-EC study was led by the Oak Ridge National Laboratory (ORNL) and Resources for the Future (RFF) and was similar in many respects to the ExternE project. A similar but separate series of publications were produced, covering the various fuel cycles and methodological questions (ORNL/RFF 1994a-b, 1995a-e). The methods were also implemented for US conditions; the results of these studies are summarised later in this section.

As with the ExternE project, the results of applying the methodology have been criticised by Ottinger (1995) on the grounds that the valuations of external effects were too conservative and failed to adequately account for all external costs. He argued that the values produced would render externalities irrelevant to decision-making, with potentially serious environmental and health consequences.

4.1.1.10 The RCG-Tellus study in New York state, 1995

This study, called the New York State Environmental Externalities Cost Study, also utilised the damage cost approach and entailed the development of a user-friendly computer model (called EXMOD) which could be used to calculate external costs for specific electricity resource options (Rowe et al 1995). The study was supported by a consortium of regulatory and research organisations in New York state, in response to an order by the New York Public Service Commission to develop a methodology to estimate external environmental damages for new and relicensed supply and demand-side management options in the state (Lee 1995:4).

The study therefore comprised two main components, the first being the development of a methodological tool and the second being its application to the circumstances prevailing in New York. The results of the valuation exercise were also criticised by Ottinger (1995) on the grounds that the study undervalued or neglected several important categories of external costs.[15]

4.1.1.11 The Schleisner et al study in Denmark, 1995

A collaborative project was carried out by various institutions in Denmark in order to assess the damage costs of energy production in the country, based on coal-fired, wind, biomass and natural gas power generation facilities. The study used a damage cost approach and relied partly on data collected in the ExternE project described earlier. It included in its scope estimates of damage costs resulting from climate change.

External cost estimates for the coal and wind options were reported by Schleisner et al (1995) and are summarised shortly.

4.1.2 Developing country studies of external costs

All of the studies described thus far have been undertaken in highly industrialised countries which have their own set of dynamics around environmental and development objectives. Much less work has been undertaken on externalities in developing countries, partly reflecting the relatively lower priority accorded to environmental goals, and partly reflecting

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[15] Notably, the criticisms were not directed at the EXMOD model, which was singled out as being an `excellent model'. This model is used later in this thesis, and is described briefly in Appendix 3.

the smaller role of the electricity generation sector in their economies. This section reviews two investigations of environmental costs which have been undertaken in developing countries.

4.1.2.1 The Carnevali and Suarez study in Argentina, 1993

Carnevali and Suarez undertook an economic assessment of the effect of the Argentinean government's energy policies in terms of their environmental impacts (1993:68). These policies in the 1970s and 1980s encouraged a two-pronged substitution: firstly, hydroelectricity and nuclear energy for conventional thermal energy (mainly coal and oil), and secondly, natural gas for coal and oil.

This study, whilst not attempting to place a value on external costs in cents per kWh, did quantify the avoided air pollution emissions as well as the avoided environmental control costs which flowed from the government's substitution policies. It was estimated that the switch to cleaner fuel options, although not motivated by environmental concerns, avoided capital expenditure of about $1 580 million by 1985. This highlighted the need to include environmental considerations in energy investment decisions which are often concerned only with microeconomic criteria (ibid: 72).

4.1.2.2 The Dutkiewicz and de Villiers study in South Africa, 1993

The only investigation to-date of external costs in South Africa was undertaken by Dutkiewicz and de Villiers (1993) under contract to the former National Energy Council (NEC), which was subsequently disbanded and absorbed into the Department of Mineral and Energy Affairs. The objective of the report was to quantify those external costs for which sufficient data existed, for four generation options - coal, nuclear, wind and solar - with a view to assessing whether internalisation of these costs would lead to a re-ranking of the competitiveness of those options (ibid: ii). The objective of the study was therefore similar to that of Hohmeyer (1988).

The methodology used was a damage cost approach, relying mainly on a review of the international literature on external costs (notably Hohmeyer and Ottinger et al), supplemented by local data where this was available. The reference year was 1989.

The study noted that there were numerous data gaps, which did not permit the quantification of external costs, although it was suggested that most of these were probably not significant (ibid: 14, 40). The cost estimates produced by the study were at the low end of the range of international studies; these are summarised in the following section. The cost estimates, while showing large relative differences between the different fuel cycles, were too small in absolute terms to make any difference to the competitiveness of renewables compared to coal or nuclear (ibid: 41). In view of the uncertainties and omissions, a fuller investigation was considered necessary.

4.1.3 A summary of international externality values in the electricity generation sector

Distinguishing features of the environmental costs produced by the externality studies undertaken to-date are the numerical discrepancies and lack of consistency when their results are compared. This uncertainty is sometimes used by sceptics to discount altogether the validity of such attempts. This, however, is an unconstructive and uncritical response, since many of the differences can be accounted for by the varying technical and environmental conditions pertaining to the studies, as well as to methodological differences. It is instructive to summarise the results of valuation studies, both as a point of reference for South African purposes, and in order to highlight the key factors which have accounted for their different results.

The estimates of external environmental costs produced by the major studies reviewed above are summarised in Table 2.1. Of those described above, Bernow et al (1991) is not included in the table, as its scope was limited to coal-fired combustion, and it used the regulators revealed preference methodology. Its estimates of external costs ranged from 6.4 c/kWh (converted from 1990 dollars) for combined cycle gas turbines, to 16.91 c/kWh for atmospheric fluidized bed combustion technologies (1991:86). The joint US-EC project is also not included since it focused on the development of methodologies, and valuation was subsequently done in the ORNL/RFF and ExternE projects.

For purposes of this study, the most important fuel cycles are coal, and secondarily, nuclear; nonetheless, it is interesting to compare the external costs of these cycles with others, which becomes especially relevant when making decisions about new investments with various resource options. As would be expected, renewable options fare much better than fossil fuel and nuclear fuel cycles, with lower external costs. Of all fuel cycles, coal had the consistently highest external cost, with the exception of nuclear in Hohmeyer's study and oil in Pearce's study.

It is evident from the table that there are wide variances in the numbers produced by the studies. These differences are due at least as much to different methodologies, as they are to different physical, technical and environmental conditions prevailing in the respective cases. Although most studies attempted to quantify impacts arising at various stages of the fuel cycle, some of the differences are a result of the fact that the scope of analysis differed in the studies, so that their results are not strictly comparable.

Table 2.1 Fuel cycle external costs (in c/kWh) estimated by various studies[1]

(adapted from Lee 1995:5; Friedrich & Voss 1993:121, Dutkiewicz & de Villiers 1993:41, Schleisner et al 1995)

Study	Fuel cycles						
	Coal	Nuclear	Gas	Oil	Hydro	Solar	Wind
Hohmeyer (1988)	14.49 to 33.05	36.45 to 77.96	14.49 to 33.05	14.49 to 33.05	no estimate	+25.51 to +64.05[2]	+21.01 to +46.12[2]
Ottinger et al (1991)	24.67	12.33	5.12	11.49 to 28.51	no estimate	0 to 1.70	0 to 0.42
Pearce (1992)	7.25 to 30.71[3]	0.31 to 1.84[4]	2.33	34.11	0.25	0.43	0.25
Friedrich & Voss (1993)[5]	0.87 to 4.65	0.06 to 1.35	no estimate	no estimate	no estimate	0.16 to 2.69	0.08 to 0.83
Dutkiewicz & de Villiers (1993)[6]	0.93	0.26 to 0.80	no estimate	no estimate	no estimate	0.01 to 0.05	0.01 to 0.04
ORNL/RFF (1994a,b; 1995a-e)	0.21 0.47[7]	0.08 0.12	0.00 0.08	0.06 0.08	0 0.06[8]	no estimate	no estimate
ExternE (ETSU 1995)	3.24 7.94[9]	0.05 1.31[10]	0.38	6.26	1.20[11]	no estimate	0.57 to 1.20[12]
RCG/Tellus (Rowe et al 1995)	1.01[13]	0.04[14]	0.08	0.54	no estimate	no estimate	0.00
Schleisner et al (1995)	0.73 to 9.52[15]	no estimate	no estimate	no estimate	no estimate	no estimate	0.07 to 0.73

Notes:

[1] All amounts in 1994 SA cents/kWh, converted from 1994 US cents at a rate of R3.66/$1, and from 1993 German Marks at R1.98/DM1; where relevant, amounts in Rands have been adjusted using a 10% annual inflation factor.
[2] + denotes an external benefit.
[3] Estimates for a `new' and `old' plant respectively.
[4] High estimate reflects risk averse valuation of health impacts of nuclear disaster.
[5] Estimates are for 4000 full load hours per year, and include external costs of back-up system.
[6] Estimates originally in 1989 Rands, annual adjustment used of 10%.
[7] Estimates for plants in the rural south-west and south-east US respectively.
[8] Estimates for retrofit on existing dams in Kentucky, and diversion project in Washington state, respectively.
[9] All but 0.02 c/kWh was due to the aesthetic value of a waterfall, which was estimated in a contingent valuation study.
[10] Estimates using a 3% and 0% discount rate respectively.
[11] The first estimate was for a site at West Burton, UK; the second was for Laufen, Germany.
[12] Estimates for various sites in the UK.
[13] Estimates are for a rural site in New York state.
[14] For a boiling water reactor, rather than pressurised water reactor.
[15] Estimates are for a `conventional coal-fired plant' defined as a 350 MW plant with desulphurisation and de-NOx equipment.

Closer inspection of externality values in the coal cycle reveals that the studies can be grouped into two categories, based on the order of magnitude of their results. The first group comprises the three earlier studies - Hohmeyer, Ottinger and Pearce - which produced external cost estimates in the range of 7.3 c/kWh to 33.1 c/kWh. The second group of studies, which were all conducted post-1992, produced lower estimates, in the range of 0.21 c/kWh to 9.52 c/kWh. In analysing the differences in these studies, the following main factors may be identified (partly drawn from Lee 1995:2-3):

- *Methodology:* While all the studies summarised in the table used the damage cost approach, they used different methods to obtain data on components in the impact pathway. The earlier studies generally used other studies' estimates of pollution emissions and impacts, and multiplied these by economic values to calculate damage costs. In most earlier cases, these economic values were based simply on control costs for the relevant pollutants and were thus not consistent with conventional economic valuation approaches. The second group of studies, on the other hand, either used more complex and specific methods to collect data on pollution emissions, dispersion and impacts, such as atmospheric models and dose-response functions, and they used (lower) valuations derived from theoretically-accepted valuation techniques. In practice, the effect has been that the earlier studies used higher estimates of emissions, concentrations and impacts.

- *Emission factors:* The earlier studies' emissions factors (measured in tons of pollutant per GWh of electricity generated) were considerably higher than in recent studies, sometimes by a factor of ten. This is partly due to technical differences in the plants which were addressed, in that more recent studies have selected newer plants which have better environmental performance in general, and in some specific cases, with desulphurisation and other control equipment.

- *Sulphate and nitrate aerosols:* The older studies contain different assumptions about emissions of SO_2 (from which damaging sulphate aerosols are formed) and about their dispersion in the atmosphere, which lead to higher external cost valuations. SO_2-related externalities accounted for 60% of total externalities in Ottinger et al's work, and 75% in Hohmeyer's study, both of which are considerably higher than the more recent studies.

- *Climate change damages:* in earlier studies, fairly high values were attributed to damages caused by climate change (impacts include, for example, sea level rise, increased drought and climatic extremes), whereas in more recent studies, analysts have argued that there is too much scientific uncertainty about the impacts of climate change (without questioning its likelihood of occurring) to make meaningful estimates of damage costs.

> Recent studies have, in turn, been criticised for under-stating the likely scale of those impacts by avoiding their valuation (Ottinger 1995:4).

If any conclusions are to be drawn from these international studies, then the first would be that external costs can be significant in absolute terms, as well as in relative terms when comparing alternative fuel cycles. International experience therefore supports the investigation of externalities in South Africa's power sector. A second observation would be that it is important not to take external costs simply at face value, but that they need to be evaluated in the specific contexts in which they were calculated. Thus it is important that the values summarised above are not simply applied uncritically to the South African situation, but that local circumstances be taken into account as far as possible. Finally, it is important that assumptions and methodologies are made explicit in order that results of the valuation exercise can be appraised in the appropriate context. Equally, it is important that all limitations in the valuation exercise be made apparent.

4.2 Externality studies in the household consumption sector

Whereas there is a large and growing body of literature describing externalities arising from electricity generation, there has been much less experience to-date with externalities caused by household energy consumption. Moreover, most of the studies which have sought to value externalities at this level, have been undertaken in North American or European countries. Nonetheless, there are some studies which have relevance for the task in this thesis: these are summarised below. Because of the difficulties in comparing the monetary results of these valuation studies, this review is more relevant from a methodological perspective than from the point of view of the monetary valuations derived.

4.2.1 US EPA studies in California, 1979

The United States Environmental Protection Agency (EPA) supported a range of studies in the late 1970s addressing air pollution and its economic impacts in two parts of California: the South Coast basin (including Los Angeles) and the San Francisco area (Kneese 1984: 57). Although the studies were not specifically linked to air pollution emitted by energy sources, they are relevant for present purposes because of their methodological focus. The studies were designed partly to test and develop valuation methodologies, and accordingly, the value of clean air was estimated using two methods: survey-based studies of individual willingness-to-pay, and second, hedonic pricing studies of property values.

The results of using these two methods were similar in both geographical areas: valuations were of a similar order of magnitude, with somewhat higher values derived from the hedonic pricing approach. It was concluded from the study that survey instruments may provide a `reasonable way' of obtaining environmental quality benefit estimates. As a cautionary note, however, it was pointed out that the results of the study were not necessarily transferable to other situations, such as where the air quality situation is not as well-defined, either through efficiently-functioning property markets, or in the personal experience and knowledge of resident individuals.

4.2.2 *The Haifa, Israel study of health effects of air quality, 1987*

Shecter et al (1987) attempted to produce valuations of human health effects of air pollution, using two methodologies.[16] Both approaches utilised primary data collection surveys - some 4 000 interviews - from which valuations were derived. The first approach attempted to develop an indirect utility function, by calculating household-level demand functions for improved environmental quality, from observed changes in demand for housing and medical care. The second method utilised more direct contingent valuation techniques to determine households' preferences for improved air quality.

The results of the two methods were of a similar order of magnitude, with higher values yielded by the survey-based method. This was as expected, because the indirect method assumed (for simplicity) that the effects of air pollution were captured by changed demand for housing and medical care (Bojo et al 1992: 172). Clearly, the more consumption goods and services which are included in the indirect approach, the closer its valuation results should be to the contingent valuation results.

4.2.3 *Nepal Forest Development Project, 1983*

In this study, the benefits and costs of a forest development programme in the Phewa Tal region of Nepal were evaluated (Fleming 1983).[17] The aims of the project were to increase the productivity of different land uses and to provide a sustainable flow of fuelwood and fodder. For this purpose, the benefits of interventions had to be quantified: these included afforestation investments, improved management of existing resources and agricultural projects.

Three benchmarks were used to value the firewood produced by the programme, all based on the opportunity cost approach: first, wood prices from existing markets in the region; second, the value of alternative uses of the closest substitute (dung), and third, the value of time spent by families collecting fuelwood from local forests. Although valuations were performed using each of the three methods, the lowest valuation was used (the second

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[16] This study is described in Bojo et al (1992: 169). Again, although the air quality impacts were not limited to energy sector emissions, this study is included here for reasons of its methodological interest.

[17] This study is described in Bojo et al (1992: 151).

method). This choice appeared to have been made in order to be prudent, rather than for any theoretical or methodological preferences. The proposed project yielded a benefit-cost ratio of 1.7, even using the lowest valuation of fuelwood.

In reviewing the study, Bojo et al (1992: 155) suggested that the valuation results were surprising: if the opportunity cost of collecting firewood is higher than the cost of the next alternative, it would make more sense for people to use the alternative source of energy rather than collect firewood.

In a more general sense, Pearce (1993: 80) has noted that the most common valuation method has been to use the closest substitute - such as paraffin or dung, and that the opportunity cost of labour time approach is not strictly correct because it does not reflect individuals' actual choices reflecting their willingness to pay.

4.2.4 *Other non-energy externality studies*

In a social cost benefit analysis of a water supply and sanitation programme in Zimbabwe, amongst the benefits which were valued, were the health improvements expected to result from improved water and sanitation services (Fredriksson & Persson 1989).[18] The value of reduced morbidity in the local community was estimated using the cost-of-illness approach. This was calculated with reference to treatment costs in the public and private health sectors, costs of lost production and costs of extra transportation. As noted by Munasinghe (1993a: 42) the resulting valuations may have understated the `true' values - which would ideally have been based on individuals' willingness to pay. Presumably data and resource constraints precluded the use of WTP measures.

With respect to mortality effects in children, the cost benefit analysis valued avoided deaths at their net future output: the difference between the present value of average production over the lifetime of children, and their average future consumption. This was acknowledged to be a lower bound value, which would understate the true value of an avoided death. Overall, the project delivered a net benefit from a societal perspective. Notably, this was the case even though no account was taken of the likely income distribution benefits of the project, since no weighting of valuations was made to account for these distributive effects (ibid: 42).

In another study, in which the costs to the South African economy of cardiovascular illnesses were estimated, the cost of illness valuation approach was also used (Pestana 1993). Medical data were used to estimate the physical incidence of these health effects. The economic costs of these effects were calculated with reference to their opportunity costs: expenditure on

[18] This study is described in Munasinghe (1993a: 41).

public and private health care, lost productivity due to absence from work, transportation expenses, ambulance costs and costs of medication. In the case of mortality, the human capital approach was used - in the absence of other valuation estimates for South Africa, and acknowledging the theoretical problems with this approach. The order of magnitude valuation produced by this study was highly significant in absolute terms.

In addition to the above studies, numerous environmental valuation studies have been undertaken in developing countries; some examples of these include the following (Munasinghe 1993a: 40-53, Pearce 1993):

- a cost-benefit analysis of land improvement in Lesotho, which was based on changes in productivity (from Bojo 1991);
- valuation of land-use in an Amazonian rain forest, using the same method (from Peters et al 1989);
- valuation of the consumer surplus from visits to a Costa Rican rain forest, using travel costs (from Tobias & Mendelsohn 1991);
- valuation of viewing elephants on safari in Kenya, using travel costs (from Brown & Henry 1989);
- valuation of water services in Haiti, using contingent valuation methods (from Whittington et al 1990);
- valuation of improved sanitation in Ghana, using contingent valuation methods (from Whittington et al 1992);
- valuation of a Thai national park, using travel cost and contingent valuation methods (from Grandstaff & Dixon 1986);
- valuation of biophysical resources in a Madagascar national park, using travel cost, opportunity cost and contingent valuation methods (from Kramer et al 1992);
- valuation of soil erosion in Mali, using an opportunity cost approach and a replacement cost approach (from Bishop & Allen 1989).

In reviewing several of these studies, Munasinghe (1993a: 62) concluded that the need was less for further development at a theoretical level, than for practical application of existing environmental economics tools in developing countries. This general view is shared by others such as Pearce (1993), Bojo et al (1992), Convery (1995).

4.2.5 Conclusions

Whilst there have been numerous international studies to-date of the value of environmental quality, including air quality, there have been relatively few studies of energy-related externalities *per se* at the household level. Several conclusions and lessons may be drawn from these studies. First, the relative scarcity of externality studies at the household energy scale is surprising, given that high levels of pollution and resource degradation have been well-documented in the international literature (for example, Smith 1987). Secondly, most studies which have been made, and which are documented in the literature, have been undertaken in industrialised countries, as opposed to developing countries. Thus the methods which have been used, have been developed in response to prevailing conditions *vis-à-vis* market formations, information and data availability, and individual awareness of environmental problems. It is evident from the literature that there have been relatively few applied studies in developing countries.

Thirdly, where valuation studies have tested various methodologies, the preferred approach has generally been to attempt to establish individual preferences for environmental conditions, using contingent valuation methods, or other market-based methods such as those based on property prices or wages. Where these techniques have not been available - usually for reasons of data constraints - alternative methods, such as those based on opportunity costs, have been used. Generally, the results of the latter have been satisfactory, although possibly at the lower range of results which may have been yielded through individual preference methods.

Finally, it is clear from the review in the last section of this chapter that the focus of this thesis, although not new from a theoretical perspective, is unusual in the context of practical externality studies in developing countries. It is to the valuation of externalities in the South African energy sector that this thesis now turns.

Chapter Three

Externalities in electricity generation

1. An overview of South Africa's electricity supply industry

The electricity sector plays a central role in South Africa's economy as the supplier of a key input into the industrial, mining and commercial sectors, as an employer and as a service provider for households. This role is likely to increase in the foreseeable future as the country's reconstruction and development objectives translate into greater economic output and improved service levels for previously-unelectrified households. At the same time, uncertainty exists about the extent to which South Africa's electricity sector is responsible for adverse environmental impacts which are unaccounted or under-accounted for in the current regulatory and pricing regimes. This chapter reviews the main externalities in the electricity sector, in order that they can be quantified in economic terms in Chapter Four.

South Africa's electricity supply industry (ESI) comprises three main sub-sectors, corresponding to their functional activities: the generation of electricity, its transmission from power stations through a high-voltage national network and finally the distribution of electricity from the transmission network to end-consumers. Whilst the focus of this chapter is on externalities in the generation sector, it is worth briefly describing the transmission and distribution sectors.

The transmission sector is owned and operated almost exclusively by Eskom, with the exception of a small amount of high-voltage transmission which is undertaken by distributors.[1] At the end of 1994, Eskom's main transmission network, operating at a voltage of 132 kV and above, consisted of a total of about 25 000 kilometres of lines, and including its low and medium voltage lines, its total network consisted of about 239 000 kilometres of lines (Eskom 1995a: 57). Since Eskom operates as a vertically integrated utility, with no separate public reporting for its divisions, there is little public information about the financial flows associated with the transmission sector.

The distribution sector of the ESI is highly fragmented and is currently undergoing fairly significant re-organisation. Until the early-1990s, the distribution industry was dominated by local authority distributors which purchased electricity in bulk from Eskom and then sold it to their commercial, industrial and domestic consumers. Although Eskom has always had many industrial and domestic consumers to whom it supplied electricity directly, it was not

[1] Until 1987, Eskom was known as ESCOM, or the Electricity Supply Commission.

until 1991 that the annual growth rate in its number of direct customers exceeded 10%. Table 3.1 shows the increase in the number of Eskom's direct customers over the period 1970 to 1994 for three categories of consumers: domestic (mainly households), non-domestic, and bulk consumers. `Bulk' consumers comprise mainly local authority electricity distributors.

Table 3.1 Number of Eskom's customers from 1970 to 1994
(Eskom 1994: 23, 1995a: 58).

	Domestic	Non-domestic	Bulk	Total
1970	98 155	16 630	265	115 050
1980	109 558	69 865	419	179 842
1990	111 709	129 872	673	242 254
1991	142 759	134 278	704	277 741
1992	397 562	143 284	718	541 564
1993	715 219	156 241	742	872 202
1994	1 053 725	152 624	704	1 207 053

It is evident from the table that the number of Eskom's customers grew considerably from 1992 onwards, and this reflects two important structural changes in the ESI during this period:

- Firstly, Eskom launched its electrification programme at the end of 1990 and connected an additional 685 000 homes from 1991 to 1994 (Davis 1996). Its stated goal is to electrify 1.75 million homes during the period 1994 to 1999, or 70% of the target contained in the Reconstruction and Development Programme - 2.5 million homes (ANC 1994: 33).
- Secondly, Eskom transferred large numbers of customers onto its accounts following the collapse of service provision in many historically-black local authorities and in some of the former homeland areas.

Thus the effect of these two factors is that Eskom's focus has shifted away from its traditional concern with bulk generation and transmission of electricity, with only a relatively small distribution function, to include also a strong focus on the distribution and marketing of electricity to end-consumers.

At the same time, significant changes have been taking place among local authority (LA) electricity distributors. Historically, this sector has been highly mainly fragmented along racial lines, with over 400 LA distributors around 1990 (Dingley 1992: 21). Recent trends include the consolidation of the jurisdictions of formerly racially-separate LAs and the re-issuing of licences to these consolidated authorities. Through this process, coupled with the take-over of supply areas by Eskom, the number of distributors has decreased since the early 1990s.

A key stakeholder in the ESI, especially with respect to the electrification programme, is the National Electricity Regulator (NER) which was established in early-1995 on the recommendation of the National Electrification Forum.[2] Whilst the principal concern of the NER is presently with matters related to the electrification programme, such as the issuing of licences to distributors and overseeing the rationalisation of the distribution industry, its role is unlikely to be permanently limited to electrification issues. In fact, in many countries with large electricity supply industries, electricity regulatory agencies have taken on a critical role in a range of governance issues, including the environmental performance of the industry. It might be expected, therefore, that the NER will widen its scope to include such issues at some point in the future once the more urgent priorities around restructuring of the ESI have been resolved.

Turning to the electricity generation sector, the key player is Eskom. Established in terms of the Electricity Act of 1922, Eskom (then ESCOM) is a self-financing public corporation run along commercial principles. Now governed by the Eskom Act of 1987 and the Electricity Act of 1987, it has an effective (but not statutory) monopoly over electricity generation in South Africa. In 1994, Eskom generated 96% of all electricity in South Africa, with the balance being produced by some local authorities with their own power stations (such as Johannesburg, Port Elizabeth and Cape Town) and by private concerns producing electricity for their own consumption (Eskom 1995a: 54). Most of the municipal power stations are used as a backup to the Eskom supply, and have not been fully operational for some time, with the exception of those in Johannesburg, which have ready access to cheap coal (Steyn

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[2] This was a body representing all key stakeholders in the electricity industry, established in 1993 to formulate policies for an accelerated electrification programme for South Africa. It was disbanded in early 1995, having reached agreement on some issues, but since it could make decisions only by consensus, it was unable to resolve several important policy questions.

1994: 7).[3] Given the dominance of Eskom in the electricity generation sector, therefore, the remainder of this chapter focuses on its own power stations. This is not to say, however, that the environmental impacts of local authorities' power stations are less significant than those of Eskom's power stations in relative terms - if anything, the opposite is true given that LA stations are generally older, less efficient and located in more densely populated areas - but their size in relation to Eskom's capacity means they do not warrant further attention in this thesis.

As at the end of 1994, Eskom had a total of 19 power stations in commission, with a total nominal capacity of 37 840 Megawatts (MW) (Eskom 1995a: 56). A breakdown of this capacity by fuel source is shown in Table 3.2.

Table 3.2 Breakdown of Eskom's generation capacity as at 31 December 1994 *(Eskom 1995a: 56).*

	Number of stations	Location	Nominal capacity (MW)	% of total
Coal-fired	12	Mpumalanga (10), Free State, Northern Province	33 568	88.7
Nuclear	1	Western Cape	1 930	5.1
Gas turbine	2	Western Cape, Eastern Cape	342	0.9
Hydroelectric	2	Free State	600	1.6
Pumped storage	2	KwaZulu-Natal, Western Cape	1 400	3.7
Total	19		37 840	100.0

It is evident from the table that 95% of total electricity capacity is based on non-renewable resources: primarily coal, but including also nuclear and small amounts of gas. The coal and nuclear power stations provide the bulk of the base electricity load, while the pumped storage schemes and gas turbines are used primarily to meet electricity demand at the peak and in cases of emergency. Pumped storage schemes are net consumers of electricity, which is used during off-peak hours to pump water up to storage reservoirs, and then allowed to run down during peak hours when electricity demand is highest.

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[3] The Cape Town electricity department recommissioned its Athlone Power station in 1995 for the purpose of meeting demand during peak periods.

Of the total capacity of 37 840 MW, some 4 531 MW or 12% was mothballed as at the end of 1994 (ibid: 56), because of the excess capacity on Eskom system - these were generally the older and less efficient stations. Peak demand in 1994, which occurred on the night of 26 July, was 24 798 MW (ibid: 2), reflecting the large amount of excess capacity, even with the standard reserve margins commonly employed by utilities for safety reasons. In 1995, however, the peak demand on the system came much closer to supply capacity.

Eskom operated nine of its coal power stations during 1994; their location and that of Koeberg nuclear power station is shown in Figure 3.1.

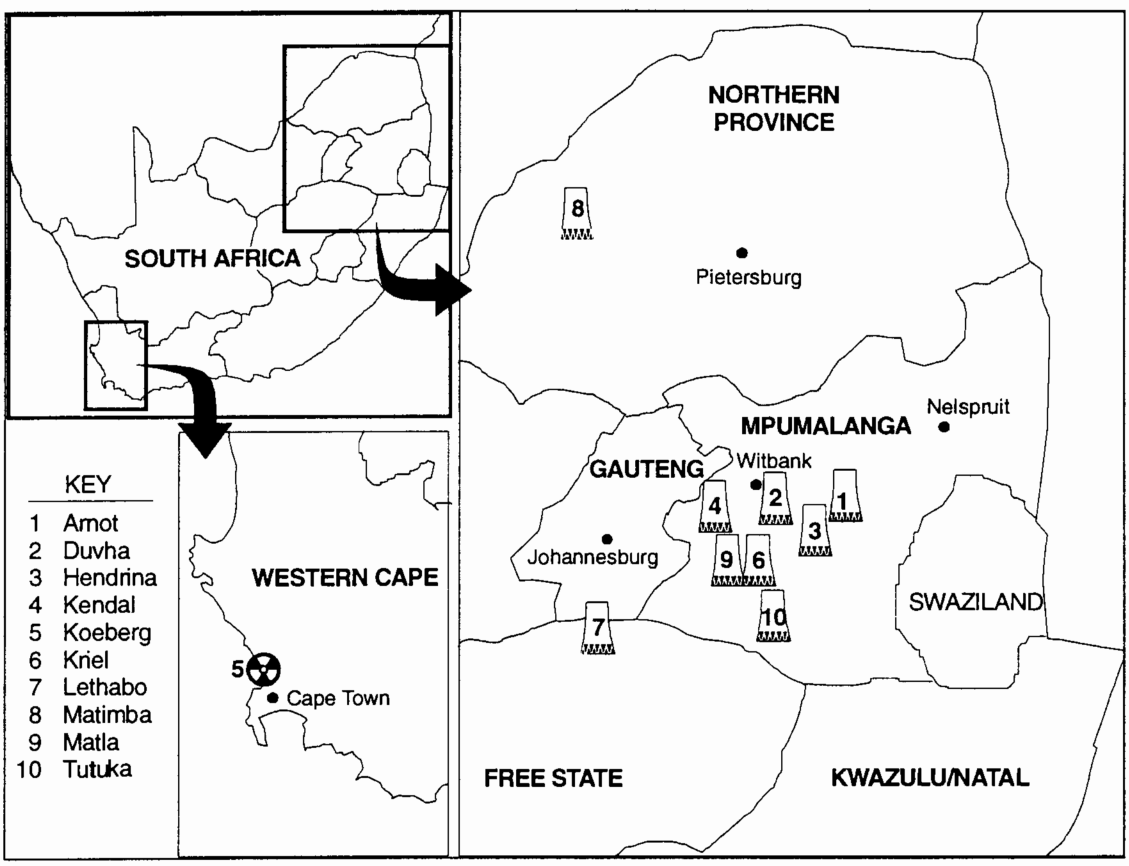

Figure 3.1 Location of Eskom's operational coal and nuclear power stations, 1994

The dominance of coal in the actual production of electricity is even more significant. Table 3.3 shows the total number of units of electricity generated (in Gigawatt-hours) by each of the main supply sources for the period 1991 to 1994.

Comparing the data shown in Tables 3.2 and 3.3, it is evident that coal generated 92% of all electricity in 1994, even though coal-fired power stations constituted a somewhat lower percentage of total capacity (89%) - this reflects the high level of utilisation of coal plants which are used for base-load. The Koeberg nuclear power station has supplied a relatively constant 5% to 6% of Eskom's electricity. Whilst the contribution of the two pumped storage schemes has been relatively constant at around only 1%, the amount of electricity generated by the hydroelectric schemes dropped considerably in 1992 and 1993; this was due to the drought which affected the flow rate of the Orange River.

Table 3.3 Breakdown of Eskom's electricity generated for the period 1991 to 1994
(Eskom 1995a: 54-55).

	1994		1993		1992		1991	
	GWh	%	GWh	%	GWh	%	GWh	%
Coal-fired	148 003	92.3	145 514	94.3	136 830	92.3	135 743	91.3
Nuclear	9 697	6.0	7 255	4.7	9 288	6.3	9 144	6.2
Gas turbine	2	0.0	0	0.0	4	0.0	0	0.0
Hydroelectric	1 074	0.7	146	0.1	752	0.5	1 980	1.3
Pumped storage	1 517	1.0	1 345	0.9	1 333	0.9	1 804	1.2
Total	160 293	100.0	154 260	100.0	148 207	100.0	148 671	100.0

It is clear, therefore, that the most significant component of the generation sector, in terms of electricity output is the coal-based sector. The primary focus of this thesis is therefore on that part of the industry. This is appropriate given that coal-based electricity is generally associated with significant externalities, and that these have been studied widely in other countries. Nuclear power, although contributing less than 10% of South Africa's electricity, has also been analysed fairly extensively internationally on the basis of its actual and potential external costs, and so this chapter will give some consideration to the impacts of the local nuclear industry.

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1.1 Eskom in an international context

Eskom is one of the largest electricity utilities in the world, when compared either on the basis of generation capacity, or on the basis of actual units of electricity produced. Table 3.4 shows the ten largest utilities in the world as reported by Eskom (1995a: 59).[4]

Table 3.4 The ten largest electricity utilities in the world in 1993 *(Eskom 1995a: 59).*[5]

Utility	Country	Annual sales (GWh)	Ranking by sales	Nominal capacity (MW)	Ranking by capacity
EDF	France	372 400	1	98 100	1
TEPCO	Japan	231 665	2	49 492	3
ENEL	Italy	197 451	3	50 888	2
Hydro-Quebec	Canada	152 099	4	29 131	7
Eskom	South Africa	143 800	5	39 746	4
Ontario Hydro	Canada	127 777	6	33 793	6
Korea Electric Power Co	South Korea	127 734	7	27 654	8
Kansai Electric Power Co	Japan	123 300	8	35 035	5
RWE	Germany	121 504	9	25 777	9
TVA	USA	118 560	10	25 622	10

[4] This is not to say that South Africa's electricity industry is one of the largest in the world - in countries such as the USA, for instance, the (much larger) electricity industry comprises a large number of smaller utilities (the biggest being the Tennessee Valley Authority (TVA) shown in the table).

[5] All data as at 31 December 1993, except for TEPCO (31 March 1994), ENEL (31 December 1992), RWE (30 June 1993) and TVA (30 September 1993). The difference between Eskom's sales in this table (143 800 GWh) and its production in Table 3.3 (154 260 GWh) is due mainly to transmission losses and to electricity which is consumed by its own power stations (mostly pumped storage stations).

It is evident from Table 3.4 that Eskom ranks in the top five utilities in the world, when measured on the basis either of capacity or of sales. Interestingly, Eskom is one of only two utilities from non-OECD countries, the other being KEPCO from South Korea; this makes South Africa the country with the lowest per capita GDP of the countries shown in the table (World Bank 1993). Eskom is therefore an unusually strong utility amongst developing countries.

Another important point of comparison between Eskom and international electricity utilities, is in their relative price levels. Here, Eskom ranks as amongst the cheapest producers of electricity in the world.[6] At the end of 1993, its industrial electricity tariffs were the lowest of a basket of industrialised countries, compared using floating exchange rates: Japan (whose average price was over three times higher than Eskom's), Germany, the United Kingdom, the United States, France, Canada, New Zealand and Sweden (Eskom 1995a: 9). This comparison is all the more striking when the resource bases of some of those countries are taken into account: in particular, several of the listed countries are predominantly hydro-electricity based - this is generally regarded as one of the cheapest sources of electricity (when compared to coal, gas, nuclear, etc.).[7] This comparison therefore begs the question: to what extent is South Africa's electricity relatively cheap because externalities are under-accounted for?

1.2 Trends in Eskom's electricity prices

A notable development in relation to Eskom's electricity prices is its ongoing commitment to reduce the real price of electricity on the back of internal efficiency gains. Eskom announced its `price compact' in 1991, in terms of which it undertook to decrease the real price of electricity by 20% over the five year period 1992 to 1996. This came on top of a 14% reduction in real price which had already been achieved from 1987 to 1991 (Eskom 1992: 5). In 1994, it made a further commitment, in terms of its so-called `RDP commitments' to reduce the real price by 15% over the period 1995 to 2000 (1995a: 9). Until 1994, all of these targets had been met, which means that 1994 prices were approximately 76% of 1987 levels in real terms. Given its sound financial status and, in particular, declining levels of external debt, there is little reason to believe that the utility will not meet its latest commitment; if this is the case, then the average electricity price in 2000 will be approximately 60%, in real terms, of the 1987 price level (Van Horen 1996). This is a dramatic decrease by any standards.

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[6] This comparison excludes utilities whose tariffs are too low to recover their costs.

[7] This is not to suggest that external costs of hydro-electricity are small or negligible: the contrary is often true in larger projects.

Eskom suggests that these price reductions will be achieved through ongoing productivity improvements, reduced operating expenditure and cost containment. This commitment is linked to the goal of becoming `the world's lowest-cost producer of electricity' (ibid: 3). It is evident that other factors underlie this decrease in electricity prices, notably the decrease in the number of employees, from 66 000 in 1985 to 40 000 in 1994 (ibid: 55). In addition, its reduced exposure to finance charges has contributed significantly to its financial health, with the debt: equity ratio having declined from 3.0: 1 in 1985, to its 1994 level of 1.7: 1 (ibid: 10). The utility's stated intention is to reduce this ratio to parity by 1998; this is achievable given that its levels of capital expenditure have declined considerably in real terms since the mid-1980s because of the situation of over-capacity (Van Horen 1996).

Whilst this question leads to issues which are beyond the scope of this thesis, it is interesting to consider the implications of this strong downward trend in electricity prices for the country's overall economic development path. Low energy prices, especially if they are amongst the lowest in the world, provide a very strong signal to energy consumers and to potential investors. All other things equal, low prices are likely to encourage a heavily energy- and resource-intensive growth path, which once embarked upon, is likely to be difficult to re-direct. Most of the world's wealthiest nations, on the other hand, have increasingly de-coupled economic output from energy input (Eberhard & Trollip 1994); indeed, electricity prices in countries such as Japan and Germany are amongst the highest in the world (Eskom 1995a: 9). Moreover, investments in electricity-intensive industries - such as the multi-billion Rand Alusaf aluminium smelter - are highly capital-intensive and make little use of the country's most abundant resource: labour.

Although there are obviously many fundamental factors influencing the performance of any economy, it is not implausible that higher electricity prices may have played a role in encouraging economies such as those of Japan and Germany to shift away from a resource-intensive base to higher value-added activities in the commercial, financial and service sectors, and that this shift has had positive implications for their economic performance in the long-term. If this is so, then it would be important to consider the South African case: whether, in fact, the goal of achieving the world's lowest electricity prices could be detrimental to long-term economic performance. The internalisation of environmental costs is obviously relevant to this question.

2. Externalities in South Africa's electricity industry

2.1 The impact pathway approach

Given that there is a wide range of environmental impacts arising from the energy sector, it is necessary to adopt a systematic approach to their identification and evaluation. The method used in this study, which has been used in the majority of international externalities studies, is the impact pathway or damage function approach (for example, ETSU 1995, Bernow et al 1995a, 1995b, Rowe et al 1994a, 1994b, 1995, Chapter Two). The damage function approach is illustrated conceptually in Figure 3.2 for the case of power station emissions.

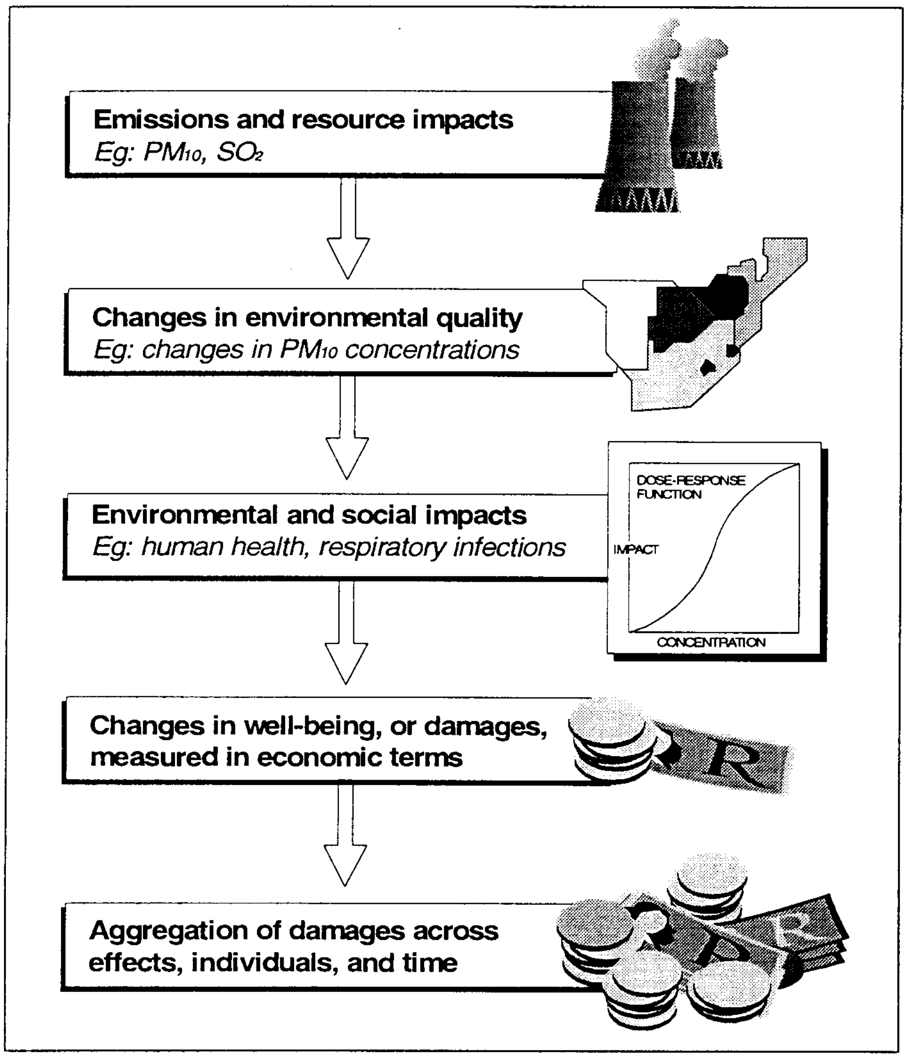

Figure 3.2 The damage function approach for power station emissions *(after Rowe et al 1995: 3).*

The damage function approach entails the identification and quantification of environmental and other damages arising at each stage in the fuel cycle: from the extraction of raw materials (such as coal or uranium), to their transport and processing, to their consumption in power stations, to the impacts of waste products arising in the electricity generation process, and their impacts on human health and amenity, and on the physical and natural environments.

It is evident from Figure 3.2 that the impact pathway approach corresponds with the real-world steps in the fuel cycle; thus an externalities study which succeeds in quantifying the impacts at each stage of the fuel cycle will have a reasonably realistic correlation to the actual impacts arising from that cycle. A successful impact pathway approach will yield results which are more realistic than other externality methodologies,[8] and will therefore be of more use for policy-making. The damage function approach is generally regarded as the preferred approach to assessing environmental externalities (Rowe et al 1995: 2).

It is important to note, however, that the damage function approach is also subject to weaknesses, most of which are related to the fact that it attempts to quantify realistic processes and events. These limitations include the following (Rowe et al 1995: 2):

- Firstly, the approach is highly data-intensive, since information is required about each step in the impact pathway. As a result, such studies, if they are to be comprehensive and scientifically sound, are costly.
- Secondly, professional judgements are required about the most appropriate data to use, since there are often conflicting views in the externalities literature, and the result can be sensitive to these judgements.
- If there are omissions, inaccuracies or biases in the data, these can be compounded throughout the assessment chain, thereby limiting the usefulness of the end results.
- Finally, the damage function approach is fairly complex and draws upon a number of disciplines, which can render such studies inaccessible to the wider audience which may be interested in their results.

Whilst the data requirements of a comprehensive impact pathway assessment are formidable, there is a relatively large body of relevant information which makes it possible to employ the approach in South Africa. Given time and information constraints, however, it is not possible to evaluate externalities at every step in the chain; moreover, beyond a certain

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[8] Less direct methods include, for example, the regulators revealed preference approach - as used, for example, by Bernow et al (1991) for Wisconsin, USA - or `top down' externality approaches - used, for example, by Hohmeyer (1988) for Germany.

point, there will be diminishing returns from widening the scope of the study to include less serious environmental and other impacts. Furthermore, a degree of judgement is required in making decisions regarding which externalities are potentially significant and which are not.

Consequently, a classification system has been developed for purposes of this study, which makes explicit the criteria used in determining the scope of the quantification exercise. In terms of this system, environmental impacts are classified according to the following three categories:

- *Class One impacts*: these are environmental impacts which are potentially serious, and for which sufficient information exists to permit an estimate of their economic value.
- *Class Two impacts*: these are impacts which are potentially serious, but for which there is insufficient data to permit an economic assessment of external costs within an acceptable range of certainty.
- *Class Three impacts*: these are impacts which, on balance of evidence and probability, are not likely to be highly material in relation to other impacts, and therefore no attempt is made to quantify them in economic terms; alternatively, the environmental costs of these impacts have already been substantially internalised.

The sections which follow, therefore, describe those environmental impacts in the coal and nuclear fuel cycles which are viewed as potentially significant. For each impact, the state of scientific knowledge is reviewed, and an assessment is made of the degree of internalisation of costs, and of whether there is sufficient information to estimate their economic values.

2.2 Externalities in South Africa's coal fuel cycle

Environmental impacts arise at most stages in the coal fuel cycle. These impacts are subject to varying levels of management and attempts to ameliorate them; in other words, some of these impacts are already fully or partially internalised into the prices of coal and electricity. In considering the coal fuel cycle, the following impacts may be identified:

- Occupational health effects in coal mining.
- Air and water pollution from coal mining.
- Water consumption in power generation.
- Air pollution from power generation: health impacts.
- Air pollution from power generation: impacts of acidic deposition.
- Air pollution from power generation: visibility impacts.
- Water quality impacts from power generation.

- Greenhouse gas emissions from power generation.

Numerous other impacts arise in the generation of electricity from coal, in addition to those described above; for example, the health impacts of electromagnetic fields (EMFs) around high-voltage transmission lines, the loss of productive land above underground mines due to subsidence, and the aesthetic impacts of large power stations and transmission lines in rural areas. Whilst any number of such impacts may have significant environmental or social impacts in their specific or local contexts, they are not considered in any detail in this study, either because they are not significant in aggregate on a national scale, or because the costs of those impacts have already been substantially internalised.

Each of the impacts listed above will be assessed in more detail in this chapter, and will be quantified in economic terms in the next chapter.

2.2.1 *Occupational health effects in coal mining*

Workers in coal mines are exposed to a number of risks. These include rock falls, methane explosions, transport accidents and accidents in the handling of materials, which may result in immediate injury or death. A second category of occupational risks results from prolonged exposure of workers over a number of years to air pollution resulting from mining activities, with illnesses such as emphysema and pneumonia occurring.

A Commission of Enquiry, the `Leon Commission' reported in mid-1995 on the state of health and safety on South Africa's gold and coal mines. The general tone of the Commission's report was sharply critical of the health and safety management in the country's gold and coal mines, and highlighted the high morbidity and mortality rates compared to other major mining countries. This was more especially the case in the gold mines, which generally operate at much deeper levels than coal mines.[9]

The Leon Commission reported on the number of injuries and fatalities in the mining industry for the period 1984 to 1993; the statistics for coal mines are summarised in Table 3.5.

These data represent averages of open-cast and underground coal mines; the rates for these types of mines, however, differ, as do accident rates from one mine to another. No data was available for the present study regarding the actual number of incidents on the mines serving Eskom's power stations. Nonetheless, this can be calculated from available data sources. The Leon Commission reported that there were 47 injuries and 9 fatalities on open-

[9] Ironically, the release of the Leon Commission report coincided with a serious accident at the Vaal Reefs Gold Mine, in which 104 miners were killed when their lift-cage dropped hundreds of metres down a mine shaft.

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2.2 Externalities in South Africa's coal fuel cycle

Environmental impacts arise at most stages in the coal fuel cycle. These impacts are subject to varying levels of management and attempts to ameliorate them; in other words, some of these impacts are already fully or partially internalised into the prices of coal and electricity. In considering the coal fuel cycle, the following impacts may be identified:

- Occupational health effects in coal mining.
- Air and water pollution from coal mining.
- Water consumption in power generation.
- Air pollution from power generation: health impacts.
- Air pollution from power generation: impacts of acidic deposition.
- Air pollution from power generation: visibility impacts.
- Water quality impacts from power generation.

cast mines during 1993 (1995: 22). Total coal production at open-cast mines was in the region of 35% of the total (Raimondo et al 1995: 12), equivalent to 64 million metric tons (Mt), which means that the fatality rate on open-cast mines was 0.14 per Mt, and the injury rate 0.73 per Mt.

Accident rates for underground mines can be estimated by removing the incidents attributed to open-cast mines from the data in Table 3.5. There is no clear trend in the fatality data shown in the Table, and so the average for the 10 year period will be used: 69 fatalities per year. If 9 fatalities occurred on open-cast mines, then there were 60 deaths on underground mines; this is equivalent to a fatality rate of 0.51 per Mt. For injury data, there is a clear downward trend in the rate of injuries over the ten year period 1984 to 1993, and so the ten year average could overstate current accident rates. Consequently, the average for the last five years will be used, which amounts to an average of 349 injuries per annum, or 2.95 injuries per Mt mined in underground operations.

Table 3.5 Accident data for South African coal mines, 1984 to 1993

(Leon Commission 1995: 19, Minerals Bureau 1994: 32)

Year	Coal mined	Fatalities			Injuries		
	(Mt)	Number	Per 1000 workers	Per Mt mined	Number	Per 1000 workers	Per Mt mined
1984	162.9	73	0.63	0.45	840	7.20	4.94
1985	173.5	93	0.78	0.54	806	6.80	4.65
1986	176.7	66	0.55	0.37	709	5.90	4.01
1987	176.6	121	1.05	0.69	550	4.86	3.11
1988	181.4	53	0.50	0.29	435	4.07	2.40
1989	176.3	54	0.52	0.31	361	3.50	2.05
1990	174.8	50	0.52	0.29	404	4.21	2.31
1991	178.2	42	0.56	0.24	361	4.80	2.06
1992	174.4	46	0.66	0.26	339	5.17	1.94
1993	182.2	90	1.71	0.49	279	5.32	1.53
Average	175.7	69	0.75	0.39	508	5.18	2.90

Of the nine coal power stations, four are served by open-cast mines, three by underground mines, and two by a mixture of both, in roughly equal proportions (Du Plooy 1995). From this, and on the basis of electricity generated at each power station, the split between coal sourced at underground mines compared to open-cast mines can be estimated at around 43: 57 for 1994. Fatality and injury rates can therefore be calculated for coal mines serving Eskom's power stations; these are summarised in Table 3.6.

Table 3.6 Calculated accident rates for underground and open-cast mines supplying Eskom

	% split	Fatality rate (Per Mt)	Injury rate (Per Mt)
Underground	43%	0.51	2.95
Open-cast	57%	0.14	0.73
Total/ weighted average	100%	0.30	1.68

Thus an average of 0.30 fatalities and 1.68 injuries occur for every million tons of coal mined for purposes of power generation. On average, 0.52 kg of coal is used for each kWh of electricity generated (Eskom 1995a: 54), so this translates into 0.156 fatalities and 0.874 injuries per thousand GWh of electricity produced. Applying this rate to 1994 electricity production of 149 443 GWh, then a total of 23 workers would have died in coal mines in order to supply Eskom in that year, and a further 131 would have been injured.

The nature of injuries sustained by coal miners varies widely, from relatively minor injuries to permanently disabling ones. A breakdown of the kind of injuries on coal mines averaged over 1993 and 1994, obtained from the Government Mining Engineer's office, shows that the main kinds of injuries were of the following kinds: arm, hand and finger injuries 37%, legs and feet 36%, trunk 13%, head and neck 9%, and the remainder unspecified or multiple injuries (GME 1995). The definition of `reportable injuries' includes those incurred in accidents of a serious nature where the worker cannot work for 48 hours or more, as well as those occurring in the `normal course of duties' where the person is away from work for 14 days or more. The range of periods for which an injured person cannot work obviously varies widely, depending on the kind of injury. According to the GME, most reportable injuries lead to considerably more than 14 days of lost work, although no data was available on the exact number of lost days. According to Rand Mutual Association (which pays compensation to injured workers), injuries vary from 6 to 8 weeks for broken legs or arms, to 8 to 24 months for more serious spinal injuries. Since over 70% of injuries during 1994 were to limbs, a central estimate of 8 weeks for injuries will be used for this study. Low and high estimates are taken as 4 and 12 weeks respectively; these will be accounted for in the valuation exercise in Chapter Four.

In addition, cash compensation is paid to workers according to a standard scale of benefits, in terms of the Compensation for Occupational Injuries and Diseases Act, in respect of all reportable injuries and deaths. These will be considered in the valuation exercise in Chapter Four.

With respect to chronic and acute respiratory illnesses resulting from prolonged exposure to air pollution, evidence heard by the Leon Commission suggested that these effects are significant (1995: 16-17). The most common effects occurring amongst gold and coal miners are tuberculosis, emphysema, hearing impairment, pneumoconiosis and silicosis. However, there have been no published studies of the pollution levels to which workers are exposed, nor of the dose-response relationships applicable to coal miners. Quantification of these impacts, in either physical or economic terms, is therefore not possible in the present study, and hence these occupational health impacts are accorded a Class Two classification.

In addition to the obvious questions raised by the above statistics about the safety management policies of the mining industry - which are beyond the scope of this thesis - the relevant question for present purposes relates to whether the health and safety risks which are faced by coal miners are reflected in their wages (and therefore the prices of coal and, ultimately, electricity) - as suggested by hedonic wage pricing studies. All other things equal, in an efficient market (that is, with conditions of free bargaining, full information, etc.), the environmental risks would be internalised into worker's higher wages (Freeman 1993: 421). However, the South African labour market does not even remotely approximate an efficient market, as evidenced most significantly by an estimated unemployment rate of 43% in 1994 (SAIRR 1995: 487). Furthermore, the iniquities of apartheid employment practices meant that there have historically been significant racial discrepancies in wages, and although these have been narrowed or eliminated in recent years, significant distortions exist. Finally, to the extent that wages may be higher for coal miners than for workers in industry, this is partly attributed to the relative strength of trade unions on the mines compared to other sectors rather than to any premium extracted by them for additional risks.[10] In this context, the labour market contains so many `distortions' compared to a hypothetical efficient market, that it would be difficult to identify a wage premium which is attributable to higher occupational risks.

The task of applying economic values to these impacts is addressed in detail in Chapter Four. For the present, however, the occurrence of deaths and injuries will be classified as a

[10] The National Union of Mineworkers (NUM) is the largest union in the country.

Class One impact, since sufficient information exists regarding the rates of injury and death to quantify them. In the case of chronic and acute illnesses resulting from exposure to air pollution on the mines, however, insufficient information exists to quantify the economic value of this externality, and so this is classified as a *Class Two* impact.

2.2.2 *Air and water pollution from coal mining*

Coal mining is undertaken in South Africa through four methods: board and pillar, open-cast mining, long-wall mining and pillar extraction (Raimondo et al 1995: 12). Of these, the first two dominate, accounting for 50% and 35% respectively of total coal extracted in 1987. The air and water impacts of coal mining vary widely, depending in the first instance upon the kind of mine concerned.

In general, Eskom secures its coal from mines located adjacent to the power stations. These mines are owned by private sector mining houses, who generally have entered into long-term contracts with Eskom to supply its needs over the lifetime of the power station (30 to 40 years on average). These long-term contracts have advantages for mining houses in that they are assured of consumer demand for their output, and for Eskom in that it can secure favourable prices for coal, which is the main variable cost in the generation of electricity. In 1994, Eskom paid an average of just under R30 per ton of coal burnt (1995a: 54). By comparison, the average price of exported coal (which is generally of a higher quality than Eskom's coal) was R85 per ton in 1993, local coal prices for wholesale industrial consumers averaged R39 per ton, and at the end of the distribution chain for low-income households, retail prices were in the region of R200 per ton (Minerals Bureau 1994: 33, Palmer Development Group 1995: 21). These price variances are attributable largely to the costs of transportation and distribution. This serves to emphasise then, the cost advantages Eskom enjoys by securing its coal supplies from mines close enough to its power stations to be able to transport coal from the mine's stockpile to the furnaces, by *conveyor belt*. In many other countries, coal is delivered to power stations by rail, which given the bulky nature of the commodity, adds considerably to the cost of electricity.

To the extent that Eskom purchases its coal from un-subsidised private sector mining companies, it cannot be said that its coal is subsidised.[11] Nonetheless, there may be implicit costs in the coal mining cycle which are not accounted for in the operations of mining

[11] An interesting question relates to whether the discount rate used by mining companies in their decisions over the rate of mineral extraction is different from economy-wide discount rates, since this could lead to lower or higher coal prices than would otherwise prevail. Data on this issue, however, is not readily available and it is beyond the scope of this thesis to pursue this further.

companies. The main external costs - over and above occupational health costs described above - arise from air pollution and water pollution in the mining process and related activities. In particular, three categories of impacts arise:

- open cast mining is responsible for increased levels of dust and airborne particulate matter caused by blasting, vehicular traffic and movement of coal and waste products;
- large dumps of discard coal are prone to spontaneous combustion when not compacted sufficiently, and where this occurs, significant amounts of air pollution can be emitted for long periods of time;
- water supplies in the proximity of coal dumps and open mine pits can be degraded by chemical run-off, thereby affecting ground water supplies as well.

The relevant question for present purposes, is whether there are significant and measurable economic costs associated with any of these impacts, and if there are, whether they are internalised or not.

With respect to the air quality impacts described in the first two points above, it appears that coal mines and their dumps have historically made a significant contribution to ambient pollution levels in the coal-mining areas (Annegarn 1995: 15). However, there is no data regarding emissions from specific mines or dumps, which makes it difficult to apportion emissions between the various coal-consuming sectors.

At the same time, the number of burning mine dumps has been reduced considerably in the past few years due to better management practices on the part of the relevant mining companies (ibid). This suggests that the environmental costs of mine management are increasingly being internalised. One of the mechanisms through which this takes place is the `Environmental Management Plan' which all mines are required to prepare and submit to the Department of Mineral and Energy Affairs before they receive a licence to commence mining activities (Raimondo et al 1995: 29). Prospective mining companies are also required to lodge a bond (a monetary deposit) up-front, to insure against the future costs of rehabilitation and decommissioning.

With respect to the water quality impacts of leaching and ground-water pollution, there is minimal information on which to base an assessment of any economic costs which may arise. Although it is not feasible to estimate external costs in the present study, it is important to note that the issues of water quality and water scarcity are receiving increasing policy attention at all levels of government.

For purposes of this study, these impacts fall both into *Class Two*, to the extent that the air and water quality impacts of coal mining may be serious, but that there is a large degree of uncertainty over their extent; and into *Class Three,* to the extent that they have already been internalised through environmental management procedures.

2.2.3 *Water consumption in power generation*

Coal-powered electricity generation accounts for about 3% of South Africa's bulk water consumption (Roome 1995). Water is an integral part of the thermal power generation process, being used not only (as steam) to drive the turbines which generate electricity, but also to cool down the steam in large cooling towers. Most of Eskom's power stations utilise conventional wet-cooling processes, although two, namely Kendal and Matimba, use dry-cooling processes which were pioneered by Eskom.[12] Average net water consumption in each of the coal power stations is shown in Table 3.7, together with the actual price paid by the utility in 1994; (refer back to Figure 3.1 for the location of each power station.)

Table 3.7 Net water consumption in Eskom's coal-powered stations, 1994

(Fraser 1995)

	Water consumed		Average price
	Total (million m^3)	Per unit (l/kWh)	(R/m^3)
Arnot	8.875	1.76	0.55
Duvha	41.009	1.71	0.59
Hendrina	27.188	1.69	0.40
Kendal	3.667	0.16	1.78
Kriel	32.910	1.97	0.79
Lethabo	31.624	1.87	0.12
Matimba	4.249	0.18	0.50
Matla	34.696	2.10	1.23
Tutuka	35.239	1.96	0.66
Total/average	219.457	1.43	0.66

[12] In these power stations, the cooling towers utilise massive fans to blow air onto hot water pipes to cool them down.

Water costs represent a small fraction of Eskom's total operating costs: in the region of 1.8% of direct operating costs (excluding depreciation and finance charges) (calculated from Eskom 1995a: 41). This, however, understates the importance of water as an input into the electricity generation process - it is an essential input into the coal fuel cycle. This is reflected in the fact that Eskom is a high-security water consumer, meaning that its supply is the last to be affected in times of drought or supply interruptions. This is also underlined by the fact that Eskom was directly involved in the construction or financing of several dams, long before other consumers demanded water in those areas.

As in the case of coal, Eskom has benefited from water prices which have been low and stable over time. There is a wide range of pricing contracts in place with respect to Eskom's water purchases, each dependent upon the specific source of supply. In general, Eskom purchases its water from the Department of Water Affairs and Forestry (DWAF), and pays on the basis of the historic costs of supplying that water, as distinct from the marginal cost. Importantly, the cost of supplying water is dependent primarily on the capital costs of constructing the necessary infrastructure. This means that if historic costing is used as a basis for water pricing, and the capital infrastructure was constructed some time ago, then the price of water will be much lower than the cost of supplying an additional unit of water today. This, together with the varying sources of supply for Eskom's power stations, accounts for both the range of water prices, and the relatively low prices in some cases. Lethabo power station, for instance, draws its water from the Vaal River which was dammed in the early 1900s, hence its water price is considerably lower than the marginal cost of new water supplies. Put differently, the price of water paid by Eskom is considerably lower than its social cost.

Since this divergence from its social cost is an important economic impact, and there is relatively good information regarding the current consumption and pricing regime, this issue is classified as a *Class One* impact. Whilst the under-pricing of water differs in nature from other external effects such as air pollution emissions from power stations, the divergence of water prices from their social costs is relevant to the question posed at the commencement of this thesis, which was to identify those environmental and resource impacts of the greatest significance to society - hence the inclusion of this impact in the present analysis. A range of alternative water pricing scenarios, and their impact on electricity tariffs, is considered in Chapter Four.

2.2.4 *Air pollution from power generation: health impacts*

The evaluation of the health impacts from power station emissions is one of the more complex, but important externalities to be considered. Quantification of health impacts requires information for five major steps in the impact pathway:

- the quantities of pollution emitted by power stations;
- the dispersion and ultimate deposition of those pollutants;
- the distribution of the `receptors', which include human populations at risk;
- the responsiveness of human health to various exposures (doses) of pollution;
- the valuation of increased morbidity and mortality.

Each of these steps is addressed below, except for the last, which is addressed in the next chapter.

2.2.4.1 *Quantity of pollution emitted by power stations*

The coals used in South Africa's power stations are generally of relatively poor quality, since the highest grade coals are exported. The average percentage ash and sulphur content, and energy content (in Megajoules per kg) of coals used by Eskom varies widely, as shown in Table 3.8.

Table 3.8 Average ash, sulphur and energy content of coal used in Eskom's power stations, 1994
(Source: Eskom Generation group)

	Ash content (%)	Sulphur content (%)	Energy content (MJ/kg)
Arnot	21	1.02	22.3
Duvha	26	1.08	22.5
Hendrina	24	0.85	22.0
Kendal	31	1.11	19.2
Kriel	26	0.88	20.8
Lethabo	39	0.59	15.2
Matimba	32	1.26	18.5
Matla	24	1.21	20.9
Tutuka	25	1.41	21.0

It is evident that the ash content is uniformly high, ranging from 21% at Arnot power station, to 39% at Lethabo; the latter also has the lowest calorific value (energy content), at 15.2 MJ per kilogram of coal. It is significant to note that with such a low energy content, `coal' could not be used in most conventional commercial or domestic processes[13]. Thus, electricity is being generated from a product which would otherwise have little or no economic value. In other words, its opportunity cost is close to zero.

The negative side of this is that, all other things equal, particulate emissions and ash production are higher than would be the case with low-ash coals, in other words, it has the potential to impose relatively significant social costs. As a consequence, Eskom's pollution control policy has been concerned primarily with the control of particulate emissions. All of its coal-power stations currently in operation, utilise electro-static precipitators (ESPs) to remove the bulk of particulate emissions from flue gases. These typically operate with an efficiency of around 90 to 99.7% (Tilley & Keir 1994), although it is important to note that it is the finest particles (that is, with the smallest diameter) which present the greatest potential health hazard, since it is these which are small enough to be respirable.[14]

A second technological option which is often used by electricity utilities internationally, entails the use of bag filters to increase the portion of particulates which are removed from exhaust gases. Where bag filters are used successfully, in combination with ESPs, efficiencies of 99.99% are possible (Hanson 1992). Eskom has installed bag filters on a trial basis in some of its power stations, as shown in Table 3.9.

Eskom's air pollution control policy has a further dimension, to address the particular atmospheric conditions prevailing in the Mpumalanga Highveld: a strong inversion layer which inhibits the dispersal of ground-level or low-level emissions, especially during winter months (Tyson et al 1988). As a result, the newer power stations have tall chimney stacks, so that emissions penetrate the inversion layer and are released into the upper atmosphere. The effects of this will be described shortly.

A final point to note in relation to Eskom's air pollution policy, is that it has decided that desulphurisation and denitrification technologies are not warranted. Thus, apart from the tall stack policy, no active measures are taken to reduce the emissions of sulphur dioxide or nitrogen oxides. This is an issue which has received much attention in recent years, although there has been no systematic investigation of the costs and benefits of respective pollution control options.

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[13] Indeed, this coal cannot even be ignited with a blow-torch!

[14] Particulate matter with a diameter of 10 μm (microns) or less is usually regarded as being in the respirable range (hence the label PM10).

Particulate, sulphur dioxide and nitrogen oxide emissions from Eskom's coal power stations in 1994 were as shown in Table 3.9. These data represent emissions for 1994 as calculated by Eskom; it should be noted that there is no independent monitoring or measurement of emissions by an environmental authority acting principally in the interests of public health. Eskom's power stations are governed by Registration Certificates, which are issued by the Chief Air Pollution Control Officer (CAPCO), in terms of which maximum permissible emission rates are specified (Petrie et al 1992). Transgressions of these limits, however, such as occurred at Hendrina power station, do not necessarily lead to direct penalties or costs; in the case of Hendrina during 1994, the CAPCO granted the utility a `temporary exemption' after `taking circumstances into account'.

Before addressing the dispersal of pollution from power stations, it is important to note another source of air pollution which is not accounted for in the emissions data quoted above, namely pollution originating from the ash dumps at power stations. The combustion of coal in power stations, much of it with a relatively high ash content, unavoidably leads to extremely large quantities of ash production. Some 22 million tons of ash were produced by Eskom's power stations in 1994, of which about 3% was re-used for cement or brick-making (Eskom 1995b: 23). This translates into a continuous production rate of 42 tons of ash per minute. This ash has no other commercial value at present.

Table 3.9 Total suspended particulate (TSP), sulphur dioxide and nitrogen oxide emissions by Eskom's power stations in 1994

	Bag filters	TSP emissions		SO_2 emissions		NO_x emissions	
		kt	kg/MWh	kt	kg/MWh	kt	kg/MWh
Arnot	3 (of 6) units	11.00	2.41	35.8	7.86	26.3	5.77
Duvha	3 (of 6) units	8.17	0.37	180.3	8.21	136.7	6.22
Hendrina	none	49.70	4.19	90.5	7.62	72.9	6.14
Kendal	none	4.63	0.24	167.8	8.65	123.2	6.35
Kriel	none	10.56	0.79	103.9	7.76	126.5	9.44
Lethabo	none	5.45	0.31	123.0	6.89	143.1	8.01
Matimba	none	22.58	1.00	193.8	8.54	92.4	4.07
Matla	none	4.54	0.24	149.4	7.97	130.9	6.99
Tutuka	none	5.79	0.33	122.2	6.97	108.9	6.21
Total/Ave		122.42	0.84	1166.7	7.88	960.9	6.49

Eskom's policy is to dispose of this ash either by back-filling mined areas, or in ash dams and on ash dumps. The latter are sited on land owned by itself, and to minimise the amount of ash liberated into the atmosphere, the dumps are covered with grasses and other vegetation where possible. In the critical period before vegetation can stabilise the dumps, waste water is used to reduce the amount of dust which can be blown into the air. Inevitably, however, particularly in windy conditions, significant amounts of dust can originate from these dumps. There is little quantitative information about the amount of pollution which so arises, and so it cannot be included in the present analysis. However, it is clear that the exclusion of this source of pollution from the air quality modelling exercise will understate the contribution of power stations to ambient pollution levels. Moreover, the fact that these ash dumps are at or close to ground level, means that their impact is relatively greater than the impact of emissions from tall chimney stacks. Consequently this source of pollution is not considered any further, and is classified as a *Class Two* impact.[15]

2.2.4.2 Atmospheric conditions and dispersal of pollution

The quantity of pollution emitted by power facilities is the first main link in the electricity generation part of the impact pathway. The next step implicit in the damage function approach concerns what happens to those pollutants after they are emitted, that is, how they are dispersed in the atmosphere and where they are deposited. For this to be assessed, information is required regarding the following key variables:

- physical emission characteristics, such as the height of chimney stacks, and the speed, volume and temperature of flue gas emissions;
- atmospheric conditions, including wind patterns (derived from long-term data), mixing heights and atmospheric stability.

With respect to the first item, data have been collected from Eskom's Generation Group for each of the nine coal power stations which were operating in 1994. These data are summarised in Appendix 1. The most significant factor to note is that all of Eskom's power stations have relatively tall chimney stacks of 200 metres or more, with the exception of Arnot (193 metres) and Hendrina (110 metres). This is consistent with the Eskom policy described earlier - in the absence of technologies to reduce sulphur emissions and given that significant quantities of particulate matter are emitted, the chimneys are designed to

[15] The considerable quantity of ash generated by Eskom's coal power stations is a major focus of its environmental management programmes, not only from an air quality perspective, but also with respect to ensuring there is minimal contamination of ground water close to ash dumps and dams. This is a major area of ongoing activity for Eskom.

penetrate the inversion layer, which would otherwise have trapped those pollutants close to the ground before they could be diluted.

With respect to data on atmospheric conditions, long-term surface wind data averaged over a period of 10 to 20 years, was drawn from Weather Bureau sources for fifteen stations in South Africa and two in Namibia (Weather Bureau 1975). Average frequency distributions for sixteen wind vectors (N, NNW, NW, WNW, W and so on) in various speed classes were obtained. These data, together with the technical power station data described above, were used in atmospheric models contained within the EXMOD model, described in the next chapter and in Appendix 3.

The results of the air pollution dispersal and valuation exercise will be described in the next chapter. It is worth briefly reviewing, here, the state of information about pollution levels on the Mpumalanga Highveld. Once again, most of the data on pollution levels in this region are derived from monitoring undertaken by Eskom's own scientists, as there is no regulatory agency in South Africa equipped to do this.[16] Eskom has operated a monitoring network in the Mpumalanga province since 1979, with various site changes having taken place over that period. Data are available for several key pollutants: sulphur dioxide, nitrogen oxides, particulates and ozone. Data on concentrations for each of these pollutants are summarised briefly below.

Sulphur dioxide levels over the past decade were reported to be generally well within annual guideline levels, with the average over the period 1979 to 1986 being around 26 μgm^{-3} (range 9 to 41 μgm^{-3}) (Turner et al 1990: 5, Tyson et al 1988: 46). The Department of Health annual guideline, by comparison, is 78 μgm^{-3}.[17] However, there were more frequent exceedances of guidelines over shorter (hourly and daily levels) monitoring periods. Whilst long-term trends showed an increase in average SO_2 levels over the period 1979 to 1983, corresponding with the commissioning of new power stations during that period (Turner 1987), Eskom reported that the trend was reversed after 1986. The most recent results from the utility's monitoring network, for 1993, reported no exceedances of SO_2 health guidelines at its six monitoring stations (Rorich & Turner 1994: 6).

Pollution data for *nitrogen oxides* suggest that this pollutant does not represent a major problem: long-term concentrations in the 1980s were around 15 μgm^{-3} compared to the

[16] Eskom has contracted consultants at various stages to audit its monitoring system, and it has generally been satisfied with the results.

[17] South Africa has a system of published air quality guidelines (not standards) which are fairly similar to the US EPA standards for the pollutants discussed in this section.

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national long-term guideline of 376 μgm^{-3} (Lennon & Turner 1992: 3). Results for 1993 showed that pollution levels were below guideline values, although short-term peaks were not insignificant (Rorich & Turner 1994: 6).

Particulate concentrations in the Mpumalanga Highveld over the period 1979 to 1983 were well below guidelines, at around 17 μgm^{-3} compared to the annual guideline of 150 μgm^{-3} (Turner et al 1990: 5). According to Eskom, its power stations were responsible for around 20% of ambient particulate concentrations, compared to 46% from smouldering coal dumps and local industries - due to the fact that its emissions were better diluted than low-level emissions. The results for 1993 showed no major changes to this situation (Rorich & Turner 1994: 7).

Finally, Eskom's monitoring of pollution levels for *ozone* found that hourly, daily, monthly and annual guidelines were frequently surpassed during the period 1983 to 1988 (Turner et al 1990: 20, 23). Annual averages were two to three times the annual guideline of 20 μgm^{-3} and the trend was increasing. The significance of this is that high levels of ozone can cause direct damage to crops and health. Ozone is not a primary pollutant, meaning that it is not emitted directly from any source, but is formed by photolytic (that is, activated by sunlight) dissociation of oxygen in other gases, so its concentration is elevated by the presence of other pollutants such as NO_x.

Research undertaken subsequent to the publication of the 1988 report by Tyson et al has found atmospheric dispersion patterns different to those originally expected. Firstly, emissions from high level stacks were found to be deposited to the ground (by downward air flows) at distances relatively close to the emissions sources - in other words, the previous assumption about long-range transport of pollutants emitted into the upper atmosphere by tall chimneys may not be correct (Annegarn 1995: 22). This means that the highest pollution impacts would be encountered within a radius of tens rather than hundreds of kilometres from emissions sources. Second, and consistent with the above point, there was found to be a decreasing concentration of gaseous pollutants, such as sulphur dioxide, as the distance from the high density emission area in the Mpumalanga Highveld increased. Thirdly, the work reported by Annegarn confirmed that power stations give rise to high peak concentrations relatively close to the power stations (1995: 16), but that other low-level sources, such as smouldering coal dumps and underground fires in abandoned coal mines, are significant determinants of overall pollution levels.

To summarise then, the data from Eskom suggest that the pollutants which approach or exceed health guidelines most frequently in the Mpumalanga Highveld, are ozone and sulphur dioxide. Of less concern are concentrations of particulates and nitrogen oxides, which are reported to be well within Department of Health guidelines.

2.2.4.3 Distribution of human population at risk

Once information exists about the quantity of pollution emitted from the power stations, and about its dispersion in the atmosphere once it leaves the chimney stacks, as well as its ultimate deposition at ground level, it is important to know what the `receptors' are at ground level. These receptors include not only human beings but also buildings, crops, fences, vehicles and other items which might be affected by pollution exposures. For purposes of quantifying the present impact, namely the health effects of pollution emissions, clearly the relevant variable is the distribution of the human population across South Africa.

Consequently, population data for South Africa, including the former homelands, were assembled from census sources, for each magisterial district in the country. The population's age distribution was also obtained, and for each magisterial district (identified by its geographical co-ordinates), entered into a database file created for this purpose in the EXMOD model. In effect, therefore, a demographic `map' was created describing the distribution of the population in relation to each of the power stations being analysed. This underlying population distribution thus provides the basis for subsequent quantification of health outcomes.

2.2.4.4 Dose-response functions for air pollutants

The dose-response relationship provides the link between ambient pollution exposures and health outcomes among exposed populations - in this case, respiratory illnesses. The human health effects of pollution exposures have been widely studied in a variety of countries in response to a range of environmental conditions. In South Africa, there have been relatively few studies of the health effects of air pollution, and although a handful of studies have attempted to find correlations between environmental quality - mainly particulate concentrations - and health outcomes - mainly respiratory illnesses - (see, for example, Terblanche et al 1992a, 1993 and Chapter Five) there have been no studies which have quantified the dose-response function for pollution exposures.

It is therefore necessary to draw on the international literature to identify dose-response functions which might be applicable to South Africa. For purposes of the project in New York state, USA, which developed the EXMOD model described in Chapter Four, extensive reviews were undertaken of epidemiological and bio-medical literature, in order to derive dose-response functions which could be used with a satisfactory degree of certainty (Rowe et al 1994b: chapter V). A strict set of criteria was applied in the selection of epidemiological studies for that purpose.[18] On the basis of a large number of studies which followed similar

[18] These are described in detail in Rowe et al (1994b: V-3 to V-4).

Table 3.10 Summary of dose-response relationships for particulates (PM10) in various US studies *(Rowe et al 1994b: V-38)*

Health outcome	Population sector	Risk factors	Probability[19]
Mortality Daily mortality risk factors given a 1 μgm^{-3} change in PM10 concentrations	>= 65 years	L $10.1*10^{-8}$ C $16.9*10^{-8}$ H $25.4*10^{-8}$	L 33% C 34% H 33%
	< 65 years	L $0.14*10^{-8}$ C $0.23*10^{-8}$ H $0.35*10^{-8}$	L 33% C 34% H 33%
Morbidity Chronic bronchitis (CB) annual risk factors given a 1 μgm^{-3} change in annual PM10 concentrations	>= 25 years	L $3.0*10^{-5}$ C $6.1*10^{-5}$ H $9.3*10^{-5}$	L 25% C 50% H 25%
Respiratory hospital admissions (RHA) daily risk factors given a 1 μgm^{-3} change in PM10 concentrations	All	L $1.8*10^{-8}$ C $3.3*10^{-8}$ H $4.8*10^{-8}$	L 25% C 50% H 25%
Emergency room visits (ERV) daily risk factors given a 1 μgm^{-3} change in PM10 concentrations	All	L $3.2*10^{-7}$ C $6.5*10^{-7}$ H $9.7*10^{-7}$	L 25% C 50% H 25%
Asthma attacks (AA) daily risk factors given a 1 μgm^{-3} change in PM10 concentrations	For population with asthma	L $0.9*10^{-4}$ C $1.6*10^{-4}$ H $5.4*10^{-4}$	L 33% C 50% H 17%
Restricted activity days (RAD) risk factors given a 1 μgm^{-3} change in PM10 concentrations	>= 18 years	L $0.8*10^{-4}$ C $1.6*10^{-4}$ H $2.5*10^{-4}$	L 33% C 34% H 33%
Days with acute respiratory symptoms (ARS) risk factors given a 1 μgm^{-3} change in PM10 concentrations	All	L $2.2*10^{-4}$ C $4.6*10^{-4}$ H $7.0*10^{-4}$	L 25% C 50% H 25%
Children with bronchitis (B) annual risk factors given a 1 μgm^{-3} change in annual PM10 concentrations	<18 years	L $0.8*10^{-3}$ C $1.6*10^{-3}$ H $2.4*10^{-3}$	L 25% C 50% H 25%

[19] As discussed shortly, these probabilities have been adjusted to 33: 34: 33 for this study, to reflect the uncertainty inherent in applying US dose-response functions to South Africa.

Table 3.11 Summary of dose-response relationships for ozone (O_3) in various US studies
(Rowe et al 1994b: VI-28)

Health outcome	Population sector	Risk factors	Probability[20]
Mortality			
Daily mortality risk factors given a 1 ppm change in O_3 concentrations	All	L 0 C $3.3*10^{-6}$ H $6.6*10^{-6}$	L 33% C 34% H 33%
Morbidity			
Respiratory hospital admissions (RHA) daily risk factors given a 1 ppm change in daily high-hour O_3 concentrations	All	L $8.4*10^{-6}$ C $13.7*10^{-6}$ H $19.0*10^{-6}$	L 33% C 34% H 33%
Asthma attacks (AA) daily risk factors given a 1 ppm change in daily high-hour O_3 concentrations	For population with asthma	L 0.106 C 0.188 H 0.520	L 33% C 50% H 17%
Minor restricted activity days (RAD) annual individual risk factors given a 1 ppm change in daily high-hour O_3 concentrations	All	L $1.93*10^{-2}$ C $4.67*10^{-2}$ H $7.40*10^{-2}$	L 25% C 50% H 25%
Acute respiratory symptoms (ARS) annual individual risk factors given a 1 ppm change in daily high-hour O_3 concentrations	All	L 0.070 C 0.137 H 0.204	L 25% C 50% H 25%

There are two final considerations regarding this dose-response relationship data, the first of which concerns its applicability to South Africa. The population characteristics in the United States and Canada, from where the epidemiological data were derived, differ considerably from those in South Africa, as do environmental conditions. As a result there is an unavoidable degree of uncertainty in applying the data in South Africa. The direction of the bias which may arise, however, (that is, whether the actual health impacts will be under- or over-stated), is not clear. At the most basic level, human physiology and responses to

[20] As discussed shortly, these probabilities have been adjusted to 33: 34: 33 for this study, to reflect the uncertainty inherent in applying US dose-response functions to South Africa.

environmental conditions do not differ according to national boundaries. However, there are marked differences in the health status of populations and in average air quality conditions in the two sets of population. The relatively lower level of income and wealth in South Africa, coupled with the higher proportion of people living in conditions of poverty or near-poverty, translates into generally poorer health status in South Africa, as reflected in higher mortality rates among infants, lower life expectancy and so on (UNDP 1994: 130). Factors such as lower nutritional status, for instance, mean that South Africans may have lower resistance to environmental hazards and therefore be more susceptible to illness (Terblanche 1995). The bias which this factor introduces means that the above dose-response relationships would tend to understate the actual health outcomes in South Africa.

However, this is only one of many differences which could introduce bias in either direction. Consequently, in order to account for the additional uncertainty arising from the use of North American dose-response functions in this study, the probability factors in Tables 4.10 and 4.11 will be adjusted to reflect this uncertainty. The probabilities assigned to low, central and high estimates will be adjusted (where this is not already the case) to a 33: 34: 33 distribution, for purposes of the modelling described in the next chapter.

The final factor for present purposes concerns the question of thresholds - the pollution level at which health effects begin to occur. Most environmental regulatory regimes, such as the South African system of guidelines or the American system of standards, are based on an assumption that there are specific levels of pollution below which threats to public health will be negligible. However, available epidemiological evidence suggests that there may not be such a threshold level for pollutants such as particulates and ozone (Rowe et al 1994b: V-7). Rather, it has been found that health effects occur even below current health guidelines, and in some studies, the dose-response function has been the same for the lowest quartile of particulate pollution as for the highest quartile (ibid: V-8). Thus the default assumptions from the New York study, of a zero threshold, are not adjusted for present purposes.[21]

To conclude, although this is a complex issue with high data requirements, there appears to be sufficient information to use the tools which are available, to quantify the economic

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[21] A more technical question concerns the shape of the dose-response function, particularly at higher levels of pollution: if there are (proportionately) diminishing or increasing human responses at high levels of pollution, this would require a more complex, non-linear function, coupled with data on current ambient pollution levels. Since there is little data available on this question, it can only be flagged here, and a linear dose-response function is assumed.

impacts of these health effects. They are therefore given a *Class One* classification for present purposes.

2.2.5 *Air pollution from power generation: impacts of acidic deposition*

Pollutants emitted into the atmosphere can have a significant impact on ground-level objects, since they are eventually deposited to the surface, through processes of wet and dry deposition (Piketh & Annegarn 1994). The experiences in Europe in the 1970s and 1980s showed that emissions of sulphur dioxide from sources in highly-industrialised areas, such as the Ruhr valley in Germany, or from the coal power stations in the UK, caused high levels of acidic deposition in neighbouring areas (Levy 1995: 59). In the case of Britain's power stations, emissions were transported over long distances and deposited in Norway and Sweden, causing measurable damage to their forests and water courses.[22] Such issues have granted the issue of acidic deposition (or in popular terms, `acid rain') a relatively high profile amongst environmental concerns, both internationally, and to an extent, in South Africa.

The view is fairly frequently expressed in South Africa, that Eskom's power stations emit large quantities of pollutants which, in turn, cause acid rain (see, for example, Coetzee & Cooper 1991: 132). In scientific circles, the potential risks were first raised in a 1988 report on air pollution in the (then) Eastern Transvaal Highveld, in which the concern was raised that the high concentration of power stations and other sources of industrial pollution, coupled with unfavourable atmospheric conditions, could lead to high levels of acidic deposition (Tyson et al 1988). This, it was postulated, could cause damage to forests, crops, buildings, fences and other materials sensitive to acidic corrosion.

Following the publication of that report, there have been a number of studies which have measured acidity and have monitored the response of environmental elements such as forest productivity, soil and water chemistry (Olbrich & Kruger 1990, Turner et al 1990, Turner 1994b).[23] The energy sector of the Southern African Development Community (SADC) even commissioned a study to establish whether emissions from South Africa were causing high levels of acidic deposition in neighbouring states such as Mozambique and Swaziland (Sivertsen et al 1994). In general, the results of these scientific studies have not yet confirmed the popular notion that `acid rain' is a serious problem in Mpumalanga province and surrounding regions.

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[22] Although it should be noted that some scientists still dispute this.

[23] For a review of these studies up to 1993, see Van Horen (1994: 11).

The phenomenon of acidic deposition and its impacts is fairly complex, and is dependent upon a range of local conditions, many of which differ markedly between South Africa and other parts of the world where acidification has been found to be a problem. In order for external costs of acidic deposition to be quantified, several factors need to be considered:

- First, wet deposition rates need to be assessed. This refers to the quantity of acidifying species (such as sulphates) washed out of the atmosphere during rainfall episodes. Tyson et al (1988: 62) reported rates of up to, and sometimes in excess of 20 kg/ha per annum, which is regarded as a critical threshold in other contexts. Monitoring by Eskom, however, found `reasonably low' levels of wet deposition over a seven year period (Turner 1993 in Piketh & Annegarn 1994: 5).
- Second, dry deposition rates should also be estimated. Since South Africa is a relatively dry country, it has been suggested that dry deposition of acidic species (that is, not through rain or mist) could be even more significant than elsewhere (Tyson et al 1988: 65). Eskom's early calculations suggested that dry deposition could exceed wet deposition by a factor of four in some regions on the Highveld, with dry rates of around 20 kg/ha per year (Turner 1994b: 6). Similarly, Olbrich et al (1993) found that dry deposition rates exceeded wet deposition rates by 2.7 times. In one study which calculated dry deposition rates at two sites (one on the Mpumalanga escarpment and the other in the Lowveld), dry deposition rates were found to be fairly low, at 5.3 and 1.2 kg/ha/year respectively (Piketh & Annegarn 1994: 4). In another study, by contrast, sulphate deposition rates were estimated to range from 49 to 81 kg/h per year (Wells 1993, quoted in Piketh & Annegarn 1994). There appears, therefore, to be a wide range of estimates of actual deposition rates at various locations.
- An important factor in assessing damages from acidification, is the sensitivity of soils and ground cover. Soils in the Mpumalanga province have varying degrees of sensitivity, and are often fairly alkaline, in which cases acidic deposition can initially have a positive effect on the productivity of soils, crops and forests. Efforts have recently been made by the CSIR to develop an information system which contains information on the sensitivity of soils and water systems in the Mpumalanga province, against which deposition data can be plotted, so that areas of greatest concern can be targeted (Olbrich et al 1995). A range of maps can be produced using this system, showing the sensitivity of soils, water, actual deposition rates, and combinations of any of these. This system, known as the Atmospheric Deposition Risk Advisory System (ADRAS) could be used as a tool in identifying areas where damages will be most significant, and undertaking appropriate ameliorative measures.

- One further dimension of acidic deposition, relates to the corrosion of buildings, vehicles, fences and other susceptible materials. Apart from anecdotal observations that this represents a significant problem in some regions, monitoring by Eskom of corrosion rates at various sites in Mpumalanga and the Vaal Triangle suggests that rates are low, although limitations on the monitoring techniques were acknowledged (Van Rensburg 1994). Quantitative information on this impact is therefore scarce.

Only one field study has been undertaken to-date in the Mpumalanga Highveld with a view to assessing whether any damage has occurred to commercial forest plantations (Olbrich & Kruger 1990). Very small changes were observed in that study, although evidence was not conclusive; it was reported that commercial forests did not appear to have suffered any substantial ill-effects from acidic deposition. International experience suggests that a critical threshold may exist, up to which few if any damages are observed, and beyond which more extensive crop and forest damage may occur (Scholes 1995). In other words, the damage function would undergo a step change at this critical threshold. For economic purposes, future damages would then be discounted back to present day values, yielding estimates of marginal damage costs. Given that the Mpumalanga Highveld is an important location for commercial forestry in South Africa, any damages which result from acidification, could have significant economic effects. For the present, however, the state of information means that this impact has to be classified as a *Class Two* impact.

2.2.6 Air pollution from power generation: visibility impacts

Reduced visibility is one result of air pollution which attracts considerable public comment and criticism. It requires no scientific or technical expertise to observe `dirty' air, hence there is frequent comment in the press and popular literature regarding hazy atmospheric conditions (see for example Clarke 1991). Reduced visibility is a phenomenon which occurs fairly widely, both in cities - notably Cape Town, which has a reasonably serious brown haze problem (De Villiers & Dutkiewicz 1993) - and in rural areas.

As with other forms of air quality monitoring, there is no systematic monitoring of visibility conditions by an independent authority acting primarily in the interests of the public. Nor are there any standards or guidelines regarding `acceptable' levels of visibility reduction. Eskom, partly in response to criticisms directed against itself as a contributor to the poor visibility in the Mpumalanga Highveld, undertook some quantitative analysis of visibility conditions in the region (Turner 1994a). This analysis utilised the results collected in its pollution monitoring network, particularly the Elandsfontein station, located in the centre of the Highveld.

Results from this analysis showed that the mean visibility range for the entire period May 1985 to December 1993 was 76 km, with a seasonal high of about 105 km for January and a low of 60 km for September (ibid: 5, 11). Average results, however, are perhaps not the most meaningful, since aesthetic values are influenced more heavily by poor visibility occurrences than by long-term averages. On this basis, the visibility range fares poorly. Whilst the range was over 100 km on 10% of the days measured - corresponding to low levels of visible pollution - it was as low as 40 km on almost half the days, as low as 30 km on over 20% of the days, as low as 20 km on almost 10% of the days, and less than 10 km on 1.1% of days - that is, 4 days per year (ibid).[24] It is perhaps the latter figures which reinforce most strongly the common view that air quality is severely degraded in parts of Mpumalanga province.

Establishing that a visibility problem exists, however, is easier than quantifying the contributing factors. Visibility can be impaired by naturally-occurring phenomena such as high levels of water vapour (which becomes `mist' at high densities) and natural dust in the atmosphere, as well as by human-induced phenomena. The latter include emissions from industrial activity, household fuels, motor vehicles, vegetation fires, changes in land cover resulting from mining, agriculture, waste production, and so on.

To a greater or lesser extent, all of these sources are present in the Mpumalanga Highveld where seven of the nine operational coal power stations are located. Eskom's analysis of the diurnal patterns of visibility conditions, which do not correlate with observed emissions patterns from its power stations, concludes that emissions of particulates and sulphur dioxide by power stations `do not play a major role in regional visibility impairment' (ibid: 6). Rather, Eskom attributes poor visibility mainly to `low level and regional sources' which include smoke from biomass burning, smouldering coal dumps and surface dust. Interestingly, Eskom's results show that the visibility range improved by about 5% per year at its Elandsfontein site from 1985 to 1993, and this was attributed to the improved control of smouldering coal dumps and the closure of old power stations with lower chimney stacks, which together outweighed the effect of drought on the amount of biomass burning and surface dust.

The relatively high degree of uncertainty, coupled with the non-transferability of experience from elsewhere, means that visibility impacts fall into the *Class Two* category for purposes of this study.

2.2.7 Water quality impacts from power generation

Power stations are important agents in the non-agricultural water sector in South Africa. Not only are they a significant consumptive user of water, but they also return significant

[24] By comparison, some US states set the criteria level for `severe' visibility degradation at 16 km (Turner 1994a: 5).

quantities of water back to the environment. Over time, Eskom's water policy has changed: in response to increasing water scarcity, two of its newer power stations employ dry cooling towers which reduce the net quantity of water consumed. In addition, it has adopted stricter policies regarding the quality of water it returns to the environment.

Eskom reports that it strives towards a `zero effluent' water policy, which means that the quality of the water it returns to rivers and dams must be at least as good as the water it draws from those sources (Eskom 1995b: 20). Thus water which is used in the thermal fuel cycle, is recycled up to the point where it is too polluted, after which it is used for less demanding purposes. For example, water which is eventually no longer being used in the fuel cycle, is used on the ash dumps to reduce the amount of ash blown into the atmosphere. In 1994, Eskom reported that about 8% (17 083 megalitres) of the water it consumed was released to public streams, and of this 10 megalitres did not conform to quality standards set by the Department of Water Affairs and Forestry (ibid). This is a fraction of a percent of its water emissions.

This is not to say, however, that the coal power stations have no impact on water quality at all. Firstly, there are upstream impacts of coal mining on water quality, which have already been noted in an earlier section. Secondly, problems sometimes arise in the implementation of the zero effluent policy, such as where leakages occur, as described above, or during periods of high rainfall where storage dams cannot accomodate increased flow volumes. In 1994, six such incidents were reported at Eskom's power stations, and although permission was obtained for these releases from DWAF (Eskom 1995b: 20), their economic costs could feasibly have been non-zero.

A third impact which arises in some cases, is positive. The power stations require water of reasonably good quality, otherwise the rates of corrosion and weathering of plant and equipment are unacceptably high, and consequently, at some power stations, water has to purified before use in the power stations. Even after that water has been through the fuel cycle and it is returned to the receiving water body, it can be of higher quality than when it was withdrawn (Fedorsky 1995). Insofar as there is no compensation or rebate accruing to Eskom in respect of this improvement, a positive externality arises. However, there are no quantitative data available regarding these changes in water quality.[25]

The valuation of water quality impacts is complex, although not impossible. The general approach would be either to impute a shadow price for water which is no longer usable; this

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[25] Another positive externality would arise if dams constructed for pumped storage schemes produced recreational benefits. Eskom's pumped storage dams, however, are not available for public use.

price would reflect the opportunity cost of rendering that volume of water usable - based on the marginal cost of obtaining water of adequate quality from an alternative source. A second valuation approach which could be used would be to quantify the expenditure incurred by downstream users in returning that water to an adequate quality - so-called `defensive expenditure'. In some cases, valuation of water quality changes also include changes in the productivity of commercial fishing or recreational fishing markets. For example, in North America, water quality impacts often include negative impacts on recreational and commercial activities in rivers and the Great Lakes (Rowe et al 1994b: XVI-2). In South Africa, on the other hand, these do not represent significant activities in the proximity of power stations; the impacts of other water consumption sectors, most notably agriculture, dominate issues of water quality.

On balance, this category of environmental impacts is not considered to represent a significant external effect (either negative or positive) for South Africa's power stations. It is thus given *Class Three* status for purposes of this study. Of greater relevance in relation to power generation's impact on water resources, are the impacts of coal mining on water quality, and the pricing of water consumption in once-through power stations, both of which have been addressed in previous sections of this chapter.

2.2.8 Greenhouse gas emissions from power generation

Electricity generation, where it is based on coal-power, is unavoidably a significant source of greenhouse gas (GHG) emissions.[26] The principal GHGs are carbon dioxide (CO_2), methane, chlorofluorocarbons and nitrous oxides, the first two of which are most significant in South Africa. South Africa was responsible for about 1.2% of global GHG emissions in 1988, making it the eighteenth largest source in the world, and one of largest sources on a per capita basis (Van Horen & Simmonds 1995). It was also the largest source of GHGs in Africa, accounting for 15% of the continent's CO_2 emissions.

Significant quantities of carbon dioxide are emitted by the electricity generation sector, and smaller amounts of methane during coal mining. Eskom is the single largest source of GHGs in South Africa, which by its calculations, amounted to some 142.9 million tons of carbon

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[26] Briefly, GHGs are relevant insofar as they are widely believed to enhance the naturally-occurring greenhouse effect - in terms of which GHGs increase the ability of the earth's atmosphere to retain warmth. The balance of scientific opinion suggests that continued emission of GHGs at present rates will lead to global climate change, with variable, but often negative consequences in many regions of the world. For more details on recent developments in the international climate change debate, see Rowlands (1995a) and on the South African energy sector's contribution, see Van Horen & Simmonds (1995).

dioxide in 1994 (Eskom 1995b). The emissions by each of the coal power stations is shown in Table 3.12.

Table 3.12 Carbon dioxide emissions from Eskom's power stations in 1994
(Source: Eskom Generation group)

	Total CO_2 emissions (million tons)
Arnot	4.5
Duvha	20.6
Hendrina	12.7
Kendal	17.8
Kriel	14.3
Lethabo	18.2
Matimba	20.0
Matla	18.2
Tutuka	16.6
Total	142.9

In addition to emissions of carbon dioxide, the power sector is also responsible for a portion of methane emissions emanating from the coal mining sector - notably underground mines. For present purposes, however, these will not be quantified: although the global warming potential of methane far exceeds that of carbon dioxide, most policy attention and quantification exercises to-date have focused on reducing carbon emissions. Moreover, less than half of Eskom's coal is sourced from underground mines which emit most of the methane. The exclusion of methane emissions has the effect, therefore, of understating the potential damages attributable to electricity generation in South Africa.

There is a vast body of international literature which has sprung up around the climate change phenomenon, addressing both its physical and political-economic dimensions; it is impossible to summarise here, all aspects of this issue. Instead, a few salient points will be made in relation to the impacts of climate change on South Africa's electricity generation industry. Firstly, South Africa is unlikely to be faced with any binding commitments to reduce its emissions of GHGs in the near future. As a `developing country' - defined in terms of the Framework Convention on Climate Change (FCCC) - South Africa will not face the same targets for GHG reductions as the industrialised countries.[27] At present, the latter's (non-binding) targets are limited to the stabilisation of GHG emissions at 1990 levels, by the year 2000 (FCCC, Articles 4.2 (a) and (b)).

A second pertinent factor follows from the first: although the country will not have any immediate onerous obligations upon ratification, it is not improbable that it could face some intermediate targets further down the road, if differentiation is made within the `developing country' category, by virtue of its relatively high level of income and emissions (Rowlands 1995b). Thus, it is important that South African policy-making takes explicit account of the climate change issue, and adopts a proactive stance in international negotiations.

Thirdly, there is considerable uncertainty around the climate change issue in various of its dimensions. Clearly, this uncertainty serves the interests of many groups and nations (for example, the oil producing countries); so, it is important to establish the boundaries of this uncertainty. One of the key uncertainties at present is around the potential impacts of climate change on specific sub-regions, such as Southern Africa. This uncertainty, exacerbated by the extremely long time periods over which it might occur,[28] makes it extremely difficult to make assessments of the economic and social costs of possible climate change: which may include, for example, the costs of increased drought in the future, and more frequent occurrences of extreme weather events (storms, floods, droughts). Several attempts have been made internationally to estimate the range of damage costs which might result from climate change (for example, Cline 1992, Fankhauser 1992, 1995, Nordhaus 1991); not surprisingly, these have produced very different estimates and have attracted their share of criticism. This uncertainty presents special difficulties for valuation exercises.

What is much more certain, however, is that the climate change issue will not disappear from the international political economy in the near future. Given the prominent role of South Africa amongst developing countries, it is essential that the issue is not ignored. Although there is considerable uncertainty surrounding possible scenarios of damages, enough work has been done internationally to permit some quantification - provided the level of uncertainty is taken into account explicitly - hence this impact is assigned to the *Class One* category. The next chapter presents a range of estimates of the damage costs which might be attributable to emissions of greenhouse gases.

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[27] The future tense is used because, by the beginning of 1996, South Africa had signed, but not yet ratified the FCCC.

[28] With concomitant importance attached to the selection of a discount rate when economic effects are being considered.

2.3 Externalities in South Africa's nuclear fuel cycle

The nuclear fuel cycle internationally has been the subject of considerable analysis in the last two decades or more, particularly in relation to the environmental and other externalities with which it is associated. Many of the environmental impacts in the nuclear industry can be characterised as having relatively low probability but high (or potentially catastrophic) impact if they do occur. Thus externality studies have usually focused on the *risk* of such impacts, and on society's valuation of those risks (rather than valuation of the impacts themselves).

In addition to the externalities associated with potential catastrophic accidents, there are two other kinds of externalities of relevance. The first relates to those environmental impacts which occur regularly and are not fully internalised into the price of nuclear electricity - occupational health hazards, air or water pollution, and so on. These can be treated in essentially the same way as impacts arising from the coal fuel cycle.

Another kind of externality may be termed a `fiscal externality' - in other words, a fiscal transfer payment (for example, a subsidy) which is made to the nuclear industry, and which is not reflected in the price of nuclear electricity (Lockwood 1992). Whilst this is not an environmental externality *per se*, it is highly material in South Africa. Consequently, it is included in the scope of Chapters Three and Four, but in order to maintain the focus of the thesis on environmental issues, the fiscal externality is not aggregated with the environmental externalities in later discussions.

2.3.1 *Environmental impacts and risks in the nuclear fuel cycle*

Several actual or potential environmental impacts of significance occur in South Africa's nuclear sector:

- The risk of catastrophic accidents (on the scale of Chernobyl) cannot be ruled out completely, although it is reduced by the existence of a strict safety regime governing all stages of the nuclear industry in South Africa. Externalities studies elsewhere, in attempts to quantify these risks in economic terms, have focused on society's valuation of those risks (Krupnick et al 1993). In South Africa, there has been no such analysis to-date, which does not make this quantification exercise feasible for present purposes. However, the fact that Koeberg power station is located about 50 kilometres from the Cape Town metropolitan area, which has a population of over 2 million people, means that the impact of any accident, should one occur, would probably be very large.
- Although international experience with the decommissioning of nuclear power facilities is as yet limited, it has become evident that the associated costs are likely to be greater than previously anticipated, mainly because of higher safety and environmental

standards than were foreseen at the time of construction of those facilities (MacKerron 1992). This means that costs have to be incurred in the future which have not been accounted for in the calculation of the present value of streams of income and expenditure over the lifetime of nuclear facilities. Thus to the extent that these future costs are excluded from current prices, an externality exists. The same could hold true for South Africa's nuclear industry, although the absence of any public information about this makes any assessment in this regard necessarily speculative.

- Nuclear power generation produces a number of radioactive waste products, which are typically categorised according to their degree of radioactivity. Low level wastes are stored at the Vaalputs site in the Northern Cape province under high standards of safety, and thus their costs are essentially internalised. However, the storage and disposal of high level radioactive wastes remain problematic (as is the case internationally), and in the interim, these wastes are stored on site at Koeberg power station (Auf der Heyde 1993). Given the extremely long time periods for which high level wastes remain hazardous (several thousand years), the potential environmental and health costs associated with future accidents are, simultaneously, highly uncertain, potentially large, and long-lasting. Their valuation thus presents significant difficulties, not least because so much is dependant upon the choice of a discount rate.

- The operation of a nuclear power station is associated with any number of small incidents and procedural breakdowns which, because of the nature of the nuclear generation process, means that workers are exposed to potentially serious occupational risks. Although difficult to quantify, these risks are not accounted for in wage rates (as in the case of coal mining) which means the risks are not internalised. This is not to deny that the nuclear industry employs stringent health and safety standards for its workers; nonetheless, the reality is that workers are exposed to potentially significant risks by virtue of the nature of the process, and are not compensated for this. At the ANC's nuclear conference held in February 1994, worker representatives highlighted the health and safety hazards to which they were exposed - notwithstanding safety procedures (ANC & Alliance 1994: 235).

- Finally, to the extent that links existed between civilian nuclear power and military nuclear weaponry - links which were partly confirmed by the admission of former President de Klerk in 1993 that the apartheid government possessed at least six nuclear weapons - there exists another category of externalities which are highly uncertain, but so large as to be of significance. Clearly, were any such weapons to be utilised (although it is difficult to imagine where or how these might have been deployed) the costs could have been enormous.

These external environmental effects, both current and future, existing and potential, are of significance in aggregate, but present considerable difficulties for any economic analysis. Insufficient information exists in the South African context to quantify society's valuation of the risks with which these impacts are associated. This scarcity of data has not been helped by the pervasive secrecy which has surrounded (and continues to surround) the industry. Furthermore, given the limited resources available for this project, these externalities will not be further assessed, save to note that they are potentially highly significant, and are therefore assigned to the *Class Two* impact category.

2.3.2 *Fiscal externalities in the nuclear industry*

A final category of externalities which arises in the South African electricity generation industry are those subsidies which have flowed to the nuclear industry since the beginning of the 1970s. As noted earlier, although this is a fiscal rather than an environmental externality, it is highly significant in relation to the amount of electricity which has been produced, and its existence results in a material divergence between the private and social costs of nuclear electricity.[29] This style of analysis is similar to the nuclear industry in the UK, where fiscal externalities were also quantified (Lockwood 1992).

Unlike the coal-fired electricity sector, which has historically received little or no financial subsidy from public funds, the local nuclear industry has enjoyed a very privileged position in this respect. The nuclear sector has received the lion's share of the Department of Mineral and Energy Affairs (DMEA) annual parliamentary grant since 1971/72. Table 3.13 shows the average annual allocation to the nuclear industry: the Atomic Energy Corporation (AEC, or its predecessor, the Atomic Energy Board), the former Nuclear Development Corporation (Nucor) and the Council for Nuclear Safety (CNS), for the period 1971/72 to 1995/96.

It is evident from the table that considerable resources have been directed to the local nuclear industry; these have been directed to three main categories of expenditure: capital expenditure, operating expenditure and servicing and repayment of loans (ibid: 7). Arguably, not all of these amounts should be attributed to the nuclear generation industry, since non-electricity aspects of the industry have also benefited from state subsidies: notably the nuclear bomb programme, research and development in non-electric areas, and the production of medical isotopes. Unfortunately there is no publicly-available information on the allocation of the subsidy to these various sectors; consequently, calculations and assumptions have to be made on the basis of the information which is available.

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[29] In order to maintain the focus of the quantification exercise on environmental and resource issues, the fiscal externality is excluded from the summary tables in Chapter Seven where comparisons are made of environmental externalities in various sectors.

Table 3.13 Subsidies to the nuclear industry in nominal and real terms, 1971/72 to 1995/96
(calculated from Auf der Heyde 1993, CSS 1995)

Period	Annual average (Nominal Rm)	% of DMEA budget	Annual average (1995 Rm)[30]
1971/72 to 1975/76	21.2	32.0%	250.6
1976/77 to 1980/81	110.7	40.8%	740.7
1981/82 to 1985/86	363.0	70.3%	1 336.5
1986/87 to 1990/91	686.2	79.7%	1 403.5
1991/92 to 1995/96	524.5	70.3%	619.3
Total 1971/72 to 1995/96	8 528.0	69.3%	21 753.3

To be conservative, only those costs which are known to be related to electricity generation will be included in this analysis; thus these represent a *minimum* estimate of the fiscal subsidy to the industry, and probably under-estimate the actual costs. On the operating expenditure side, the relevant costs include those of conversion, enrichment, fabrication, decommissioning and waste disposal: according to Auf der Heyde (1993: 25), these totalled R3 062 million (1995 Rands) for the 9 fiscal years from 1987/88 to 1995/96, which represents 46% of total operating expenditure, or 37% of total nuclear funding (calculated from Auf der Heyde (1993: 5,7).

With regard to capital expenditure, the costs of the conversion, enrichment and fabrication plants (but not the so-called `Y-plant') are assumed to relate to electricity generation; these allocations amounted to R2 686 million in 1995 Rands, or 62% of the total capital expenditure reported in Auf der Heyde (1993: 24). Correspondingly, it is assumed that the same proportion (62%) of finance charges is attributable to the electricity component of the subsidy; this amounts to R1 620 million (in 1995 Rands) for the nine years to 1995/96.

The total of these allocations for the nine year period 1987/88 to 1995/96 is R7 368 million, in 1995 Rands. A portion of the nuclear allocation prior to 1987/88 should also be apportioned to the nuclear electricity sector, but since this information is not publicly available, further assumptions are required. It will be assumed (conservatively) that all relevant capital expenditure is included in the above estimate (and likewise for corresponding finance

[30] Nominal Rands adjusted to real 1995 Rands by Production Price Index, using CSS (1995).

charges); for operating expenditure, it will be assumed that the percentage of total nuclear funding attributable to the electricity sector, was the same as after 1987/88: that is, 37% of the total. This yields an estimate of R4 876 million in 1995 Rands (calculated from Auf der Heyde (1993: 5). To this should be added the allocation to the Council for Nuclear Safety (CNS), which also represents an external cost of operating a nuclear power facility: this amounted to R54 million in 1995 Rands, for the period 1983/84 to 1995/96.

The above estimates are summarised in Table 3.14.

In summary, it is estimated that the total public allocation to the nuclear industry, for purposes of electricity generation, was R12 298 million (in 1995 Rands) for the period 1971/72 to 1995/96. This represents 57% of the total government subsidy of the nuclear industry for that period, and can be taken as a minimum estimate, in the absence of full public disclosure of this information.

Table 3.14 Estimate of nuclear subsidy attributable to electricity generation (Rm 1995)

	1987/88 to 1995/96 (Rm)	% of category[31]	1971/72 to 1986/87 (Rm)	% of category	Total
Operating exp.	3 062	37%	4 876	37%	7 938
Capital exp.	2 686	62%	0	0%	2 686
Finance charges	1 620	62%	0	0%	1 620
CNS	54	100%	0	0%	54
Total	7 422	-	4 876	-	12 298

This subsidy has not been internalised into the price of nuclear electricity - the effects of doing so are described in the next chapter. Given that there is sufficient information at this aggregated level to quantify the extent of the fiscal externality, it is accorded a *Class One* ranking.

[31] The denominator here is the total allocation to each category (respectively, operating expenditure, capital expenditure, finance charges and CNS).

2.4 Summary of external effects

Thus far, a range of potentially significant external effects in the South African coal and nuclear fuel cycles have been described, in terms of their scale and frequency. These are summarised in Table 3.15; it is evident that five main externalities - those falling into Class One - are quantified in this study, four of which are environmental impacts *per se*.

Table 3.15 Summary of potentially significant environmental impacts and their classification in this study

	Class One	Class Two	Class Three
Coal fuel cycle			
Coal mining: morbidity & mortality	✓	✓	
Coal mining: air & water pollution		✓	✓
Generation: water consumption	✓		
Generation: air pollution & health impacts	✓	✓	
Generation: air pollution & acidification		✓	
Generation: air pollution & visibility		✓	
Generation: water quality impacts			✓
Generation: greenhouse gas emissions	✓		
Other impacts (EMFs, aesthetics, etc.)			✓
Nuclear fuel cycle			
Fiscal subsidy	✓		
Other impacts (risk of accident, waste disposal, decommissioning costs, etc.)		✓	

Chapter Four

The external costs of electricity generation

In this chapter, an economic quantification is undertaken for the five Class One externalities described above. The valuation results are expressed in three forms: as total Rand values, in mills per kilowatt-hour of electricity produced,[1] and in cents per joule of energy. In the final chapter of the thesis, these valuations will be compared with the externalities in the household sector. The next section addresses further theoretical and methodological questions regarding the valuation approaches adopted, particularly in relation to morbidity and mortality effects.

1. Valuation of morbidity and mortality: further methodological issues

In the review of the literature in Chapter Two, it was noted that there are a number of valuation approaches, both direct (such as CVM and hedonic pricing) and indirect (such as opportunity cost approaches). It is important for purposes of this chapter, that further consideration is given to the valuation methods to be used, especially in relation to morbidity (ill-health) and mortality (death).

1.1 Valuation of morbidity

Two broad methods can be used in valuing morbidity effects: first, those based on individual preferences, that is, willingness to pay (WTP) for environmental and health improvements, or willingness to accept compensation (WAC) for deterioration, and second, those methods based on resource or opportunity costs (Freeman 1993: 343). In the neo-classical literature, the individual preference approach is generally preferred because it is consistent with the underlying basis of microeconomic theory, in which utility or welfare is a function of a person's willingness to pay for it. Furthermore, WTP (or WAC) approaches are generally considered more appropriate because they are believed to yield a more complete estimate of a person's valuation of illness: WTP values include, typically, not only the opportunity costs of illness, but also a person's valuation of the discomfort or displeasure from being less than fully healthy. By contrast, it is argued that opportunity cost approaches focus only on the direct costs of illness - such as expenditure on medical care and foregone income from not working - and do not place any value on the more subjective (but very real) discomfort or displeasure experienced whilst being sick, which should probably have a non-

[1] 1 mill equals R0.001, that is, a tenth of a cent.

zero value (ibid). Empirical data in the US suggest that WTP often exceeds the direct cost of illness by a considerable amount, depending on the health effect: according to Rowe et al (1994b: X-30), by a factor of 1.3 to 2.4, and according to Cropper and Oates (1992: 715) by a factor of 3 to 4.

Individual preference approaches for valuing morbidity are, however, not necessarily the ideal method in the South African context. There are several reasons for this: firstly, there have been few if any studies in this country of people's WTP or WAC with respect to health effects (or any other environmental element, for that matter). Thus the empirical basis for such estimates is weak. Secondly, even if such information was available, it would be problematic to use it in a context of an extremely unequal distribution of income: all other things equal, the WTP for health improvements of a poor person would be lower than that of a wealthy person. Using such WTP valuations may be consistent with economic efficiency criteria (as embodied in the Pareto criterion), but could have consequences for policy decisions which are extremely inequitable.[2] If an equitable outcome is desired, as it is in South Africa, it therefore becomes necessary to make adjustments to the valuation results, giving greater weight to the valuations of targeted (poor) individuals or groups. As noted in Chapter Two, this is the suggested approach in theory, although practical difficulties mean that it is seldom applied in reality.

Thirdly, WTP and WAC valuation approaches may, in fact, understate the value of a health risk - the inverse of the commonly prevailing situation in wealthier countries (as embodied in the ratio of WTP to cost of illness reported above). To illustrate, it could happen that society's valuation of a person's health, as expressed for instance in its budgetary allocation to public health care in its hospitals and clinics, exceeds the amount which a person would be willing to expend on medical care - for the simple reason that their income may be insufficient to allow them to pay. Thus, for example, it is possible to foresee a situation in which a cancer patient is unable (`unwilling') to pay for expensive radiotherapy treatment (and would therefore have a low WTP were this to be measured), but where society deems it worth expending the required amount to provide that person with the best treatment. In this context, a valuation approach which is based on WTP might give an assessment of an individual's willingness to pay (determined as it is, by income), which is less than the actual

[2] For instance, all other things equal, it would be economically efficient - but not necessarily equitable or socially acceptable - to construct a new power station near a poorer residential area rather than a wealthier area, since the latter's valuation of clean air would be higher than the former's.

amount expended by society - which is also determined by its income, including that of relatively wealthier persons.

Consequently, the valuation approach adopted in this study in respect of morbidity effects is based on the opportunity cost approach. The valuation of health effects therefore includes actual expenditure on health care (both public and private), transport costs, medication and so on, and foregone income, such as lost time at work. This cost of illness approach has the added advantage of being less abstract and easier to measure in practice. This is not to say, however, that it will not also embody biases (including possibly against the poor), because it depends (inter alia) upon the ethics and altruistic tendencies of society and its system of government. Nonetheless, in the context such as that prevailing in the newly-democratic South Africa, the bias against the poor is likely to be lower than under a conventional WTP approach.

In collecting the necessary data, surprisingly little information about typical health costs was found to have been published or reported in the literature.[3] Consequently, the opportunity cost data were derived specifically for this thesis and were generally elicited from several health specialists working in the field. Their estimates of typical treatment regimes and costs were used, together with other secondary sources of information wherever these existed. The main opportunity cost estimates are summarised in Appendix 2; these are applied to the valuation of physical health outcomes in both Chapter Four and Chapter Six.

1.2 Valuation of mortality

Turning to the valuation of premature death, the complexities and controversies are significant. Not least of all, is the ethical problem which arises in reducing human life to a finite monetary value, and the implications this holds for policy-making. The economics literature frequently makes a distinction between the valuation of human life *per se*, and the value individuals or society place on the statistical risk or probability of early mortality (Freeman 1993, Pearce et al 1991: 5). This argument points out that there is a difference between simply placing a value on an individual's life, and placing a monetary estimate on a person's trade-off between various risks of death. This distinction does not always appear to satisfy opponents of valuation; however, more compelling perhaps, is the argument first presented in Chapter Two: namely that in many respects, valuations are made already by individuals and social groups, implicitly in many of their activities, often without making the trade-offs explicit. Such a trade-off is evident, for example, in the allocative decision

[3] This was especially surprising given the fundamental re-assessments underway at the national policy level, of the efficacy of public spending in the primary, secondary and tertiary medical sectors.

between preventative primary health care and high-level health care, such as heart transplants. Whilst the allocation of health-care resources illustrates an implicit valuation of mortality by one social group (represented by the government of the day), this is not to say that it is necessarily an optimal valuation; this is particularly evident in the context of apartheid government, where black people had no influence on the valuation and allocation decisions of the ruling class.

Thus to assign some monetary value to a statistical life merely makes transparent or explicit whatever judgements are being made. Furthermore, provided the values are used in a decision-making context which seeks to balance the full range of interests as well as possible, the use of monetary values for early death can serve an important strategic purpose: for example, by highlighting the losses suffered by society due to inadequate supplies of potable water and sanitation services, a case can be made, perhaps more strongly, for investment in improved service levels.

The economic valuation of early death should also not be seen or used out of context: the placing of monetary values on death and other environmental elements, is used merely as one element in decision-making, along with a range of other inputs: social, political and ecological.

As in the case of morbidity effects, there are a number of methods which may be adopted to value premature death. Whereas the cost of illness approach was used for morbidity effects, the equivalent does not exist for mortality. One approach which is sometimes used, is the `human capital' approach: essentially, this entails valuing a lost life at the discounted value of future income which that person might have been expected to generate (Jones-Lee 1994: 298). Most simply, average per capita GDP would be used as a proxy for that person's income; this was done, for instance, in a study of external electricity costs by Dutkiewicz and de Villiers (1993), as well as in a study of the costs of cardiovascular disease by Pestana (1993). Many economists are quick to point out, however, that this approach is seriously flawed and has little basis in welfare economic theory:

- If there is social differentiation, for example between classes, age groups, males and females, employed and unemployed, then different values will be given to these groups, on the basis of their `capital value' (Jones-Lee 1994: 299). In South Africa's case, income is very unequally distributed, with the richest 10% of the population earning an estimated 51.2% of total income, and the poorest 40% earning only 3.9% (SAIRR 1995: 377); in this context, the human capital approach would yield massive differences in valuations. Indeed, it is possible that an unemployed or disabled person would have a negative `value', or put differently, that his or her death would have a positive external value. By the same token, in most societies, adult males of about 25 years of age will have the highest `value' (Freeman 1993: 324). Not only is this kind of outcome highly inequitable, it is demonstrably not consistent with the valuation of human lives demonstrated by the willingness, for instance, of society to pay for the care and support of unemployed or disabled people. This willingness is expressed indirectly through the mechanisms offered by representative government in its budgetary process.

- Secondly, the human capital approach is highly sensitive to the choice of a discount rate: for example, for a male child between 1 and 4 years of age in the US in 1987, at a discount rate of 2.5%, its human capital value would have been $761 000, compared to only $60 000 at a discount rate of 10% (ibid: 325). Thus any impression created by this approach that it is scientific or rational is somewhat illusory - it remains highly sensitive to choices about variables such as the discount rate, and these choices depend heavily on the judgement of analysts.
- On a global scale, this approach can yield extremely inequitable results, given wide divergences in levels of income and wealth. To take an extreme example:[4] assuming the French government places a value of *x* on the results of nuclear weapon tests, then it could be economically efficient (defined in the Pareto sense) for the government to test its nuclear bombs on a subsistence-based island economy in the Pacific Ocean, as long as *x* exceeds the valuation of consequent deaths and injuries amongst local inhabitants. This would be the case even if it were to compensate victims or their next-of-kin for foregone earnings (if the Hicks-Kaldor criterion is relaxed). Depending on the value of *x*, however, the same might not be true if the nuclear bombs were tested in rural France or Germany where income levels are much higher. Clearly, the implications of using this kind of valuation approach could yield outcomes in cost-benefit calculations which are difficult to countenance in the face of social justice and ethical considerations.

An alternative valuation approach used for mortality entails the use of individual preference approaches: not so much a person's willingness to pay to avoid death or willingness to accept compensation for death, but the valuation of a changed probability or risk of death (Jones-Lee 1994: 300). Such decisions are made on a daily basis: for example, in paying a higher price for a ticket with an airline or bus service which is considered safer than its competitors, or in the decision to install and wear safety belts in motor vehicles. In the context where valuations of saved lives are to be included in a cost-benefit analysis, it is possible also to make adjustments to the values based on a social welfare function, if the starting income distribution is not accepted as optimal. Other valuation methods discussed in the literature include those based on life assurance values, court awards, implicit public

[4] This example may be extreme, but it is not implausible - as recent events have shown.

sector valuations and value of time methods; however, all of these have been dismissed as `seriously deficient' (Jones-Lee 1994: 303-306).

The choice of values of premature death for South Africa is made difficult by two factors: first, there have been no studies of revealed preferences in this country from which values can be derived. Secondly, as noted already, there are sharp inequalities in the distribution of income and wealth, which presents problems (from an equity perspective) if differential valuations are used for different income groups - as would apply if the human capital approach was adopted.

Taking each of these issues into account, the valuation approach adopted in this study for mortality effects will draw on international studies of revealed preference, adjusted for South African income levels. At least two major international externality valuation exercises have undertaken their own reviews of the literature and on that basis, selected a range of values for premature deaths. These estimates, which are based on revealed preference approaches, are shown in Table 4.1. The study by Rowe et al (1994b) was undertaken for New York state, USA, and drew upon North American valuation studies; likewise, the study by ETSU (1995) estimated average values for the European Union. Both of these valuation ranges are similar to the valuations suggested by Jones-Lee (1994: 313) for the United Kingdom.

Simply applying these foreign valuations to South African conditions would clearly be problematic from a theoretical perspective, since individual valuations of the risk of death must, by definition, take account of income levels. Assuming these valuations vary in direct proportion to income, an adjustment can be made to the North American and European values to reflect average South African income levels.[5] These adjustment factors are also shown in Table 4.1.

The figures in the bottom right cell of Table 4.1 represent the average of the adjusted valuations for the American and European studies in the last row of Table 4.1. If these numbers are rounded down to the nearest R100 000, the following valuations for premature deaths are derived:

[5] In effect, this means an income elasticity of one has been assumed. It might be expected that at either extreme of the income distribution, elasticity would be less than one, whilst in the middle of the distribution, the function would be relatively income elastic. In the absence of any data to the contrary, it seems reasonable to assume an elasticity of one.

low estimate	R800 000
central estimate	R1 200 000
high estimate	R1 900 000.

Table 4.1 Valuations of premature deaths used in international studies and in this study

	USA	European Union	South Africa
GDP per capita ($ 1992)	23 240 *(World Bank 1994: 163)*	19 678 *(calculated from World Bank 1994: 163)*	2 193[6]
Income adjustment factor	10.6	9.0	1
Mortality valuations low estimate central estimate high estimate	 $1 700 000 $3 300 000 $6 600 000 *Rowe et al (1994b: X-14)*	 ecu 2 100 000 ecu 2 600 000 ecu 3 000 000 *ETSU (1995: 49)*	 - - -
Income-adjusted valuations (1995 R)[7] low estimate central estimate high estimate	 586 981 1 139 434 2 278 868	 1 115 333 1 380 889 1 593 333	 Average: 851 157 1 260 162 1 936 101

These estimates will be used in this study for purposes of attaching an economic value to premature mortality. This kind of `benefit transfer' is a highly controversial area in applied welfare economics, particularly where the results of such analyses have trans-boundary implications. The most acute such instance arises in the case of damages due to global warming, where differential valuations of premature mortality could have inequitable policy implications - compare, for example, the effects of lost lives caused by sea level rise in

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[6] Calculated as follows: GDP for 1992 of R309 085 million (Central Statistical Services 1994b: 21-6), divided by population estimate for 1992 of 38.5 million (1994 estimate of 40.4 million, reduced by annual growth rate of 2.44%) (ibid: 1-3).

[7] The US and EU valuations are divided by the income adjustment factor to give South African valuations. The following exchange rates are used: $1 = R3.66, ecu 1 = R4.78.

Netherlands, with Bangladesh. This area remains the subject of vociferous debate. Fortunately, the present analysis avoids the worst of these problems because of the fact that the valuation set is being applied only to the population in Southern Africa; nonetheless, it is important to highlight the methodological and ethical complexities with which this method can be associated in other contexts.[8]

A final consideration for present purposes concerns the possible differentiation of valuations depending on variables such as age, gender, race, and location. Whilst the use of different valuations might yield results which more accurately reflect the influence of these variables on the economic standing of individuals, this is not done in this thesis, for the following reasons. Firstly, the empirical basis for introducing a wider range of valuations for different social groups, is weak. The only method would involve adjusting the above estimates based on average incomes for each group being identified (in the same way as this was done in Table 4.1 when deriving valuations applicable to South Africa). However, this would introduce the same problems as are attributable to the human capital valuation approach: particularly the ethical and equity considerations arising in policy decisions encompassing different social groups.[9]

Secondly, the modelling of health outcomes does not permit differentiation between, for example, employed and unemployed victims of pollution. Thus it would not be possible to apply different valuation sets for the range of externalities being considered in this thesis. Finally, the marginal increase in `accuracy' which might be achieved through further disaggregation of the above valuation sets, would require a disproportionately large increase in data collection and modelling sophistication. In the face of the other uncertainties in the analysis, this does not seem to be justified. The final chapter of the thesis will return to this issue in a critical assessment of the results and methodologies employed.

To summarise, this section has suggested that the appropriate valuation method for mortality is the `benefit transfer' approach, whereby international WTP valuations have been

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[8] Of course, even in this analysis, there are cross-border considerations. Consider an investor in an electricity-intensive industry, comparing South Africa with another possible site. All other things equal, if the value of a lost statistical life in the other country is higher, then the investment would be made in South Africa, with this country therefore bearing any attendant negative environmental effects.

[9] Because such comparisons are not being made on an international scale in this study (for example, between South African externalities and those elsewhere), the equity issue does not arise in adjusting international valuations to reflect different income levels in South Africa, as was done in Table 4.1.

adjusted for use in South Africa on the basis of income differentials. In the previous section, by contrast, the `cost of illness' or opportunity cost approach was proposed for morbidity effects. This apparent inconsistency is explained by the fact that there are no South Africa-specific WTP data for morbidity, whereas there are opportunity cost data. The latter are considered more appropriate and accurate than any adjusted WTP values might be.

2. Valuation of external costs in South Africa

In this section, valuations are derived for each of the five Class One externalities which were described in Chapter Four.

2.1 Valuation of morbidity and mortality on coal mines

It was estimated in Chapter Four that the average fatality rate of coal miners supplying the coal power stations was 0.156 per thousand GWh produced, equivalent to 23 deaths in 1994. The valuation of these deaths is shown in Table 4.2 below.[10]

Table 4.2 Valuation estimates for mortalities on coal mines, 1994, in total and in mills/kWh

	Low estimate	Central estimate	High estimate
Total (Rm)	18.4	27.6	43.7
Total in mills/kWh	0.12	0.19	0.30

With respect to injuries on the mines, costs include the following main components: costs of medical treatment, opportunity costs of not working, and compensation costs paid to injured workers. Other costs, such as travel costs are not included in this calculation because, although relevant, they are small in relation to the other costs involved, and the fact that some of these costs have to be estimated with no reference to empirical data, does not warrant their inclusion.

Table 4.3 summarises the range of estimated opportunity costs for injuries. The medical costs will vary widely from case to case and few empirical statistics are available regarding these costs; consequently a range of estimates will be used, of R2 000, R5 000 and R8 000 per

[10] In this calculation, and others to follow, the unit cost in mills per kWh is based on a denominator of 148 003 GWh - the total of coal-fired generation in 1994 (see Table 3.3).

injury.[11] For lost productivity, the number of days away from work (4, 8 and 12 weeks as discussed in Chapter Three) are valued at the average wage rate applicable on the coal mines. In 1993 this was R2 794 per month (CSS 1994a: 4-16), which, if adjusted by 10% per annum to 1994 levels, equates to R3073 per month. Finally, compensation payments are based on the schedule of benefits stipulated in the Compensation for Occupational Injuries and Diseases Act, 1993 (Schedule 4). These payments are payable for temporary disablement, at a rate of 75% of the worker's monthly remuneration (up to a maximum of R6 669 per month). However, because this compensation is paid out of an insurance fund, which is financed by mining companies, it can safely be assumed that these costs are already internalised - thus they are not included in the present calculation.

The injury statistics described in Chapter Four suggested that there would have been 131 injuries on the coal mines supplying Eskom in 1994. The results of applying the above cost estimates to this number of injuries are shown in Table 4.3.

Table 4.3 Valuation estimates for injuries on coal mines, 1994, in total and mills/kWh

	Low estimate	Central estimate	High estimate
Medical costs	R262 000	R655 000	R1 048 000
Lost productivity	R402 536	R805 126	R1 207 689
Total cost	R664 536	R1 460 126	R2 255 689
Cost in mills/kWh	0.005	0.010	0.015

The valuation of these impacts does not yield amounts of great significance when expressed as a cost per unit of electricity generated, even though the impacts can be significant at the local scale where they occur.

It should be noted that an important category of occupational morbidity has not been included in the above valuation estimates, namely the occurrence of respiratory and other diseases resulting from prolonged exposure of coal miners to high levels of dust and other airborne particulate matter. This externality was classified in Chapter Three as a Class Two impact, because although its extent and economic value is considered to be significant,

[11] These amounts were estimated based on discussions with the Government Mining Engineer.

insufficient data exist at present to quantify it. Thus the effect of this omission will be to understate actual externality values.

2.2 Valuation of water consumption in power stations

A review of the national pricing policy for bulk water supply was commenced in 1995, with a view to developing a policy which provides adequate signals to water consumers regarding the economic cost of water (DWAF 1995). By the end of 1995, no firm estimates had been made regarding marginal costing scenarios for water supply, although several values have been presented. One benchmark which is informing the policy debate, is the cost of supplying water to the Highveld from the Lesotho Highlands Water Scheme. Again, some uncertainty exists regarding the economic cost of that water, especially because the feasibility studies are several years old; however, a value of R1.50 per m^3 is currently used as a benchmark value for the long-run marginal cost of supply (Roome 1995). Some estimates of the economic cost of supplying additional water on the Highveld are higher, at about R3.00 per m^3, although since there has been little analysis to underpin those estimates, they will be discarded as outliers for present purposes (ibid).

For purposes of this study, the central estimate of the economic value of water supplies by the power stations is R1.50 per m^3, with low and high values of R1.20 and R1.80 respectively, which encompass the range of estimates used at present.[12] The external costs implicit in these water prices, defined in terms of the divergence between the private and social costs of water, and expressed in terms of Rands per m^3 consumed, total Rands and mills/kWh of electricity generated, are shown in Table 4.4. The external costs are calculated with reference to the data in Table 3.7 in Chapter Three. To illustrate, in the case of Arnot power station, the current water price is R0.55 per m^3 (from Table 3.7). This is R0.95 below the central estimate of R1.50/m^3, and at the rate of water consumption for 1994 of 8.875 million m^3 (also from Table 3.7), represents an external cost of R8.4 million for the year.

[12] Based on discussions with Klaus Triebel (Department of Water Affairs and Forestry).

Table 4.4 Valuation estimates for water consumption external effects

	Low estimate		Central estimate		High estimate	
	R/m^3	Rm	R/m^3	Rm	R/m^3	Rm
Arnot	0.65	5.8	0.95	8.4	1.25	11.1
Duvha	0.61	25.0	0.91	37.3	1.21	49.6
Hendrina	0.80	21.8	1.10	29.9	1.40	38.1
Kendal	0.00	0.0	0.00	0.0	0.02	0.1
Kriel	0.41	13.5	0.71	23.4	1.01	33.2
Lethabo	1.08	32.7	1.38	43.6	1.68	53.1
Matimba	0.70	3.0	1.00	4.2	1.30	5.5
Matla	0.00	0.0	0.27	9.4	0.57	19.8
Tutuka	0.54	19.0	0.84	29.6	1.14	40.2
Total value	-	120.8	-	185.8	-	250.7
Average mills/kWh		0.82		1.26		1.69

The central estimate for the above calculation is that the externality - the under-pricing of water consumption - based on a marginal economic cost of R1.50 per m^3 of water, amounts to 1.26 mills per kWh or 0.13 c/kWh on average. As noted in Chapter Three, water costs represent less than 2% of Eskom's operating costs (excluding depreciation and finance charges) so that even a large increase in the average price of water (127% in the case of the central estimate) would have a relatively small impact on electricity prices. This is discussed more fully later in this chapter.

2.3 Valuation of health effects of power station air pollution emissions

The valuation of health effects resulting from air pollution emissions is the final step in the impact pathway analysis. As noted at the beginning of this chapter, the valuation approach adopted in this study is one based on the opportunity cost approach (as opposed to, for example, the willingness to pay method).

Data exist regarding each of the main building blocks in the impact pathway - air pollution emissions, wind patterns, demographics, dose-response functions, and valuation data. To bring each of these together is a complex task and to this end, a computer modelling tool (named EXMOD) has been utilised. EXMOD is described in more detail in Appendix 3 but, briefly, it was developed over the period 1993 to early-1995 for the New York State

Environmental Externalities Cost Study, with support from the Empire State Electric Energy Research Corporation, the New York State Energy Research and Development Authority, the New York Department of Public Service and the Electric Power Research Institute. The aim of that study was to develop a user-friendly damage function tool with which impacts of new or relicensed electricity supply and demand management options could be evaluated. The work in the New York project was undertaken collaboratively by the Tellus Institute, Boston, and RCG/Hagler Bailly, Colorado and was summarised in four comprehensive volumes (Rowe et al 1994a, 1994b, Bernow et al 1995a, 1995b),[13] as well as a shorter paper (Rowe et al 1995). Following a review of international externalities studies, and particularly of models which could be of potential use in the South African context, EXMOD was investigated and selected for use in this study.

Briefly, the approach adopted in valuing health impacts was as follows. First, technical data about Eskom's nine coal power stations were collected and input into EXMOD; these data are summarised in Appendix 1, Tables A1 and A2. Data about fuel type and composition were also included (refer Table 3.8). Second, demographic data were collected and arranged into a database file for each magisterial district in South Africa, from 1991 census data, including total population, age distribution, land area, geographical co-ordinates and average altitude of the district. Aggregated data were also obtained for the following neighbouring countries: Lesotho, Swaziland, Mozambique, Zimbabwe, Botswana and Namibia. Third, long term surface wind data were obtained for fifteen monitoring stations in South Africa (and two in Namibia) and this was also entered into the appropriate database format.[14] Fourth, emissions data were obtained from Eskom for each of its nine operational power stations for the main air pollutants: particulates, sulphur dioxide and nitrogen oxides (refer Table 4.5). Fifth, dose-response data from North America were used, as described in Tables 3.10 and 3.11, adjusted to reflect a higher degree of uncertainty in South Africa.

Using its air quality dispersion models, EXMOD was designed to take this data into account following the damage function methodology, and to calculate the resultant damages, in physical terms (number of illnesses) and monetary terms (total damage costs or costs per kWh).[15] Consequently, the model was run nine times, corresponding to data sets for each of

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[13] All of these reports will be published in a two volume book by Oceana Press, New York state, in 1996.

[14] Whilst this data is a good description of long-term wind patterns, it does not necessarily describe well the particular atmospheric conditions around the power stations - refer to the discussion of uncertainties in Chapter Seven.

[15] Refer to Appendix 3 for more details.

the nine operational coal power stations. It should be noted that EXMOD includes default US data for every component of the impact pathway, as well as facility data and demographics; all of this was replaced with South African data, except for the dose-response functions and the atmospheric models - both of these issues were discussed earlier. Consequently, there is a reasonably good data set underlying this analysis.

The results of the model computations are summarised below. Table 4.5 shows the total physical health impacts, both in terms of morbidity and mortality, of air pollution emissions from all nine coal power stations.

Table 4.5 Physical health effects resulting from power station air emissions, 1994
(Source: EXMOD computations)

Health outcome	Unit	Low estimate	Central estimate	High estimate
Asthma attack	Occurrence-day	525,900	1,405,300	1,920,100
Acute bronchitis	Person	3,535	6,927	11,292
Chronic bronchitis	Person	430	1,060	1,634
Outpatient/GP visit	Visit	2,477	6,044	8,862
Mortality	Death	56	174	266
Resp. symptom day	Occurrence-day	2,834,700	5,479,700	7,910,700
Resp. hospital adm.	Admission	360	672	962
Restricted activity	Occurrence-day	443,300	1,005,900	1,534,500

Several observations can be made regarding these physical health outcomes. Firstly, there is a reasonably wide range between the three estimates: this is appropriate given the uncertainty inherent in the modelling exercise. Secondly, the incidence of some of these health outcomes is relatively high: notably, asthma attacks, respiratory symptom days and days of restricted activity. Asthma attacks are defined as a notable increase in asthma symptoms, such as shortness of breath, wheezing and use of more medication than normal. A relatively high proportion of the South African population - some 15% - suffers from asthma, which makes the results in Table 4.5 plausible.[16] In the case of `respiratory symptom days' the definition includes relatively minor symptoms of the respiratory system - coughs,

[16] This estimate comes from Terblanche (1995); in the New York state study referred to here, by comparison, the incidence was just under 5% (Bernow et al 1995).

wide range of estimates has been made, and not surprisingly, there is disagreement over a number of assumptions in these estimates. One variable alone - the discount rate - clearly has a fundamental influence on the valuation results because of the long time frames over which economic impacts are likely to occur.

One of the most recent estimates has been made by Working Group 3 of the Intergovernmental Panel on Climate Change (IPCC), the body of leading social and natural scientists informing the international negotiation process. The analysis undertaken by this group has suggested that the annual costs of global warming will be in the region of 1.5% to 2% of gross world product (GWP) by the time CO_2 concentrations reach double their natural levels - somewhere around 2050 or 2060 on the basis of current trends.[19]

By relating global damage costs to current emissions of GHGs, it is possible to estimate a cost per ton of GHG, or more particularly, CO_2 emissions. Corresponding with the high level of uncertainty over future damage costs, there is an equally wide range of `per ton' damage cost scenarios. The IPCC, in its 1995 Second Assessment Report, `does not endorse any particular range of values for the marginal damage of CO_2 emissions', but instead referred to published estimates which fall into the range of $5 to $125 per ton of carbon (IPCC 1995). One such estimate, by Fankhauser and Pearce (1993), both of whom were centrally involved in this aspect of the IPCC Working Group III's work, amounted to 14 Pounds per ton of carbon, equivalent to R80 or $22 per ton (reported in Pearce 1995a: 31).[20] This value will be used as the central estimate in this thesis. For the low estimate in this study, the lowest value referred to in the IPCC report will be used: $5 per ton, equivalent to R18 per ton. The choice of a high value is made more complicated by the wide range of estimates which have been published. The high value referred to by the IPCC, of $125 per ton, will not be used here since it is far in excess of most valuations which have been published to-date (see, for example, Rowe et al 1994b: XV-2, Nordhaus 1991) which are mostly in the range of $5 to $20 per ton of carbon. For present purposes, the high estimate of damages will be taken as being double the central estimate, that is, R160 per ton (or nearly $44 per ton) of carbon.

These values all refer to tons of carbon; since Eskom emission data refers to carbon dioxide, it is necessary to convert these valuations into values per ton of carbon dioxide. Based on the molecular weight ratio of 12/44 for carbon to carbon dioxide, these values are equivalent to

[19] This analysis has been criticised by some analysts as being far too conservative, with alternative estimates of global costs being in the range of 12% to 130% of GWP by the year 2050 (Meyer & Cooper 1995). Their high values come from taking a maximum value of a statistical life, and applying it to the entire population.

[20] An exchange rate of R5.68 to the UK Pound is used.

R4.91 per ton of carbon dioxide for the low estimate, R21.81 for the central estimate and R43.63 in the high case. Rounding these values up, they become R5, R22 and R44 per ton of CO_2 respectively. On this basis, the estimates of damages which may result from South African power station emissions of GHGs (as summarised in Table 3.12) are shown in Table 4.7.

Table 4.7 Valuation estimates for CO_2-induced climate change damages

	Low estimate (Rm)	Central estimate (Rm)	High estimate (Rm)
Total value (Rm)	714.5	3 143.8	6 287.6
Average mills/kWh	4.83	21.24	42.48

It is evident from the calculations in the table that at any of the per-ton damage costs, global damage costs attributable to GHG emissions from the South African power sector are highly material. For the central estimate, external costs are in the region of 2.1 c/kWh, which is very significant in relation to current electricity prices. The policy implications of this are *not* simply that these costs should be internalised into the price of coal-fired electricity, since there would be no benefit for South Africa of doing so if other emitters of GHGs did not (the `free rider' problem). Rather, the policy response will depend on the international regimes being negotiated in terms of the Framework Convention on Climate Change, which explicitly recognises the differential responsibilities of industrialised and developing countries in the mitigation of climate change impacts. This issue will be addressed further in Chapter Seven.

2.5 Valuation of subsidies to nuclear industry

The production of nuclear electricity in South Africa involves two kinds of externalities: environmental and fiscal effects. In Chapter Three, it was established that, while environmental externalities may be significant, there is insufficient information to value them in this study. Fiscal externalities, however, were classified as a Class One impact.

The cumulative subsidy which was directed to the nuclear industry from 1971 to 1995 amounted to R21 753.3 million (in 1995 Rands - refer to Table 3.13), of which R12 298 million was estimated to be directly related to the production of electricity. In order to calculate an average external cost, this subsidy should be spread, as far as possible, over the lifetime of the assets to which they are related.

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[20] An exchange rate of R5.68 to the UK Pound is used.

There is currently some uncertainty over the future of Koeberg power station and the Atomic Energy Corporation (AEC). Thus three scenarios will be used for purpose of calculating the fiscal externality:[21]

- low estimate: the subsidy ceases from 1996/97 onwards and the power station operates at full capacity until 2023;
- central estimate: the subsidy is phased out from its current level to zero by the year 2000,[22] while operations remain at their current level;
- high estimate: support for the industry terminates and production of electricity ceases at the end of 1996.

Table 4.8 summarises the external costs for each of these three scenarios; subsidy figures are drawn from Chapter Three and Auf der Heyde (1993), whilst electricity generation figures are based on actual production since the power station was commissioned, and assumed output for the remainder of its forty year life at 1995 levels.

Table 4.8 Valuation of fiscal externalities in the nuclear industry

	Low estimate	Central estimate	High estimate
Total subsidy (Rm 1995)	R12 298	R12 995	R12 298
Cumulative generation (GWh)	370 848	370 848	109 029
External cost, mills/kWh (nuclear elec)	33.16	35.04	112.80

Under any of these three scenarios, it is clear that the nuclear sector has benefited from a highly significant fiscal subsidy which has not been reflected in the price of nuclear electricity. Clearly, a portion of the fiscal allocation to the nuclear industry has not been used to *produce* electricity - for example, the costs of operating the Council for Nuclear Safety - but

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[21] These options span the range of possible policy decisions about the future of the nuclear industry. The central estimate is based on the Atomic Energy Corporation's `2000 plus' business plan which it developed in 1993/94.

[22] Thus the allocation to the electricity component of the industry for the next five years 1996 to 2000 is assumed to be 57% (the same percentage as for the period to 1995) of the annual allocations: R489.2 million for 1996/97, R366.9 million for 1997/98, R244.6 million for 1998/99, and R122.3 million in 1999/2000 (all in 1995 Rands).

this is not the point. What is relevant, is that none of these costs would have been incurred if South Africa had not developed its nuclear capacity, and even the costs of maintaining safety standards are external costs related to the production of nuclear electricity. Consequently, their evaluation as an externality is consistent with economic theory - with the caveat that this is an *average* external cost calculated over the lifetime of the plant.

Unfortunately there is no publicly available information about the revenues derived from the Koeberg power station's output - this would allow for a calculation of the gross profit (or loss) of producing nuclear electricity. However, given that the marginal (private) cost of supplying electricity for all of Eskom's power stations is believed to be in the region of 4.0 cents per kWh (Davis 1995: 28), and with an average fiscal externality in the region of 3.5 c/kWh, the social cost of nuclear electricity is likely to be approximately double the average of coal-powered electricity. This underlines the high cost at which South Africa's nuclear complex has been constructed.

It should also be noted that the central scenario assumes that subsidy allocations to the nuclear industry will be scaled down to zero by 2000 - consistent with the so-called `AEC 2000 plus' business plan; however, the trend in the 1995/96 and 1996/97 budget allocations does not appear to be in line with this vision. Rather, the subsidy has not been reduced to the extent envisaged in the AEC's own business plan, which means the central estimate may understate the amount of the fiscal externality.

3. Summary of valuation results

The combined effect of valuing the five Class One externalities is summarised in Table 4.9. The estimates reflect the results of the valuation exercise undertaken in this study; as noted earlier, only the most important externalities, for which there is sufficient information, have been quantified in this study. Consequently, the valuations constitute an incomplete estimate of the external costs of electricity generation in South Africa and, as such, reflect a *minimum* estimate. Table 4.9 also summarises the level of uncertainty related to each of the externality valuations, taking into account the reliability of the data sources and the completeness of information about various stages in the impact pathway.

The externalities in Table 4.9 can also be expressed in terms of a cost per unit of electricity generated. This is done in Table 4.10. The weighted average external cost takes into account the relative proportions of coal and nuclear electricity generated by Eskom (148 003 GWh and 9 697 GWh respectively for 1994).

Table 4.9 Summary of valuation results for Class One and Class Two externalities in 1994, total Rm

	Level of uncertainty	Low estimate	Central estimate	High estimate
Coal mining: injuries & mortalities[23]	moderate	19.1	29.1	46.0
Generation: water consumption	moderate	120.8	185.8	250.7
Generation: air pollution & health impacts[24]	moderate	622.9	854.2	1 046.6
Generation: greenhouse gases	high	714.5	3 143.8	6 287.6
Nuclear: fiscal subsidy	moderate	12 298.0	12 995.0	12 298.0
Total		13 775.3	17 207.9	19 928.9

Table 4.10 Summary of externality valuations for coal and nuclear cycles in mills/kWh, 1994

	Low estimate	Central estimate	High estimate
Coal mining: injuries & mortalities	0.13	0.20	0.32
Generation: water consumption	0.82	1.26	1.69
Generation: air pollution & health impacts	4.21	5.77	7.07
Generation: greenhouse gases	4.83	21.24	42.48
Total coal fuel cycle	9.99	28.47	51.56
Nuclear: fiscal subsidy	33.16	35.04	112.80
Weighted average external cost	11.41	28.87	55.33

Expressed as an external cost per kWh of electricity produced, the central estimate for all Class One externalities included in this study is therefore 28.87 mills/kWh, (2.89 cents per

[23] The external costs of coal miners' morbidity (chronic and acute illness) have not been quantified because insufficient information exists regarding their pollution exposures. The figures here are the sum of those in Tables 4.2 and 4.3.

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kWh), with a lower bound of 11.41 mills/kWh and an upper estimate of 55.33 mills/kWh. It is evident from the above tables, that the highest external costs are attributable to greenhouse gas emissions and the fiscal subsidy in the nuclear industry. If both of these are ignored, then the central estimate of damages is 7.23 mills/kWh (0.72 cents/kWh), with low and high ranges of 5.16 mills/kWh and 9.08 mills/kWh respectively.

These results can be placed into context by comparing them with two sets of variables: international externality valuations and South Africa's current electricity price. Firstly, with reference to the international externality valuations reported in Chapter Two (see Table 2.1), it is evident that the present results fall within the range of reported results, although at the lower end of the scale. This is as expected, given that several potentially important externalities (such as acidification) have not been quantified here; also, the tendency in this study has been to err on the conservative side where judgement has had to be exercised.

The second comparison is with current electricity price levels in South Africa. The relative significance of the externalities obviously varies depending on the choice of a benchmark tariff. For present purposes, three of Eskom's tariffs can be used: its average industrial tariff, the prepayment domestic tariff and the weighted average tariff for all consumers. These tariffs at the end of 1994, were 10.06 c/kWh, 25.81 c/kWh and 11.76 c/kWh respectively (all inclusive of 14% VAT, Eskom 1995a: 2, 1994: 26). The results of this comparison are shown in Table 4.11.

Table 4.11 Average external cost as a percentage of industrial, prepayment domestic and average tariffs, 1994

	Low estimate (%)	Central estimate (%)	High estimate (%)
Average industrial tariff	11%	29%	55%
Low-income domestic tariff	4%	11%	21%
Average Eskom tariff	10%	25%	47%

Table 4.12 makes the same comparison, but this time compared only to Eskom's average tariff for 1994, and with a breakdown for each of the five Class One externalities.[25] This

[25] The percentages are calculated by dividing the values in Table 4.10, adjusted for the split between coal and nuclear electricity (148 003: 9 697), by the weighted average tariff of 11.76 c/kWh.

makes it possible to evaluate the effect on prices of any combination of these externalities. For instance, if the damage costs related to greenhouse gas emissions were ignored for reasons of uncertainty or in response to developments in the international political arena, then the central estimate for the externalities would amount to almost 8% of average prices for 1994. Likewise, if nuclear externalities are ignored, then the central estimate of external effects would be about 23% of average price levels.

Table 4.12 Average Class One externalities as a percent of Eskom's average tariffs, 1994

	Low estimate	Central estimate	High estimate
Coal mining: injuries & mortalities	0.1%	0.2%	0.3%
Generation: water consumption	0.7%	1.0%	1.3%
Generation: air pollution & health impacts	3.4%	4.6%	5.6%
Generation: greenhouse gases	4.1%	18.1%	36.1%
Nuclear: fiscal subsidy	1.7%	1.8%	5.9%
Total[26]	9.7%	24.5%	47.0%

The results in Table 4.12 show the significance of the Class One externalities taking into account the relative mix of coal and nuclear electricity, in relation to average tariffs for 1994. Because nuclear electricity accounted for just over 6% of the coal-nuclear total, its relative impact on electricity prices becomes much smaller than when compared to nuclear-generated electricity alone. Much more significant in this comparison, is the impact of assumed damage costs from climate change, representing 18% of average tariffs in the central case. Figure 4.1 shows that damages from GHG emissions completely dominate the other externalities. Of the environmental externalities experienced within the country and its neighbours in the relatively short term, health impacts of air emissions are most significant, representing between 3% and 6% of current electricity prices.

Figure 4.1 shows the share of each externality as a percentage of total externalities in the central estimate. It is evident from this that damages attributed to GHG emissions dominate other externalities, accounting for almost three-quarters of total Class One external costs. Health impacts of air pollution emissions account for just under 20% of the total.

[26] The total % is based on the weighted average external cost, as per Table 4.10.

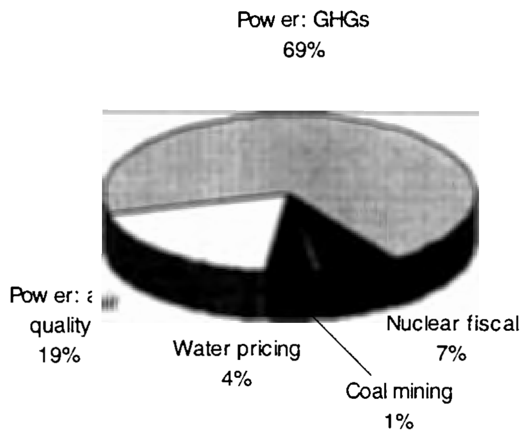

Figure 4.1 Relative significance of externalities in the coal and nuclear fuel cycles, central estimate

Finally, the external costs calculated in this chapter can be expressed in relation to the amount of energy which is delivered by the electricity generation process. This will allow for these externalities to be compared with externalities in the household energy sector. In 1994, Eskom used an average of 0.520 kg of coal per kWh generated; at the same time, the average calorific value of coal burnt by its power stations was 20.09 MJ/kg and the overall thermal efficiency was 34.4% (Eskom 1995a: 54); thus the amount of energy delivered (before transmission and non-technical losses) was 3.59 MJ per kWh. This compares closely with standard statistical estimates of 3.6 MJ/kWh (NEC 1990: 99). Based on electricity output from the coal power stations in 1994 of 148 003 GWh, the total delivered energy was 531.3 * 10^6 Gigajoules.

Thus if the total external costs (as per Table 4.9) are expressed as a function of units of energy delivered, the results in Table 4.13 are derived. For purposes of this comparison, the fiscal externality in the nuclear cycle is omitted, for two reasons: first, it is not an environmental externality, so for purposes of making comparisons with environmental externalities in the household sector, it will be excluded. Secondly, the valuations for the fiscal externality represent average historical external costs - or sunk costs - whereas the other valuations in this chapter are average external costs for 1994, and as such, are closer to the margin.

Table 4.13 Average external costs of electricity generation externalities in relation to quantity of delivered energy

Externality	Rands per Gigajoule		
	Low estimate	Central estimate	High estimate
Coal mining: injuries & mortalities	0.04	0.05	0.09
Generation: water consumption	0.23	0.35	0.47
Generation: air pollution & health impacts	1.17	1.61	1.97
Generation: greenhouse gases	1.34	5.92	11.83
Total	2.78	7.93	14.36

Presented this way, the data are not especially meaningful. However, in the final chapter of the thesis, these average external costs will be compared with the external costs arising in the household energy sector, using the same formulation. Also in the final chapter, important limitations and weaknesses of this analysis will be outlined.

Following a similar approach to the last two chapters, the following two chapters investigate the external effects arising from the household energy consumption sector.

Chapter Five

Externalities in the household energy sector

Energy consumption patterns in South African households are highly differentiated, with households using a wide range of energy sources: electricity, coal, fuelwood, dung, paraffin, gas, candles and batteries. For many households, consumption choices have been severely constrained by political and economic factors, and as a result they have relied on fuels which are more polluting, hazardous and inconvenient. Consequently, significant externalities arise in this sector. Moreover, many household externalities (such as air pollution) are non-point source effects, which presents particular difficulties from a policy and management point of view.

This chapter reviews the main external diseconomies in the sector, in order to identify those which are potentially serious and about which sufficient information exists to quantify them. To begin with, current household energy patterns are described briefly, including trends which have begun to emerge in recent years.

1. An overview of household energy consumption in South Africa

Household energy consumption is far from homogenous across South African society. Several characteristic features may be identified. The first and probably most noticeable, is that the level of access to electricity is low. In 1990, approximately 33% of households were connected to the national grid (Van Horen et al 1993a). Almost all white households enjoyed electricity services, and only a few black townships were connected.[1] This level of access is low in relation to other developing countries with similar income levels; for example, access in Argentina is about 88%, Venezuela 86%, Thailand 75% and Brazil 65% (Eberhard & van Horen 1995: 48).

Reasons for the relatively low level of access to electricity in South Africa have been analysed extensively elsewhere (ibid: 90-100, Steyn 1994, Theron et al 1991), but several points are worth noting in the current context. Firstly, the low level of access is certainly not attributable to inadequate supply capacity: indeed, South Africa was among the first countries in the world to have electricity - street lights were installed in Kimberley in 1882, even before London (Steyn 1994: 3). Currently, Eskom has considerable surplus capacity, with a number of power stations having been mothballed. Rather, the low level of access to

[1] Soweto was the first historically black township to be electrified, commencing in 1979, but even there, the quality of supply was extremely poor and unreliable.

electricity can be explained by the political economy of apartheid. As noted in Chapter One, legislative responsibility for provision of services such as electricity has historically rested with racially-based local governments. Given the extremely limited resources, both financial and technical, of historically black local authorities, coupled with their political illegitimacy, very little progress has occurred in the extension of electricity services to the poor.

The result is that only a few urban townships were connected to the national grid, and almost no rural villages: in 1990, levels of access were about 60% in urban areas compared to only 6% in rural areas (Eberhard & van Horen 1995: 50). Since the launching of the national electrification programme in 1991, levels of access have risen, particularly in urban areas. By the end of 1994, it was estimated that overall access had risen to 44%, with about 72% of urban households, and 12% of rural households having an electricity supply (Davis 1996). Estimates were that by the end of 1995, total access to electricity was close to 50% (Barnard 1995). This improvement was a result of 1 million new connections during that period, of which Eskom was responsible for almost three-quarters.[2] Indeed, the electrification programme is one of the few service delivery programmes in which the Reconstruction and Development Programme's goals are being met or exceeded.

A second important characteristic of household energy consumption patterns, is that most poor households use multiple sources of fuel to meet their energy needs. This point is best understood by considering energy consumption from a demand-side or end-use perspective, as suggested by the Integrated Energy Planning (IEP) methodology.[3] In some respects, energy is analogous to money: it is demanded not for its own sake, but for the services which it can provide. Thus, households use energy for cooking, space heating, water heating, lighting, and to power appliances such as irons, radios and televisions. In the same way, industrial consumers demand energy to provide motive power, heat or cooling. Therefore, when analysing energy consumption patterns, and when developing policy interventions, the optimal match must be found between the range of energy supply options (electricity, wood, coal and so on) and demands for these energy services.

The phenomenon of multiple fuel use has been widely-observed in low-income households in South Africa (Williams 1994). In many respects, it is a result precisely of consumers' practice of matching available supply options with their demand for various energy services. Thus, for instance, many households use candles for lighting, wood for space heating and

[2] The rate of connection during 1995 was just over one thousand per day.

[3] The IEP methodology is widely used in energy planning and policy analysis as the appropriate approach for analysing energy policy options (refer to Munasinghe 1990, APDC 1985, Eberhard 1992, Eberhard & Theron 1992).

cooking, paraffin for heating water and cooking, and batteries to power a radio. Even in newly-electrified households, non-electric sources of energy continue to be used, especially for the more energy-intensive services such as space heating and cooking, and particularly where cheaper alternatives (such as coal or wood) are available. This pattern of multiple fuel use is reflected in low electricity consumption levels in newly-connected households: at the end of 1995, the average monthly consumption of Eskom's new customers was only 90 kWh (Barnard 1995), compared to 500 to 800 kWh in middle to high income households which use only electricity.[4]

A third important characteristic of household energy consumption in South Africa relates to the significant role of biomass resources (fuelwood, animal dung, crop residues and other organic matter). Fuelwood accounts for the bulk of delivered energy in rural areas, with between 80% and 100% of those households using wood (Van Horen et al 1993b: 16). Fuelwood is used for cooking, space and water heating, and sometimes even for lighting. In urban areas, however, it is much less commonly used, except in newly-urbanised families, and appears to be used in most cases as kindling for coal fires (Williams 1994: 12). A recent estimate is that approximately 17 million rural people depend on fuelwood as their main source of energy (Van Horen 1993). In most rural homes, wood is burnt without stoves, using traditional `three stone' fires; as will be seen later, this results in major pollution problems.

Fourthly, coal is an important source of energy for many households, particularly those close to the coalfields on the Highveld. Due to its high transport costs, coal is much more expensive in areas further afield, where other sources of energy such as wood or paraffin are usually more economical. Coal is especially dominant in the urban townships of Gauteng, where cold winters make space heating essential. Moreover, coal stoves represent important assets for poor households since they serve multiple purposes simultaneously: they can cook, heat water and heat the home, and serve as a focal point for social activities. These factors, together with the sizeable investment which a stove represents for poor people, accounts for the continued use of coal, even in homes which have been electrified (Williams 1994: 12). This tendency is also largely a result of the unreliability of electricity supplies in many townships, which means that people are forced to retain old coal stoves as backups. The net effect of these factors is that coal is widely used. Altogether, about 1 million

[4] 90 kWh is the moving average of all Eskom's customers connected since 1991, including urban and rural consumers. Consumption levels show a gradual but steady incline over time, but at this stage, the average is low because it is dominated by the large number of recently-connected customers.

households are estimated to use coal (Palmer Development Group 1995), corresponding to some 6 million people.

A fifth important characteristic of household energy use in South Africa is that paraffin (kerosene) is used very widely in unelectrified homes for cooking, heating and lighting. Unlike coal and wood, however, it is seldom used in electrified homes, except as a back-up in the case of electricity black-outs. One reason for its pervasiveness, is that the distribution system for paraffin and for paraffin appliances is very extensive, even in rural areas. Moreover, the informal distribution network in urban townships, which involves *spaza* shops (informal traders), means that households with low and irregular incomes are able to purchase paraffin in small quantities and at most times of the day or night, thus making it affordable (if not cheap). Paraffin appliances are also widely available and relatively inexpensive. This informal distribution system, while holding obvious advantages for poor households in terms of having access to an affordable energy source, also leads to serious health hazards which will be described later in this chapter.

Sixthly, liquefied petroleum gas (LPG) is also used in many unelectrified households, although much less-commonly than paraffin. Generally, LPG is used by households with slightly higher incomes, and only in urban areas where distribution points are relatively close to peoples' homes. This is because gas is slightly more expensive than paraffin, and considerably more expensive than coal or wood; furthermore, it requires some effort to transport cylinders to and from depots for refilling. Levels of LPG usage are highest in the Western Cape where average income levels are higher, and where coal and other fuels are not available as cheaply (McGregor 1994: 28).[5]

Seventh, candles are commonly used in unelectrified households for lighting. Generally, they are completely substituted by electricity after connection to the grid, except for use as backup in the case of power failures. Apart from the higher cost and lower quality of light emitted by candles compared to electricity, they are also associated with fire hazards.

A final characteristic of energy use in the household sector of relevance to the current discussion, is that many dwellings in South Africa have very poor thermal performance (Thorne 1994). In other words, houses are often energy-inefficient, in that they have no ceilings or insulation, and do not take advantage of solar heating (by facing North) in colder regions, or of natural shade in warmer areas. The result of this is greater energy demands to achieve the same level of utility, which in turn obviously has negative environmental

[5] In some middle-income residential areas in Johannesburg and Cape Town, an old piped gas network also still operates, although there has been no new investment of this kind for several decades.

implications. This is especially true of informal dwellings (shacks) and even of low-cost (so-called `matchbox') formal houses where these have been built without regard to their energy efficiency.

To conclude, it is only in the middle-income and high-income sector, in which electricity supplies have long been enjoyed, that the fuel mix is relatively homogeneously based on electricity. In the remainder of the population, a wide range of fuels is used, with significant variations between areas, depending on climatic factors, income levels, access to various fuels, and social factors. It is in the latter group that significant externalities exist; the following sections describe these in more detail.

2. Externalities in the household energy sector

Compared to the electricity generation sector, there are fewer externalities in the household sector. As a general rule, it is fair to say that the latter effects have received less attention in the literature or from policy-makers and analysts. There are several reasons for this:

- The politics of apartheid meant that energy and environment problems in historically black residential areas were accorded a low priority by the state and public utilities under its control (such as Eskom). Thus little attention was directed towards identifying and ameliorating any environmental problems which existed. Even those authorities ostensibly most closely in tune with local environmental problems - local governments - had little or no capacity, nor indeed, an interest in attempting to address such environmental concerns as existed.[6]

- There were (and still are) no strong institutions into whose ambit household environmental problems naturally fell. By contrast in the electricity sector, for instance, it was natural that Eskom would implement procedures to manage its own environmental impacts. In the household sector, there has never been a single institution, whether in the government or private sector, which has had responsibility for household environmental concerns.

- The nature of many of the externalities in the household sector is that they are more diffuse and dispersed than in the electricity sector. Millions of households are responsible for emissions of air pollution, and the range of other effects which occur,

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[6] This is particularly evident from the way in which smoke-control zones promulgated by the Atmospheric Pollution Prevention Act (No 45 of 1965) were enforced only in historically-white residential areas, but not in black townships where pollution problems were much more acute (Van Horen 1994: 81).

whereas in the power sector, for instance, there is only one major institution, and it has nine point sources of emissions. Thus to monitor problems and/or intervene in the household sector is more complicated.

- The nature of property rights and ownership differs considerably between energy sub-sectors. For instance, in the Mpumalanga Highveld where air pollution impacts from power stations are felt most acutely, much land is owned by commercial farmers and firms. By contrast, the rural lands from which most households obtain their wood supplies, are under communal ownership. Whilst it would be simplistic to suggest that this factor alone (poorly-defined property rights) causes increased degradation in the latter sector, it is clear that the interests of private land-owners are directly compromised by pollution and degradation affecting their land. Given their historically greater influence over political and government decision-makers compared to rural inhabitants, it follows that their concerns have received more attention.

In this chapter, the following externalities are evaluated, with a view to assessing whether they are potentially significant and whether there is sufficient information to permit their economic quantification; the same classification system (Class One, Two or Three) is used as for externalities in the electricity generation sector:

- Air pollution from coal combustion.
- Air pollution from wood combustion.
- Accidental paraffin poisoning in infants.
- Accidental fires and burns.
- Natural effects of fuelwood scarcity.
- Social effects of fuelwood scarcity.
- Greenhouse gas emissions from household energy use.

Each of these is analysed below, and Class One impacts are quantified in Chapter Six. It is interesting to note that some of the above effects differ from the conventional conception of an externality (which is often portrayed, for example, as the loss of utility suffered by a person living downwind of a factory which pumps pollution into the air). This conception is not quite pertinent in the cases of air pollution from wood (and to a lesser extent, coal), paraffin poisoning, fires and burns, and social impacts of wood scarcity. In these cases, the persons consuming the fuel and thus causing the external effect, are the same as, or close to, the person suffering the external effect. Thus, for instance, a person who burns wood in her home, and then also breathes the pollution caused by that wood fire, is experiencing the loss of utility (the externality) herself. It might therefore be argued that the negative effects arise from her own conscious actions and are therefore not an externality in the conventional sense.

Whilst it is true that these effects are not the same as conventional externalities, it is considered appropriate to include them in this analysis, for the following reasons: firstly, the external diseconomy resulting from these effects is seldom, if ever, confined to the person who purchased that fuel. The distribution of external costs, in almost all cases, is spread between energy consumers, the public health sector and neighbours and friends who assist with any accidents. Second, the household is not a single indivisible entity, since the decision to purchase a fuel is often made by the woman in the household, whereas the external diseconomy affects not just herself but other members of the household. Finally, even where environmental and other effects are borne largely within a household, perhaps even by the purchaser and consumer of the energy source, it remains true that the social cost of the item diverges from its private cost. These divergences, or external costs, are the focus of analysis in this chapter.

2.1 Air pollution from coal combustion

In the earlier analysis of air pollution from power stations, the impact pathway approach was used to trace the effects of air pollution emissions through to human health outcomes. In the case of household coal combustion, there is little or no information in South Africa on the quantity of pollutants emitted by coal consumers. However, there have been several studies of air quality and human exposures to pollution. These are described briefly below, under three headings: outdoor pollution levels, indoor pollution concentrations and finally, the effects of exposure on health.[7]

2.1.1 Outdoor air quality

The earliest monitoring of air quality in South Africa commenced in 1955, but the concentration of financial resources in wealthier white local authorities, and the neglect of township interests, meant that this monitoring was based mostly in city centres. Beginning in the 1970s, some surveys were done in Soweto, although these were intermittent (Annegarn 1990: 16). A more comprehensive picture of township air pollution emerged in the 1980s when various monitoring programmes were established in Soweto (Turner et al 1984, Kemeny et al 1988). The scope of these programmes has since widened to include not only sulphur dioxide and particulates, but also nitrogen oxides, carbon monoxide, ozone, and other pollutants (Tosen et al 1991). Recently, similar studies have been carried out in other

[7] For a review of international air pollution studies, refer to Van Horen (1994: 20-27).

coal-burning areas in Gauteng, such as Sharpeville. The results of all of these studies present a bleak view of air quality in townships.

Soweto's air quality has been studied more widely than that of any other township in South Africa, partly because its air is so heavily polluted, and partly because electrification was expected to bring about substantial improvements in air quality. For various reasons, however, coal continues to be used on a large scale in Soweto and, consequently, air pollution problems remain serious.

Eskom commenced its monitoring programme in Soweto in June 1983. Results from the first few months of operation showed that concentrations of total suspended particulates (TSPs) frequently exceeded the US Environmental Protection Agency (EPA) primary and secondary 24 hour standards (Turner et al 1984).[8] Long term sulphur dioxide (SO_2) levels approached the EPA standards, while nitrogen dioxide (NO_2) levels were well within the limits. Measurements carried out by Eskom from August 1990 to July 1991, again revealed that concentrations of particulates were unacceptably high. The mean annual concentration of fine particulate matter (PM10, particles with a diameter of less than 10 μm) over this period was 112 μgm^{-3}, more than double the US standard of 50 μgm^{-3} (Turner & Lynch 1992: 2). In addition, there were 84 days during the year under examination in which the 24 hour EPA standard was exceeded (Sithole et al 1991: 7). Strong seasonal and diurnal fluctuations in concentrations confirmed that particulate pollution is closely related to household coal usage patterns (see Figures 5.1 and 5.2), as well as to the dust created by vehicles travelling on unpaved roads.

[8] As noted earlier, South African air quality guidelines are similar to United States EPA standards, both of which follow the convention of being determined by health-related criteria (as opposed to economic criteria). Moreover, health-based standards implicitly assume a threshold value below which health effects are either assumed not to occur or to be acceptably low. As noted earlier, recent epidemiological findings do not support the existence of these thresholds.

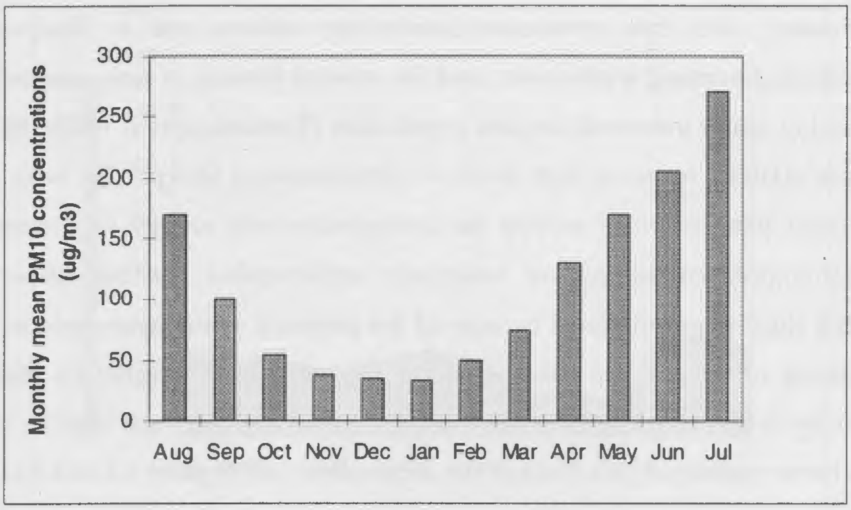

Figure 5.1 Monthly mean PM10 concentrations in Soweto, 1990 - 1991 (*Turner & Lynch 1992)*

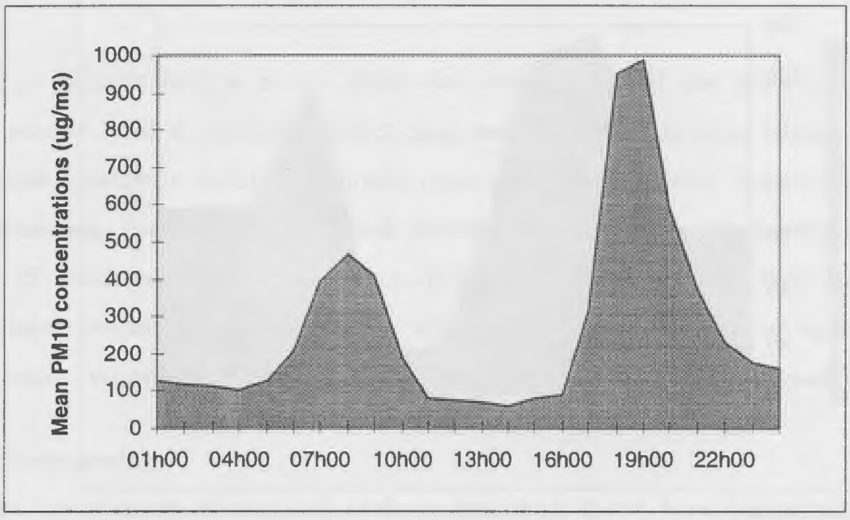

Figure 5.2 Average diurnal distribution of PM10 concentrations in Soweto (Turner & Lynch 1992)

Levels of gaseous pollutants were also monitored, and it was found that SO_2 levels followed similar cyclical profiles - winter levels far higher than those in summer, and strong morning and evening peaks. The measurements recorded were mostly within the health guidelines, although these were occasionally exceeded during winter when temperature inversions occur close to ground level, thus hindering the dispersal of pollution. While NO_x levels were higher than those of SO_2, they were well within the government's health guidelines. The diurnal and seasonal distribution of NO_x suggested that the bulk of this pollutant is derived from the extensive vehicular traffic in the township.

Another large study, the Vaal Triangle Air Pollution Study (VAPS) based in urban residential areas had an outdoor component (in addition to an indoor component which will be described shortly), with two continuous monitoring stations: one in Sharpeville, a township in which coal-burning is prevalent, and the other at Makalu, a non-residential site located downwind of major industrial sources of pollution (Terblanche et al 1992b: 553). The results from these stations revealed high levels of particulates in Sharpeville, with a very strong daily cyclical pattern which reflects the living habits and energy requirements of residents. Climatological conditions are extremely unfavourable for the dispersal of pollutants emitted close to ground level because of the presence of stable inversions, which are especially strong in winter. The site at Makalu, by contrast, is affected by high-level emissions which are transported over greater distances, and shows much smaller diurnal variation due to better mixing of pollutants in the atmosphere (see Figures 5.3 and 5.4).

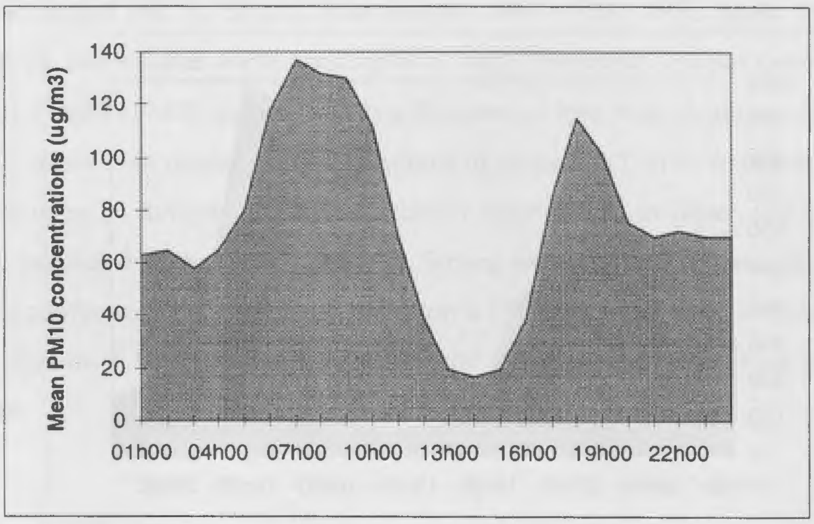

Figure 5.3 Average diurnal distribution of PM10 concentrations, Sharpeville, March to July 1991 (*Terblanche et al 1992b)*

The levels of gaseous pollutants (SO_2, NO_x) monitored at both sites were within health guidelines, but their diurnal profiles followed similar patterns to those for particulates (ibid). These findings confirmed that emissions in Gauteng townships such as Sharpeville come from low level sources, and are closely related to peoples' daily living patterns. The results from Makalu indicate high background pollution levels.

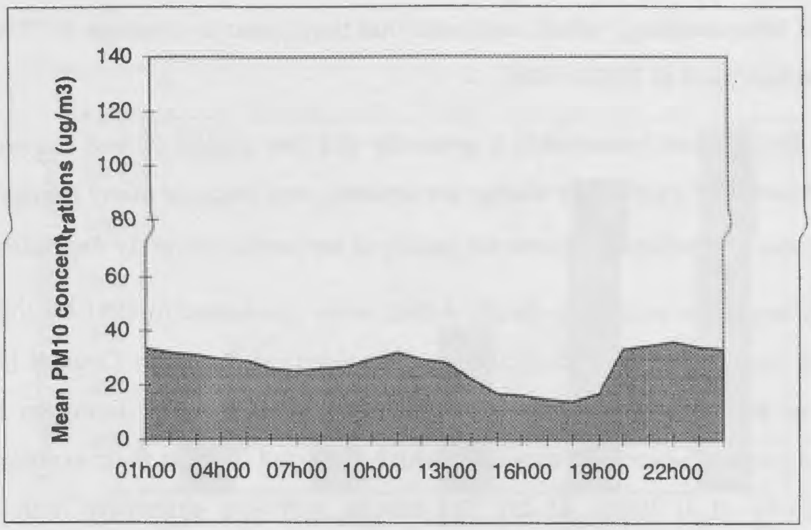

Figure 5.4 Average diurnal distribution of PM10 concentrations, Makalu, March to July 1991 (*Terblanche et al 1992b)*

The nature of urbanisation in South Africa has been such that the poorest communities frequently occupy land in locations which pose serious threats to their health. Whilst this reflects simple economic realities in some respects, it also reflects the racial politics of apartheid planning. For instance, the black township of Zamdela was planned and located downwind of industrial pollution sources in Sasolburg, not because this was the only affordable land available, but because it was on the opposite side of town to white residential areas - where pollution levels were obviously not as serious (Lawson 1991: 61).

2.1.2 Indoor air quality

Pollution studies in South Africa, such as those described above, have historically measured *ambient* levels of pollutants at fixed monitoring stations (Turner et al 1984, Kemeny et al 1988). In recent years, however, some pollution and health studies have utilised the Total Exposure Assessment (TEA) approach which recognises that the health effects suffered by people are a function both of the concentration of pollutants in the air, and of the duration of exposure (Smith 1988: 17). The appropriate monitoring response is therefore to measure the personal exposures of study participants to the pollutants under examination. Thus the data inevitably include exposures from both indoor and outdoor activities. The importance of performing monitoring in this way rather than with traditional, fixed outdoor stations, is evidenced by recent data from a study of school children aged 8 to 12 years in the Vaal

Triangle, south of Johannesburg, which indicated that they spent an average of 75% of their time indoors (Terblanche et al 1992b: 553).[9]

The coal used by low-income households is generally of a low quality (C and D grade). As a consequence high levels of particulate matter are emitted, and because many homes have no chimneys or are poorly ventilated, indoor air quality is frequently severely degraded.

The first personal exposure studies in South Africa were conducted in 1991 by the Council for Scientific and Industrial Research (CSIR) and the Medical Research Council (MRC). In one component of the project, a group of 45 children, aged 8 to 12, living in and near Sebokeng, carried personal exposure monitors which collected data on their exposures to air pollution (Terblanche et al 1992a: 41-43). The results indicated extremely high levels of exposure to total suspended particulates (TSPs), with 12 hour levels exceeding the US EPA 24 hour standard of 260 μgm^{-3} in 99% of cases. Every single case exceeded the World Health Organisation (WHO) `lowest-observed-effect' level of 180 μgm^{-3}.

Average concentrations on a summer weekend day were 387 μgm^{-3} and 620 μgm^{-3} for electrified and non-electrified areas respectively. As expected, increased coal consumption in winter was reflected in higher exposure levels. Average concentrations on a winter weekday were 1 168 μgm^{-3} in electrified areas and 1 363 μgm^{-3} in non-electrified areas. This relatively small difference between electrified and non-electrified areas was an important finding and was attributed to the close spatial proximity of electrified and non-electrified areas, to the continued use of coal even in electrified homes, and to the high background pollution levels in areas where dispersal conditions are unfavourable.

Interesting comparisons can be made with a similar study, the Vaal Triangle Air Pollution Health Study (VAPS), which aimed to assess whether South Africa's air pollution control programme adequately protects human health (Terblanche et al 1992b: 550). As part of VAPS, exposure levels were recorded for Gauteng children from high-income and middle-income suburbs in which electricity was the only energy source. The sample was drawn from parts of the Vaal Triangle which were known to be relatively polluted by industrial sources. These measurements revealed levels of TSPs well below those experienced by Sebokeng school children (refer to Figure 5.5). Nonetheless, 63% of exposures exceeded the EPA 24 hour standard, with the median level on a winter school day being 310 μgm^{-3} (Terblanche et al 1992b: 553). These children came from homes which were fully electrified, which confirms that background pollution levels were very high.

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[9] Clearly, traditional monitoring stations, located on the roof of the town hall or post office, are not very helpful in measuring the pollution levels to which people are exposed.

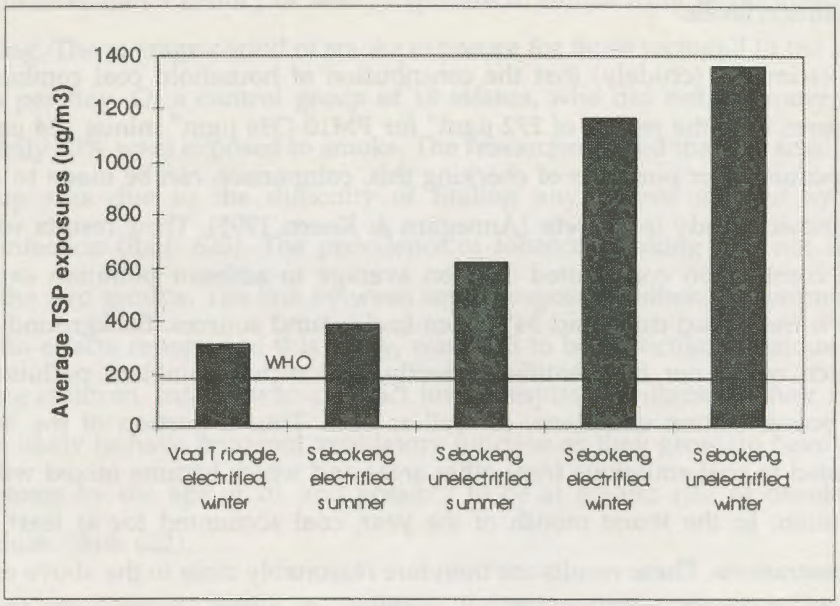

Figure 5.5 Summary of 12 hour exposures to Total Suspended Particulate (TSP) concentrations in Gauteng and Vaal Triangle residential areas, compared to WHO guideline

These data can be used to make estimates of the contribution to air pollution exposures which results from household coal combustion. Average 12 hour exposures to TSPs in coal-using parts of Sebokeng were reported as 1 363 μgm^{-3} in winter and 620 μgm^{-3} in summer, yielding an average of 991 μgm^{-3} for all seasons. Of relevance, however, is not the exposure to total suspended particulates (TSP), but to particles with a diameter of less than 10 microns (PM10), since these fall into the respirable range (Lerer 1995). Larger particles generally do not enter human lungs where respiratory ailments are caused. Since the quality of coal burnt by households is generally relatively poor, the same factor is used as for Eskom's coal, namely that 40% of TSPs fall into the respirable (PM10) range. Thus the average exposure to PM10 in coal-burning townships is estimated to be 396 μgm^{-3}.

Not all of this can be attributed to pollution emissions from household coal combustion. Other sources include emissions from industrial sources, power stations, dust from unsurfaced roads, burning coal dumps and veld fires. The contribution of household coal may be estimated (crudely) by using the results from the VAPS study as a `control' group. In that study, TSP exposures averaged 310 μgm^{-3} in winter, 40% of which (that is, the PM10 portion) is 124 μgm^{-3}. This can be taken as a worst-case estimate of background pollution concentrations since it reflects pollution exposures in an area with known industrial pollution, and is from winter, when pollution levels are usually higher (due to adverse dispersal conditions). On the other hand, the level of road dust in that area would be lower since most roads are surfaced; however, it is unlikely that this would alter the estimate of background pollution levels.

Thus it can be estimated (crudely) that the contribution of household coal combustion to pollution exposures is in the region of 272 μgm^{-3} for PM10 (396 μgm^{-3} minus 124 μgm^{-3}), or 69% of total exposures. For purposes of checking this, comparison can be made to a recent source apportionment study in Soweto (Annegarn & Kneen 1995). Their results were that household coal combustion contributed 50% on average to ambient pollution exposures, compared to 16% from road dust, and 34% from background sources. Background sources were those which could not be identified directly, and include ambient pollution from industrial and power station emissions, as well as coal. Thus a portion of the 34% was probably attributed to coal emissions from other areas and which became mixed with other sources of pollution. In the worst month of the year, coal accounted for at least 74% of particulate concentrations. These results are therefore reasonably close to the above estimate, which will therefore be used for purposes of estimating health outcomes - hence this is classified as a *Class One* externality.

The data described here concentrated on only one aspect of the indoor air environment: exposures to particulate matter. This may represent the most serious energy-related pollution hazard to urban households in South Africa (especially since the coal has a high ash content). Nonetheless, a range of other pollutants are also produced by coal combustion, such as sulphur dioxide, carbon monoxide, nitrogen dioxide, polycyclic aromatic hydrocarbons and benzo(a)pyrene (Terblanche et al 1992a: 17). To-date, no South African studies have measured exposures to these pollutants; consequently, these are classified as *Class Two* impacts.

2.1.3 Health impacts of coal-related pollution

Although there have been no studies which have calculated dose-response functions in respect of pollution exposures to particulate matter, several epidemiological studies have documented various health outcomes.

At the broadest level, acute respiratory infections (ARI) have been reported to be the second most important cause of death in South African children, after gastro-related illnesses (Von Schirnding et al 1991: 81). In some areas, such as Cape Town, ARI (specifically, pneumonia) is in fact the leading cause of infant mortality. Both international and local experience has found strong links between air pollution and the incidence of respiratory illnesses.

The first medical study to document health impacts associated with fuel usage, was conducted in 1980 in an outpatient clinic in Natal, and attempted to establish whether smoke exposure was significantly greater in a group of children with severe lower respiratory disease than in a group of children who displayed no such symptoms (Kossove 1982: 622-

624). Of 132 infants (under 13 months of age) who had severe lower respiratory disease, 70% were found to have had a history of heavy exposure to smoke from wood fires for cooking and/or heating. The average period of smoke exposure for those included in the sample was 6 to 7 hours per day. Of a control group of 18 infants, who did not have any respiratory symptoms, only 33% were exposed to smoke. The researcher noted that the small size of the control group was due to the difficulty of finding any infants without symptoms of respiratory infection (ibid: 623). The prevalence of tobacco smoking was not significantly different in the two groups. The link between smoke exposure (albeit unmeasured) and the adverse health effects reported in this study, was held to be particularly serious in the case of very young children. Infants who contract lower respiratory illness in their first year of life are more likely to have impaired ventilatory function as they grow, to have respiratory illness symptoms by the age of 20, and possibly to be at greater risk of developing chest disease as adults (ibid: 622).

A subsequent case-control study of children in Mpumalanga province attempted to determine whether exposure to ambient pollution resulted in any detectable effects on their respiratory status (Zwi et al 1990: 647). The study compared 1 031 children from schools in the province with a control group of 978 children from less polluted areas. The exposed children experienced higher incidences of respiratory symptoms than the control group, and after correcting for age, the exposed children were found to be slightly shorter on average (0.83 cm). While this could be attributable to other factors such as parental smoking and genetic make-up, this finding confirmed a similar observation in an earlier study by Coetzee et al (1986: 339).

In the latter study, a group of (middle-income sector) children from schools in the heavily industrialised Sasolburg area, were compared with a control group of children from nearby rural towns where pollution was negligible (ibid: 339-343). Questionnaires were administered to 674 children in Sasolburg and 332 from the control group and, surprisingly, found no important differences in the reported incidence of respiratory illness. There was however, a significant difference between boys from the two groups in the most important measure of lung function: forced expiratory volume in 1 second (FEV_1) was lower in the pollution group. Other measures of lung function were also found to be slightly worse in the study area. Children in the control area were bigger (height and mass) than those from the study area, and in the case of boys, this difference was significant. The results of the study indicated that the trends were unlikely to be affected by smoking habits of the families, but that social class from which the children originated may have been a contributory factor (ibid: 339).

The CSIR/MRC study of children in Sebokeng and Lekoa referred to earlier has also provided an indication of the adverse health effects associated with domestic coal usage

(Terblanche et al 1992a: 52-53). A health survey was conducted with mothers of 1 563 children aged 8 to 12 years, living in townships in Gauteng. Coal was found to be the main fuel for cooking and heating, and was used by over 50% of households. The study examined the prevalence of respiratory illness of participants living in three areas: those without electricity (using mainly coal), `partially electrified' areas (households with an electricity connection, but using other fuels as well), and `fully electrified' areas (electricity only). Respiratory symptoms were classified as either lower respiratory illness (LRI, defined as presence of bronchitis, pneumonia, asthma, phlegm, coughing and chest wheezing during the past two months), or upper respiratory illness (URI, defined as presence of a running nose, earache and sinusitis during the past two months) (ibid: 58-59). The results of the health survey indicated that children from homes which used coal for cooking had a 120% higher prevalence of URI than those which used paraffin (ibid: 60). The general trend in the data pointed towards higher prevalences of URI in groups using coal, than those using electricity or paraffin. The prevalence of LRI indicated the same pattern as for URI: that is, coal users had higher risks by a similar order of magnitude. Significantly, the health survey was carried out in summer which represents the most favourable possible outcome, as pollution exposure levels are at their lowest.

Where coal was the primary domestic fuel, it was found to be the most significant predictor of respiratory illness when compared to crowding, socio-economic status and parental smoking. This corresponded with the perceptions of the participants, the majority of whom perceived that domestic coal burning was the primary source of air pollution, followed by motor vehicle emissions (ibid: 55). On the whole therefore, the study found that a higher prevalence of upper and lower respiratory illness was associated with use of coal, compared to electricity, paraffin and gas.

Thus a common theme in the South African literature is that air pollution from coal is serious and that significant health effects may arise. For purposes of estimating these health effects, the same dose-response data will be used for household exposure to pollutants as was used for exposure to power station emissions (refer to Chapter Three). This is appropriate, since physiological responses to pollution exposures will be the same pollutant will not depend upon its source; moreover, the pollution data described here, refer to personal exposures of household members. The calculations of health outcomes are described in the next chapter.

2.2 Air pollution from wood combustion

Rural areas generally do not experience high levels of outdoor air pollution, because of the lower spatial densities of households, the low volume of vehicular traffic and the absence of concentrated industrial sources of pollution. However, indoor air pollution levels are sometimes extremely high, due to emissions from wood fires. Moreover, the practice in South Africa is that wood fires are frequently made in open fireplaces in the centre of the room used for cooking (or outdoors, weather permitting), and very seldom have chimneys for ventilation.[10]

Anecdotal evidence suggests that levels of air pollution inside many rural homes can reach exceptionally high levels. In spite of the potential seriousness of the problem, pollution monitoring and control efforts have, until recently, almost entirely neglected rural households. The first and only study in South Africa to-date to measure the levels of pollution encountered in rural homes, has been conducted by the CSIR and MRC in parallel to the urban study in Sebokeng/Lekoa referred to earlier.[11] This study utilised both personal monitors (for particulates), worn by 17 children, and fixed monitors which measured concentrations of sulphur dioxide, nitrogen dioxide, carbon monoxide, total volatile organic compounds and total suspended particulates (CSIR 1992: 1).

The results of the study indicated that exposures of the children to total suspended particulates (TSPs) exceeded safety guidelines in *all* cases (ibid). Personal exposures varied between 1 044 μgm^{-3} and 8 330 μgm^{-3}, with a mean of 2 367 μgm^{-3}. The fixed monitors located in the cooking areas reported average TSP concentrations over a 12 hour period of 1 558 μgm^{-3}, while those in sleeping areas averaged 734 μgm^{-3} over the same period. These exposures are far in excess of those measured in any of the urban studies referred to earlier in this chapter, even those recorded during winter when pollution concentrations are at their highest.

From these measurements, an average TSP pollution exposure can be assumed to be 2 367 μgm^{-3} for these households. The PM10 portion of this amounts to 947 μgm^{-3}. Clearly, there is high uncertainty inherent in extrapolating the results from one micro-scale study of only 17 participants, to the entire fuel-consuming population: some 17 million people. It is difficult to assess whether the measurements of the study reported above are representative of other

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[10] This differs from other parts of Africa, for instance, where the practice of rural households is to use wood stoves of one kind or another. Thus the efficiency of combustion in South African households, using a traditional three-stone fire, is low; pollution is correspondingly higher.

[11] In early 1996, plans were being made for a major WHO-supported study of the health effects of fuelwood use in a rural Southern African settlement (and of possible mitigation interventions).

areas. Consequently, it is worth reviewing briefly the findings from international monitoring studies.

Two studies which were performed under the auspices of the WHO's Human Exposure Assessment Location (HEAL) project, reported very high concentrations of respirable particulates in households using fuelwood (WHO 1987, 1988). In the first study, carried out in Maragua, Kenya, 24 hour measurements of respirable suspended particulates (PM10) and other pollutants were made in 36 households where most of the cooking was done on open fires using firewood or crop residues (WHO 1987: 1). The mean of the 24 hour average measurements was 1400 μgm^{-3}. During the average seven hour period in which most households were using fires each day, levels were in the region of 3 500 to 4 000 μgm^{-3}. In the evenings, during the main cooking times, and when ventilation conditions were poorest, particulate concentrations of up to 36 000 μgm^{-3} were found (ibid: 19).

The second HEAL study was conducted in The Gambia in 1987, and reported similar levels of particulates in a small group of 18 households where wood was used in open fires (WHO 1988: 1). The 24 hour average particulate concentrations were 2 000 μgm^{-3} in the dry season, and 2 100 μgm^{-3} in the rainy season. These particulates, however, included larger particles from a source other than wood smoke (probably sand dust). Maximum and minimum extremes (also 24 hour averages) differed between rainy and dry seasons: in the rainy season, the range was 150 μgm^{-3} to 6 200 μgm^{-3}, while in the dry season, the range was from 600 μgm^{-3} to 3 500 μgm^{-3} (ibid: 12). Assuming that the proportion of PM10 to TSP is also 40%, these results equate to a range of 60 μgm^{-3} to 2 480 μgm^{-3} for the entire season.

Other studies have been conducted in countries where people use predominantly fuelwood, and have also found extremely high average concentrations of particulates inside homes: these obviously vary widely, but are usually in the range 1 000 to 8 000 μgm^{-3} (WHO 1992: 71). Over shorter periods, especially during evenings when cooking is done, particulate concentrations have frequently been found to be even higher (ibid, Pandey et al 1989: 428).

Thus the results of the single South African study are well within the range of international measurements and, indeed, are at the lower end of the scale. This finding was confirmed in another comparison, by Terblanche et al (1995). Thus the data described above will be used for present purposes: a central estimate of 947 μgm^{-3}, with a low estimate of 474 μgm^{-3} and a high estimate of 1 421 μgm^{-3} (50% on either side of the actual measurement). A wide variance between the central and other estimates is appropriate given that the quantity of wood consumed by households varies across the country, with lower amounts being consumed in warmer regions where space heating requirements are lower. Notwithstanding the uncertainty inherent in this sector, sufficient information exists to classify this externality as a Class One impact.

2.3 Morbidity and mortality from paraffin poisoning

2.3.1 *The extent of the paraffin poisoning problem*

One of the major advantages of paraffin as a source of energy for unelectrified households is that it is widely available throughout the country, via a distribution network which is informal and flexible enough to cater to the needs of low-income consumers. As mentioned earlier in this chapter, poor households are able to purchase small quantities of paraffin at most times of the day and night, from informal *spaza* shops in townships. This flexibility is simultaneously an advantage insofar as low-income households have access to and can afford the fuel,[12] and the cause of a major health hazard in the form of paraffin poisoning.

The latter hazard results from the practice of dispensing paraffin in small or irregular quantities, most commonly into empty beverage containers. Not surprisingly, infants in the home frequently mistake the paraffin for juice or water, with the result that they accidentally ingest the paraffin; this form of trauma is most common in infants between the ages of 12 and 36 months (De Wet et al 1994, Ellis et al 1994).

Paraffin poisoning is not a notifiable health problem and there is no single data source regarding the incidence of paraffin poisoning on a national basis. However, there have been several regional or micro-scale studies, as well as household surveys which have reported cases of paraffin poisoning. At a national scale, medical researchers have estimated that at least 16 000 children are hospitalised each year, yielding a rate of poisoning of about 30 cases per million litres of paraffin sold (Yach 1994: 717).

Average annual paraffin consumption has been estimated at about 180 litres per household; total household consumption in South Africa is about 520 million litres per annum (McGregor 1994: 27,28), which equates to about 2.9 million households using paraffin at present. This represents about 60% of unelectrified households in South Africa and is therefore probably on the conservative (low) side, since most unelectrified households use paraffin. Based on a hospitalisation rate of 30 cases per million litres consumed, a total of 15 600 cases per annum could be expected to occur - close to Yach's estimate.

This estimate can be corroborated by considering micro-scale household surveys which sometimes also report cases of paraffin poisoning. In one health and safety study of unelectrified households in Gauteng, it was found that of 2 124 people surveyed, about 6.5% had experienced incidents of paraffin poisoning, and that some had even experienced deaths in their households from the same cause (Terblanche et al 1992b: 68). It was reported in the

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[12] Which is not to say that paraffin is cheap when purchased in small quantities: unit prices tend to be far higher than average.

same study that paraffin poisoning represented a major cause of hospital admissions. In a follow-up study by the same authors, it was found that 4% of a sample of 1 240 rural children and 1% of 430 urban children had experienced incidents of paraffin poisoning (Terblanche et al 1993: 95). In another survey conducted amongst over 1 600 unelectrified households in the Eastern Cape, 3.1% of households reported having suffered a case of illness due to paraffin ingestion; furthermore, death had resulted in ten households (IPR 1993 in Van Horen 1994: 35). Thus between 1% and 6.5% of paraffin-using households in these surveys have reported cases of poisoning.

Applying an incidence range of 1% to 6.5% to the estimated 2.9 million households using paraffin, produces a range of 29 000 to 188 500 cases of poisoning nationally. The period over which these occur is unclear, since in each of the quoted surveys, respondents were asked whether they had ever experienced such incidents, so the replies probably refer to a period of several years. Assuming that the average recall period is five years, and that each household had no more than one case of poisoning, then the annual rate suggested by this data is in the range of 5 800 to 37 700 hospitalised cases per year.

Based on the above data, the central estimate of Yach will be used, that is, 16 000 cases per annum. Further, low and high estimates of 5 800 and 37 700 cases per annum will be used for purposes of estimating external costs.

Of importance also, are data on the effects of poisoning. National averages such as those above, obviously mask regional and localised variances, but it is clear that the incidence of poisoning is highest in unelectrified areas (Roberts et al 1989: 23) and more so in rural areas than urban areas (Ellis et al 1994: 727). Most medical reviews confirm that paraffin poisoning is the most common cause of childhood poisoning in South Africa (Ellis et al 1994: 727, De Wet et al 1994: 735, Korb & Young 1985: 228). Of importance for present purposes, however, is the fact that not all cases of paraffin poisoning are reported, and of those which are reported, only a portion are hospitalised. In general, it appears that a large portion of reported cases are treated as outpatients (that is, not admitted overnight). Treatment regimes reported by various studies are summarised in Table 5.1.

For purposes of quantifying the rate of morbidity and mortality from paraffin poisoning, it can therefore be estimated that just under half of reported cases are treated as outpatients, and that the average admission period of the remainder is 2.4 days. Furthermore, approximately 1.3% of cases are assumed to result in death. Based on the central estimate of 16 000 cases per year, this amounts to 208 deaths per annum, with a low and high estimate, respectively, of 75 and 490 deaths per annum. This information is sufficient for the externality valuation exercise in Chapter Six; hence this is classified as a Class One externality.

Table 5.1 Treatment regimes for paraffin poisoning reported in various studies

Study	% outpatients	% admissions	Average admission days	% fatalities
Roberts et al (1990)	78%	22%	-	0.74%
De Wet et al (1994)	46%	54%	1.1	-
Ellis et al (1994)	20%	80%	2.21	
Crisp (1986)	-	-	2 - 5	1.4%
Korb & Young (1985)	-	-	-	<1%
Joubert (1990)	-	-	2.8	2.1%
Average	48%	52%	2.4	1.3%

2.4 Effects of fires and burns

In addition to the above health hazards, unelectrified households also face relatively high risks of injury, death and loss of property through accidental fires. Fires are frequently caused by accidents with candles, paraffin stoves, gas stoves and sometimes wood fires, and can be particularly devastating in high-density informal settlements where occupancy rates are high, and dwellings are constructed with flammable wood and cardboard materials.

As in the cases of air pollution and paraffin poisoning, there has been no systematic study to-date of the incidence and effects of energy-related fires on a national basis. There have, however, been several smaller studies which have investigated the issue. In general, their results suggest that, while the number of burns cases might be lower than the number of paraffin poisoning cases, their individual effects can be more serious, including a greater number of deaths. At the same time, there is greater variation in the severity of burns and in the length of stay in hospital. Treatment can include, in addition to time spent in general wards, time in the Intensive Care Unit (ICU), physio-therapy, plastic surgery, as well as subsequent visits to hospitals. The costs of these accidents are therefore not insignificant.

Dealing firstly with deaths from burns, a study by Kibel et al (1990b: 398) of injury-related mortality in South African children from 1981 to 1985, found that burns (from all causes) were one of the top four causes of mortality in children, accounting for 11% of all deaths, or about 310 per year. Notably, this study excluded the TBVC territories[13] and therefore would seriously understate the actual mortality figures for the country. On the other hand, the figure includes burns from non-energy causes, which may include causes such as scalding from hot water and cooking fluids (De Wet et al 1977: 969). The portion attributable directly to energy consumption (rather than cooking or heating) must therefore be established.

In another study, of the total 358 childhood burn deaths admitted to the Salt River State Mortuary in Cape Town from January 1990 to December 1991, domestic accidents, mainly related to cooking or heating, were responsible for 21% of child deaths (Lerer 1994: 169). Further, housefires accounted for 75% of burn fatalities; the bulk of these were caused by accidents with candles. Together, it has been estimated that around two-thirds of burn fatalities are due to non-electric forms of energy (Lerer 1995: 26). Using the data reported by Kibel et al (1990b) this translates into 208 fatalities due to accidents with energy carriers. Given that this figure excludes the TBVC territories, it will be used as the low estimate for the national fatality rate. For the central estimate, recent census data can be used: in 1990, there were 2 646 deaths caused by burns amongst black South Africans (Lerer 1995: 23). If two-thirds of these were caused by energy-related accidents, then there were approximately 1 773 fatalities in 1990. This should also be adjusted to account for the higher level of electricity use in 1994, and correspondingly lower use of candles, paraffin and gas: [14] based on an increase in electricity access from 33% in 1990 to 44% by the end of 1994, the number of fatal burn incidents can be expected to have decreased from 1 773 to 1 330 per annum. This will be used as the central estimate for purposes of quantifying the external costs in the next chapter.

For the high estimate, the central estimate will be increased by 20% - this is an arbitrary amount, but it appears unlikely that there could be an upward deviation of the same size as the lower deviation. Hence the upper estimate of energy-related burn fatalities is 1 596 per annum for 1994. Since there are sufficient data to permit a reasonable estimate, this is classified as a Class One externality.

Turning to burns injuries, similar sources of data exist. One study of injuries to children, from the Red Cross Children's Hospital in Cape Town for the period 1984 to 1989, found

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[13] The former Transkei, Bophuthatswana, Venda and Ciskei homelands, which because of their nominal `independence' from South Africa, were removed from official government statistics.

[14] This assumption is a simplification, although it is reasonable to assume that electricity substitutes almost entirely for candles (used for lighting) and to a large extent, for paraffin and gas - these are the main causes of accidental fires and burns.

that burns accounted for 11% of all injuries treated during that period, equivalent to an average of 1 247 per year (Kibel et al 1990a: 387). Of these, 69% were treated as outpatients and the remaining 31% were admitted for more lengthy periods. In a study by Hudson et al (1994: 252), accidents with primus stoves were found to account for 12% of total admissions to the burns unit.

However, there are insufficient published data to allow for national estimates to be made. Consequently, some additional data were collected for purposes of this thesis, from admissions records for the three main hospitals in the Cape Metropolitan Area which, collectively, are likely to treat the vast majority of serious burns patients in the Peninsula. Hospital admissions records generally include data on the causes of injury, including whether burns were caused by candles, paraffin and gas. For those cases where no cause was specified in admissions records, a pro-rata apportionment has been made to all the specific causes. These data are summarised in Table 5.2.

Table 5.2 Summary of energy-related burns admissions in Cape Metropolitan Area, 1994

(Source: hospital admissions records)

Hospital	Outpatients	Admissions	Ave. admission days
Red Cross			
- Trauma Unit	26	20	4.5
- Burns Unit	-	74	4.5
- Convalescent Home	-	21	22.3
Somerset			
- Burns Unit	-	51	5.6
Tygerberg	-	68	22.6
Total/weighted average	26	234	11.6

Thus in 1994, there were at least 234 admissions to these Cape Town hospitals for treatment of burns where the cause of the injury was positively attributed to candles, paraffin or gas accidents. These cases accounted for 26% of total burns-related admissions in the three hospitals; this estimate represents the minimum possible number of cases, given that not all patients in the Peninsula are likely to have been referred or transferred to one of these burns units. Moreover, less serious burns are probably not reported as comprehensively.

These data can be extrapolated to produce an estimate of the number of burns cases admitted to hospitals nationally. Using the 234 admissions derived above for the CMA, and given that there were approximately 600 000 people living in unelectrified households in that area in 1994[15] - who can be safely assumed to have been at risk of such fires - then the injury rate for the CMA was 39 per 100 000 unelectrified people. In 1994, there were some 22.4 million people in unelectrified homes throughout South Africa; this yields an estimated number of hospital admissions in the whole country, of 8 736 for 1994. Using the same average number of admission days as for the CMA, then there were approximately 101 338 admission days due to energy-related fires in South Africa in 1994.

Whilst this would probably be a lower estimate for the CMA as a whole, it will be taken as the central estimate on a national basis, for the reasons that, firstly, the incidence of fires and burns in rural areas is likely to be lower than in urban areas, because of the lower densities and less flammable nature of houses outside of urban areas. Secondly, usage of paraffin and gas is higher in the Western Cape than in other provinces such as Gauteng, North-West, Northern Province and Mpumalanga (where coal and wood are commonly used). The effect of these two factors would therefore be to reduce the national average compared to the CMA. For purposes of the economic valuation in the following chapter, the central estimate is 8 736 admissions during 1994; the low and high estimates will deviate by 33% from this, amounting to 5 853 and 11 619 respectively. For the present, this externality is classified as a Class One impact.

A final impact to be considered in relation to the effect of fires, is on the property of fire victims. When fires occur, particularly in informal settlements, the effects are frequently devastating, with shacks being razed to the ground in a matter of minutes. Once again, there is no national register or systematic form of record-keeping about the incidence of such fires. Nonetheless, some indication can be obtained of the problem's significance, by reviewing secondary sources of data. For instance, a review of a local newspaper, which reports mainly on the northern parts of the Cape Metropolitan Area, *(Die Burger)* revealed articles in which at least 978 shacks were said to be destroyed by fires in 1994. The same reports said that at least 19 people had lost their lives in those fires. The bulk of these fires occurred in just two informal settlements: Marconi Beam, and Town B in Khayelitsha. Given that this newspaper probably did not report on fires occurring in other parts of the Cape Peninsula, the above figures cannot be said to provide a complete picture of fires damages in the wider area.

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[15] Calculated from Eskom's Statistical Yearbook (1994: 62) - which reported that there were 98 683 unelectrified houses in the CMA - using 6 persons per household, and rounded up to 600 000. This is likely to be a conservative estimate, as informal settlements are probably under-counted.

Consequently, it would be problematic to attempt to extrapolate these figures to a national basis, especially since the underlying source is hardly scientific.

Thus, although the costs of property damages may be significant in aggregate, and particularly in relation to the income levels of affected households, there is insufficient information to allow this externality to be quantified. Accordingly, it is classified as a Class Two effect.

2.5 The natural-environmental effects of wood scarcity

The large majority - some 80% to 100% - of rural households rely heavily on fuelwood supplies to meet their cooking and heating needs. Fuelwood contributes about 8% of total primary energy consumption in South Africa (Eberhard & Trollip 1994: 45), which although lower than other African countries, is highly significant in the rural economy. Patterns of consumption are also complex; before describing the main externalities, therefore, the next section will summarise the state of knowledge about fuelwood scarcity in rural areas.[16]

2.5.1 *Fuelwood scarcity in South Africa*

Little doubt exists that fuelwood is becoming increasingly scarce in many rural areas. Beginning with a study of energy consumption in rural villages by Best in 1979, numerous studies have found situations of biomass scarcity in rural areas. Best's study included three rural villages: Malefiloane near Mokhotlong, Lesotho, Jozanna's Nek near Sterkspruit in the Transkei and Mashunka near Tugela Ferry in Kwazulu (1979: 5). Only in the latter area were fuelwood supplies abundant, and consequently annual wood consumption in the areas of scarcity was considerably lower, with women in these villages remembering that wood has become scarcer and that standing trees have become smaller (ibid: 25,71). In his paper, Best stressed that firewood consumption cannot be singled out exclusively as the major cause of fuel scarcity: `the forces of agriculture, veld burning, overgrazing and settlement play dominant roles along with firewood collection' (ibid: 71).

A study by Gandar in the Mahlabatini District of Kwazulu found that in spite of a large increase in population density during the period 1956 to 1975, the density of wood cover in part of the study area had increased by 38% due to shrub encroachment in rangeland (n.d.: 4.5). The number of tall trees, however, had decreased by 50%, partly as a result of the clearing of land for farming. The other major reasons for the cutting of live trees were their use in the construction of huts and kraals, and their use in bush fencing. Gandar noted that

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[16] The Department of Mineral and Energy Affairs recently commissioned a study to synthesise the state of knowledge about biomass resources in South Africa; unfortunately, however, its report had not yet been published by the end of 1995.

during that period, the amount of dead wood available was certainly adequate to meet firewood demand. His observation was that people were prepared to walk long distances to gather dead wood before resorting to cutting down living trees for firewood.

These observations were consistent with the findings of a study by Liengme in a well-wooded rural area in Gazankulu (1983: 245). Her results indicated that, while firewood was the major use for wood collected in the area, most of the live wood which was cut, was intended for building purposes. Further, wood supply was adequate at that stage, although it was expected that a combination of factors would threaten the sustainability of supply: population increases, fuel gathering for adjacent towns, escalating demand for agricultural land and wasteful harvesting methods (ibid: 255).

In all of these studies, fuelwood collection was not seen as a primary cause of wood scarcity. This finding, however, was not confirmed in a study by Eberhard (1986), which included surveys of six rural villages (in addition to five peri-urban areas). In all rural areas of this study, respondents reported that wood was becoming increasingly scarce. The focus of the study on energy consumption patterns meant that other causes of wood consumption, such as for use in building, or land clearing, were not explicitly addressed. However, the fact that green wood was observed in headloads of fuelwood (ibid: 40) suggests that in these cases, fuel usage was at least partially contributing to fuel scarcity.

A study by Griffin et al (1992) from the Wits Rural Facility, of six rural settlements in Gazankulu, one of which was home to Mozambican refugees, found that wood was the main source of energy in all 424 households surveyed. While some of the settlements appeared to have sufficient supplies of wood available in the vicinity, a demand-supply modelling exercise indicated that, even in the best-case scenarios, present consumption patterns of fuelwood would not be sustainable in the long-term (ibid: 126). Many households were forced to use dung (against their preferences), or to purchase wood from vendors, even though income levels were extremely low. Again, it was not possible to apportion wood collection between energy and other needs, although measurements of the quantities of wood in buildings and other structures such as fences, indicated that a considerable quantity of wood stayed out of the cooking fire (ibid: 69).

A study by Aron et al (1991: 89) was the first to attempt to provide an aggregated picture of national wood consumption and supply. The study utilised a conventional gap analysis based on per capita wood consumption data which had been collected in a number of micro studies in the past decade. These average consumption rates were extrapolated across the total rural population in the homelands, to estimate total fuelwood consumption for 1980. Based on projected population growth rates for the ensuing 20 years, estimates were made of consumption in 2000. An estimate of the annual sustainable fuelwood supply in the homelands was also made, which yielded a slight overall surplus of fuelwood in 1980, but a large deficit by 2000. Whilst the study acknowledged the criticisms normally addressed at such analyses of fuelwood gaps, it nevertheless stated that natural woodland in the homelands would be entirely denuded by 2020 `if demand were to remain at constant levels' (ibid: 96).[17]

In spite of the usual limitations of this kind of calculation, the study served at least two important purposes: firstly, it was the first attempt to provide an overall picture of the national fuelwood situation, thereby opening the way for improved data collection exercises in the future. Secondly, its dire conclusions have made a strong impact on energy and development planners, even if they have tended to focus excessively on the projection of complete deforestation by 2020. For instance, the Department of Mineral and Energy Affairs has subsequently established a large project, the Biomass Initiative, the first phase of which entails a national research and data collection effort, leading to subsequent implementation phases.

2.5.1.1 The consumption of dung and crop residues

Traditionally, discussions of fuelwood scarcity and deforestation also address the use of dung and crop residues for energy needs. The conventional view is that the burning of dung and crop residues represents a loss of fertiliser and nutrient value for the soil. The conventional view of dung is also that it is the least preferred fuel, which is used by households with the lowest incomes. In terms of the `energy ladder' model of household fuel usage, consumers move through successive stages corresponding to some level of modernisation, with dung at the bottom of the ladder and electricity at the top (Smith 1988: 30). While a full critique of the energy ladder model need not be provided here (see Eberhard & van Horen 1995: 66-72), one point is relevant: namely that it is not merely the level of income or wealth which determines what fuels people will use, but also the availability of various energy sources. For instance, a large-scale study of dung usage in the former homelands by Bembridge et al (1992: 51) found that the quantity of dung used was greatest for those households with access to the most cattle - generally the wealthiest households. In other words, availability of dung was the key determinant of its use as a fuel,

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[17] Contemporary thinking has shifted away from this `fuelwood gap' type of modelling (where differences between demand and supply are extrapolated into the future), to a more complex approach in which fuelwood use is seen as one of many competing demands on resources and in which social, economic and political mechanisms play stronger roles in the management and consumption of resources. Refer to Van Horen & Eberhard (1995) or Van Horen (1994: 37-40) for a fuller treatment of this debate.

contrary to the view implied by the energy ladder concept, which is that only the poorest households use dung. The point is that the relative availability of various energy sources is the key determinant in peoples' energy mixes, and that this does not bear any fixed relation to variables such as income.

Bembridge et al (1992: 49-52) found that dung was widely used for various domestic purposes in the homelands, with fuel accounting for 86% of the quantity used, followed in order of importance by floor cleaning, manure, plastering, decorating and medicinal purposes. Per capita use of dung for fuel varied widely between areas, and averaged 408 kg per annum. Similarly, Eberhard (1986: 45) found that a high proportion of households in the six rural villages surveyed by him, used dung and crop residues to supplement their fuel needs. Mean per capita consumption of dung was lower than in the study quoted above, ranging from 53 to 231 kg per annum. In one village, dung provided approximately 21% of the total energy (ibid: 55). This was considered to be a symptom of increasing fuelwood scarcity (ibid: 111). The environmental effect of this, however, was considered to be negligible. An estimation of total national annual dung production (from cattle, sheep, goats, pigs and chicken) showed that dung consumption for cooking and heating was less than 1% of total dung production (ibid: 46-47). Dung was usually collected from the area closest to the home, and was still used extensively as a fertiliser in the fields. Gandar (1984: 7) also found that domestic use accounted for only about 1% of total livestock dung production in the Mahlabatini area.

The assumption that combustion of dung and crop residues will lead to a decline in soil fertility, therefore appears to be an exaggeration giving rise to unwarranted alarm. Dung which is dry enough to be burnt at the fireplace, has generally lost most of its nitrogen and is therefore not particularly effective as a fertiliser (Foley 1988: 7). The analysis by Bembridge et al suggests that the effect on agricultural production of removing dung for domestic purposes is not significant (1992: 67). In reality, the problem of declining soil fertility is frequently related to structural causes of instability in agricultural systems, such as increasing population densities, which place greater pressure on available agricultural land, thereby shortening the fallow period during which soil fertility is renewed. In South Africa, over-use of land is a result also of the state's resettlement policies carried out in terms of its `grand apartheid' plans (Cooper 1991, Wilson & Ramphele 1989).

It would therefore be simplistic and misleading to link poor soil fertility or low agricultural productivity, with household energy use patterns, when, in fact, they are more directly linked to the political, social and economic forces which have shaped the rural environment. Likewise, it is necessary to look beyond mere calculations of wood consumption and supply, to the complex array of political-economic and social factors which influence human-environment relationships.

2.5.2 *Impacts of wood scarcity on the natural environment*

Wood scarcity is partly a physical phenomenon afflicting the natural environment. Its effects on the natural environment depend on a number of factors. Firstly, where the wood is dead, its collection has a small effect on remaining ecosystems (Gandar 1982: 148), apart from a decrease in the nutrient value of decomposing organic matter. In many parts of rural South Africa, it appears that the scarcity of dead wood means that people have begun to cut living branches and trees, particularly where insufficient labour is available to absorb the extra time required for travelling to distant supplies of dead wood.

Secondly, the ability of natural ecosystems to sustain wood collection, depends on the system's type: for instance, savanna woodland has considerable ecological resilience, whereas indigenous forests are ecologically more fragile (Gandar 1984: 11). Clearly, the impacts of fuelwood collection will vary from area to area, depending on the vegetation structures. In South Africa, no systematic study has yet been performed which matches biomass consumption with the ecological resilience of the natural systems from which the resources are drawn.

The present discussion is not the place for a scientific analysis of all environmental impacts of biomass use. However, some major impacts are highlighted below in the context of the specific environmental elements affected by fuelwood scarcity: the land, water and hydrology, fauna and flora.

2.5.2.1 *Land and soil*

The removal of trees does not necessarily lead to increased soil erosion, although when combined with overgrazing of ground cover, it can have serious consequences. It has been estimated that adequate tree cover can reduce the potential for soil erosion by 50%, due to protection offered by the tree canopy, the leaves it drops and the denser herbaceous cover underneath (Gandar 1982: 149). Rates of soil loss in parts of South Africa are very high, especially in rural areas with high population densities and high annual rainfall. Again, it would be simplistic and incorrect to attribute this erosion to wood collection for energy and other domestic needs. Rather, its causes lie (inter alia) in inappropriate agricultural practices on commercial farms, the high population densities associated with the unequal distribution of land, and the pressures on the land which result from a struggling agricultural system. Poverty has the effect of constraining peoples' choices and of shortening their time horizons, so that, for example, overgrazing by livestock may denude the vegetation to the point where destructive processes of erosion are set in motion. Evidence of this is readily apparent in many parts of Kwazulu-Natal and the Eastern Cape, where population densities are relatively high.

The removal of wood for energy and other needs is therefore one of many factors which contributes to land degradation. Attempts to quantify its relative contribution have not been made, and anyway, it is not clear that such efforts would be worthwhile given that such a complex range of political and social forces shape rural land-use patterns.

2.5.2.2 Water and hydrology

Where woodlands have been denuded, it is possible that water flow patterns will be dramatically altered. During heavy rainfall, discharge rates will be higher, with the converse applying in dry periods (Gandar 1982: 149). The result is greater variability in flow rates from dry to wet seasons, with a less effective `sponge effect' which evens out the peaks and troughs in runoff. Cooper (1988) has argued that this process resulted in far greater damage in the floods of 1987 and 1988, which followed a prolonged drought. When the rains arrived, river catchments had been denuded of ground cover, silting of rivers and dams was very high, and the land was generally unable to absorb the heavy rains. The river levels were therefore higher than they would otherwise have been, and so the silt-laden floodwaters broke the banks of many rivers and small dams, aggravating the flood damage which would have occurred anyway.

On a less dramatic scale, the clearing of wood from gullies to which bushes and trees are confined in higher grassy areas, may lead to severe gully erosion because of the unchecked runoff (Gandar 1982: 149). Once the process has begun, eroded gullies rapidly become deep dongas, which severely affect the potential of the land. The area lost for agricultural purposes is significant, and moreover, the dongas dissect landscapes, making (especially mechanical) transport hazardous, time consuming, or impossible. There have been no estimates of the amount of productive land lost in this way nationally, but casual observation in many rural areas suggests that this may not be insignificant.

2.5.2.3 Fauna and flora

Apart from the obvious loss of vegetation which occurs when wood is removed from living trees and shrubs, complex interactions occur in the affected ecosystems on both macro and micro scales. Habitats of large animals may be degraded to the extent that they can no longer survive at all, while smaller organisms may be able to cope with the stresses imposed on them. The loss of habitats, especially in areas where heavy land-use causes total deforestation, represents an obvious threat to the diversity of biological systems. This issue - biological diversity - was a central issue at the Earth Summit in 1992. Although the biodiversity in South Africa's natural woodlands is of a far smaller order than that in tropical rain forests, the issue takes on greater significance in the context of the many uses people have found for natural woodlands and their components: medicinal, cultural, ornamental, and many others.

From the above description of effects, it is apparent that the impacts of wood consumption may be significant, but that quantification of these effects is not possible given the state of existing information. Consequently, the natural environmental effects of wood scarcity is classified for present purposes as a Class Two impact.

2.6 Impacts of wood scarcity on the social environment

Apart from the physical impacts of wood scarcity, a number of significant social impacts arise, many of which have particular gender dimensions. The collection of fuelwood and its consumption in the preparation of meals are time-consuming tasks, which usually rest with women. To the extent that increased scarcity occurs, this frequently places additional pressure on women, since their social position in most rural communities is that of nurturer and provider (Makan 1994).

The effects of increasing scarcity of fuelwood are felt in many ways by rural households. Firstly, more time is spent in the collection process. In Gandar's study conducted in the Mahlabatini district of Kwazulu, the average time taken to collect one headload of wood varied from about 2.5 hours in the valley lowveld areas where wood supplies were more abundant, to 4.5 hours in the high grassland areas (1984: 3). Households required between 2 and 3 headloads per week, so that total time spent gathering fuelwood varied from 6.75 hours to just over 9 hours. These are merely averages, and in one extreme case, he encountered a group of women who spent 9.5 hours walking a total of 19 km to collect headloads weighing 40 kg each (ibid: 4). The people interviewed were certain that the distances they travelled to collect wood had increased; this perception was most emphatic among older women who could remember when wood supplies were closer to their homes. Similar results were found in Eberhard's study of six rural villages: average distances walked were from 6 to 9 km, two to three times per week, requiring 2.6 to 6.2 hours each time (1986: 33-34). This yields an average of between 5.2 hours and 18.6 hours per week. Bembridge and Tarlton (1990: 89) reported similar results from a study in an area of Ciskei.

It is clear from these and other studies that women spend considerable amounts of time collecting fuelwood. Unfortunately, no longitudinal studies have been undertaken to measure trends in these variables in particular communities, although an indication has been obtained from the perceptions of the people themselves. In the majority of cases, it has been reported, especially by older women, that the time required for collecting wood has increased, as wood has become more scarce. In many respects, this is a very unproductive use of time. If fuel collection is included as part of food preparation, then in many cases, households spend more time preparing food than growing it: clearly, not a productive situation (Gandar 1984: 5).[18]

On the other hand, it has been argued that the time spent collecting wood is relevant only insofar as it relates to the availability of labour (Leach & Mearns 1988: 17) - in other words, to labour's opportunity cost. If surplus labour is abundant, then it may not matter that collection of fuelwood takes longer and longer; conversely, even if wood is abundant, the scarcity of labour may pose a very serious problem. Once again, this observation widens the analytical focus, from a concern only with an energy issue (availability of fuelwood), to a concern with broader developmental issues (labour availability). The division of labour between men and women is therefore also an important determining factor in households' coping strategies.

It is difficult to make generalisations regarding the availability of labour in South Africa's rural areas: the specifics vary considerably from one area to another. Nonetheless, the high degree of male absenteeism in many places which act as sending areas for the migrant labour system, means that, in general, women carry out most domestic and productive responsibilities. This includes fetching water, and tending of fields and other agricultural work which may otherwise have been done by men. As a consequence, it is frequently the case that women's labour time is fully utilised, and therefore increased time spent collecting wood is not easily accommodated. Although there remain considerable uncertainties regarding the empirical data about wood collection, there are sufficient data to permit an estimate to be made in Chapter Six of the external costs of wood scarcity - hence this is a Class One externality.

In addition to the extra time expended on wood collection, there are other social impacts. Firstly, as the length of wood collecting journeys increases, so women attempt to economise on time, by collecting larger headloads of fuelwood in order that fewer trips can be made. Again, no studies have been made of individual communities, comparing changes in the size of headloads over time, but comparisons between communities with different wood availability supports the view that headloads increase in size as distance travelled increases. Bembridge and Tarlton, for example, found a large headload of 33 kg being carried by a small, old woman whom they estimated could not have weighed more than 40 kg herself (1990: 89). Table 5.3 below summarises the average number of headloads per week, the average mass of all headloads and the heaviest headload measured in a number of studies.

[18] Not only time, but physical energy also is expended in the collection of wood.

Table 5.3 Comparison of average mass of headloads, average collection trips per week per household, and heaviest bundles in various studies.

Area and Study	Average mass of headload	Average trips per week	Heaviest headload
Amatola Basin, Ciskei (Bembridge & Tarlton 1990)	24 kg	5.1	36 kg
Mahlabatini, Kwazulu (Gandar 1984)	38 kg	2.3	-
Gazankulu (Liengme 1983)	30 kg	3.5	67 kg
Malefiloane, Lesotho	21 kg	4.4	32 kg
Jozanna's Nek, Transkei	15 kg	3.6	34 kg
Mashunka, Kwazulu (Best 1979)	21 kg	3.4	40 kg

While some variations from the trend occur, these data support the argument above, that households making fewer (and normally longer) trips to collect fuelwood, carry heavier loads. This is (literally) a physical burden borne mainly by women, and imposes extra stresses on their physical well-being; in extreme cases, it is not impossible that spinal damage may occur (Gandar 1982: 151). By any standards, carrying a headload of these sizes over long distances and rough terrain, is a physically strenuous activity.

Researchers have identified several coping strategies adopted by households when faced with scarce wood supplies. Often these involve economising in domestic activities, such as improving the efficiency of fireplaces or cooking with more than one pot at a time. Sometimes, however, the responses are likely to decrease the overall welfare of the household. In some cases, fewer meals may be cooked, food may be cooked for shorter times, eating patterns may shift towards food which requires less energy to cook (and which may be less nutritious) or food may be cooked in bulk and stored until a later meal; the lack of refrigerators in most rural homes means that food which has been stored for a few days may become unfit for consumption (Eberhard 1986: 38, Bembridge & Tarlton 1990: 92).

Less time may also be spent on socially necessary functions, including recreation, spending time with children, family and friends, and cultural activities. The pressure on traditional social structures, caused by numerous other factors too, may be further heightened for this reason.

Table 5.3 Comparison of average mass of headloads, average collection trips per week per household, and heaviest bundles in various studies.

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Less time may also be spent on socially necessary functions, including recreation, spending time with children, family and friends, and cultural activities. The pressure on traditional social structures, caused by numerous other factors too, may be further heightened for this reason.

Finally, where households are unable to compensate for increasing scarcity in the ways described above, they may be forced to spend more of their incomes on commercialised fuels, including wood being sold by vendors. One of the striking findings of Eberhard's study of six rural villages around the country, was the extent to which fuelwood had been commercialised (1986: 35). In five of the villages, almost half of the households surveyed purchased some of their wood requirements. In a survey conducted for Eskom of three rural areas being evaluated for electrification planning purposes, Gandar (1989) found that in all three areas, expenditure on energy constituted a high proportion of monthly income: from 15% to 17% (1989: 13,26,39). From 68% to 86% of wood users purchased all their wood, and accordingly, expenditure on wood made up a large portion of the monthly energy budget. The commoditisation of fuelwood has also been reported by McClintock (1988: 44) in a study of the Upper Tugela Location in Kwazulu, and by Eberhard and Dickson (1991: ii) in a study of two areas in Bophuthatswana.

For purposes of the externality valuation exercise in the next chapter, sufficient data exist to make an estimate of the external costs of wood scarcity - it is therefore classified as a Class One impact.

2.7 Greenhouse gas emissions from household energy consumption

The consumption of energy by households leads, unavoidably, to the emission of greenhouse gases, particularly carbon dioxide. In principle, it is important that there be no double-counting of emissions, which would occur if GHGs were counted both at the stage of electricity generation (as in Chapters Three and Four) and at the stage of final energy consumption in households. By focusing on low-income, unelectrified households in this analysis, this potential problem is avoided.

Although there have been government-sponsored studies undertaken of South Africa's GHG emissions (so-called inventory studies), these have not reported emissions at a sufficiently disaggregated level to identify emissions from the household sector. Nonetheless, other studies have estimated the quantity of emissions by low-income households, based on average energy and carbon contents of wood, coal and paraffin, their combustion efficiencies, and based on national fuel consumption quantities (Cowan 1992: 4). From this calculation, it was estimated that total carbon dioxide emissions were 25.4 million around 1990, made up as follows:

- wood emissions - 18 million tons;
- coal emissions - 6 million tons;

- paraffin emissions - 1.4 million tons.[19]

This breakdown is important insofar as the net impact of emissions from these sub-sectors differs. In the case of the hydrocarbon fuels - coal and paraffin - it is clear that the GHGs would not have been liberated if these fuels had not been consumed. However, in the case of fuelwood, the carbon content of wood follows a natural cycle, beginning with the growth phase when trees are net sinks (absorbers) of carbon dioxide, through to the stage when carbon is liberated after the death of the standing stock of trees, during decomposition. Thus the combustion of fuelwood simply speeds up the natural cycle of GHG emissions. The relevant question then is whether the stock of fuelwood is being depleted faster than it regenerates. As noted earlier, there are some parts of the country where demand exceeds sustainable supply, although the opposite holds in other parts of the country. In a study of national supply and demand balances, Aron et al (1991: 89) found that there was a slight excess of supply over demand around 1980.[20] Even if this balance has shifted into a slight net deficit situation by the mid-1990s, it is unlikely that this is significant in aggregate, and therefore it would be inappropriate to attribute damage costs to GHG emissions from fuelwood consumption.

Thus, for purposes of calculating damage costs from GHG emissions, a figure of 7.4 million tons will be used, reflecting the combustion of coal and paraffin by households.

2.8 Summary of external effects

The externalities described in this chapter are summarised in Table 5.4 below. It is évident that, of the seven categories, six have been classified as Class One impacts, with only two impacts not having sufficient information to permit their quantification (one category, namely the effects of fires and burns, had both a Class One and a Class Two impact).

These Class One externalities are valued in economic terms in the following chapter.

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[19] GHG emissions from other energy sources, such as gas, candles and batteries, were not considered because these fuels make up a small fraction of total energy demand and are not material in relation to wood, coal and paraffin.

[20] The authors projected historical trends, and suggested that a deficit would arise and increase over time, until supplies were fully depleted by 2020. Problems with this kind of gap analysis have been referred to earlier.

Table 5.4 Summary of potentially significant environmental impacts and their classification in this study

	Class One	Class Two	Class Three
Air pollution from coal combustion	✓		
Air pollution from wood combustion	✓		
Morbidity & mortality from paraffin poisoning	✓		
Effects of fires and burns	✓	✓	
Effects of wood scarcity on natural environment		✓	
Effects of wood scarcity on social environment	✓		
Greenhouse gas emissions	✓		

Chapter Six

The external costs of household energy use

1. Valuation of externalities in South Africa's household energy sector

In this chapter, the six Class One externalities described in the previous chapter, are valued in economic terms:

- air pollution from coal combustion;
- air pollution from wood combustion;
- morbidity & mortality from paraffin poisoning;
- effects of fires and burns;
- effects of wood scarcity on the social environment; and
- greenhouse gas emissions by households.

In so doing, a similar methodological approach as in Chapter Four is adopted, where the opportunity cost method is used to value health effects.

1.1 Valuation of air pollution impacts from coal combustion

It was estimated in Chapter Five that the incremental pollution concentrations to which people are exposed as a result of coal combustion is in the region of 272 μgm^{-3} for fine particulate matter (PM10). By applying the same dose-response functions as for power station emissions (refer Tables 3.10 and 3.11), it is possible to estimate a physical burden of mortality and morbidity. The final step in the damage function is estimating the value of the economic externality.

The physical health impacts of the calculated pollution exposures on the 6 million people who are assumed to be exposed to this level, are shown in Table 6.1 (calculations are shown in Appendix 4).

The physical health outcomes in Table 6.1 can be translated into economic values, by applying the opportunity costs of each outcome (consistent with those used earlier, and as described in Appendix 2). The results of this calculation (in 1995 Rand values) are shown in Table 6.2.

The central estimate is that annual costs are in the region of R307 million, with the largest portion of this being attributable to acute and chronic respiratory illness (bronchitis). These amounts can also be translated into an external cost expressed in relation to units of delivered energy (joules); the results of doing so for this externality, together with the other Class One externalities described in this chapter, are summarised later in Table 6.11.

Table 6.1 Physical health effects resulting from particulate emissions by coal-using households

Health outcome	Unit	Low estimate	Central estimate	High estimate
Asthma attack	Occurrence-day	22 032	39 168	132 192
Acute bronchitis	Occurrence	427 950	855 899	1 283 849
Chronic bronchitis	Person	32 912	66 921	102 027
Outpatient/GP visit	Visit	522	1 061	1 583
Mortality	Death	10	17	24
Resp. Symptom day	Occurrence-day	326 400	750 720	1 142 400
Resp. Hospital adm.	Admission	29	54	78
Restricted activity	Occurrence-day	87 765	175 530	274 266

Table 6.2 Valuation estimates for annual mortality and morbidity burden from household coal pollution (1995 Rm)

Health outcome	Low estimate	Central estimate	High estimate
Asthma attack	4.6	10.8	45.6
Acute bronchitis	86.0	172.0	258.1
Chronic bronchitis	27.3	73.9	140.8
Outpatient/GP visit	0.1	0.2	0.4
Mortality	8.0	20.4	45.6
Resp. Symptom day	2.4	7.5	14.3
Resp. Hospital adm.	0.1	0.3	0.6
Restricted activity	8.3	22.1	43.2
Total	136.8	307.2	548.6

It should be noted again that these estimated costs include only the health effects of particulate emissions and not carbon monoxide, sulphur and nitrogen oxides, or ozone. Consequently, these results present only a partial picture of the health effects arising from coal combustion. Furthermore, they also exclude impacts of these emissions on the health of people living outside of coal-using areas: clearly, pollution emissions from townships affect the welfare not only of those residents themselves, but also of people in neighbouring areas. These effects are omitted from the above calculations.

1.2 Valuation of air pollution impacts from wood combustion

Pollution levels within rural households are known to be high - frequently higher than levels in coal-using urban households - but at the same time there is considerable uncertainty about actual levels across the entire rural population in South Africa. In Chapter Five, the low, central and high estimates of particulate exposure (for PM10) were 474 μgm^{-3}, 947 μgm^{-3} and 1 421 μgm^{-3} respectively. The wide range of estimates reflects this uncertainty. This uncertainty is particularly significant given that there is such a large number of wood-consuming people in South Africa - approximately 17 million - which means that any under- or over-estimates will be extrapolated over a large base.

Using the same dose-response functions as before, the physical health impacts of the calculated pollution exposures are shown in Table 6.3 (for calculations, see Appendix 4).

Table 6.3 Physical health effects resulting from particulate emissions by wood-using households

Health outcome	Unit	Low estimate	Central estimate	High estimate
Asthma attack	Occurrence-day	108 783	386 376	1 956 717
Acute bronchitis	Occurrence	2 113 001	8 443 088	19 003 635
Chronic bronchitis	Person	162 502	660 146	1 510 210
Outpatient/GP visit	Visit	2 579	10 464	23 432
Mortality	Death	48	159	360
Resp. Symptom day	Occurrence-day	1 611 600	7 405 540	16 909 900
Resp. Hospital adm.	Admission	145	531	1 160
Restricted activity	Occurrence-day	433 340	1 731 531	4 059 705

The results summarised in Table 6.3 reflect two underlying factors: first, the large number of people exposed to high pollution levels in rural areas. Thus, in the case of chronic bronchitis, the central estimate suggests that about one in 25 people will suffer from this long-term ailment, whilst in the case of acute bronchitis, approximately one in two people are likely to be afflicted every year.

The second relevant factor is that these calculations are based on assumed dose-response functions which were derived from North American studies. The limitations of transferring these functions to South African conditions were noted earlier, however, these problems become particularly acute when applied to the rural population. This is because assumed relationships between pollution exposures and medical treatment responses (admissions to hospital, visits to doctors or outpatient clinics, etc.) are less likely to be accurate in rural areas where incomes are often extremely low and health care facilities are remote. As a result, certain health outcomes - particularly those involving visits to doctors and hospitals for the treatment of relatively minor respiratory ailments - may be over-stated by the above calculations. On the other hand, however, it is likely that other health outcomes will be understated due to the absence of adequate treatment and health care services: notably mortality effects.

The net effect of these under- and over-statements is difficult to discern in the absence of empirical data on treatment ratios for various illnesses in South Africa. Consequently, no adjustments will be made to the above estimates. The effect of multiplying the above physical health outcomes by their economic costs, yields the results summarised in Table 6.4.

Table 6.4 Valuation estimates for annual mortality and morbidity burden from household wood pollution (1995 Rm)

Health outcome	Low estimate	Central estimate	High estimate
Asthma attack	22.5	106.6	675.1
Acute bronchitis	424.7	1 697.1	3 819.7
Chronic bronchitis	134.6	728.8	2 084.1
Outpatient/GP visit	0.4	2.1	5.9
Mortality	38.4	190.8	684.0
Resp. Symptom day	12.1	74.1	211.4
Resp. Hospital adm.	0.6	3.1	8.4
Restricted activity	41.0	218.2	639.4
Total	674.3	3 020.8	8 128.0

The central estimate is that annual costs are just over R3 billion, nearly a factor of ten higher than the estimate for coal-derived pollution. This reflects, firstly, the higher pollution exposures used in this calculations, and secondly, the larger population at risk. As with the previous externality, the most significant health outcome in economic terms is acute bronchitis.

1.3 Valuation of morbidity and mortality from paraffin poisoning

It was estimated in Chapter Five that approximately 16 000 cases of paraffin poisoning are hospitalised each year, with low and high values of 5 800 and 37 700 respectively. Furthermore, it was reported that approximately 48% of cases are treated as outpatients, and that the remaining 52% are admitted for an average of 2.4 days each. The mortality rate is around 1.3% of admissions, yielding estimates of 75, 208 and 490 deaths per annum.

Although there are obviously different treatment regimes for paraffin poisoning cases, depending on the specific circumstances of each, these do not vary as widely as for other externalities (such as burns). Direct costs incurred in treating poisoned infants are usually not highly material, since treatment generally involves observation and administering of minor medication. Average costs are estimated to be R650 for admitted patients, and R146 for children treated in outpatients sections (refer to Table B2 in Appendix 2). The weighted average treatment cost of a paraffin poisoning case is therefore R408. By comparison, it was estimated in a study by the Medical Research Council that the average cost of treating poisoning victims in Western Cape hospitals in 1990 was R256 (De Wet et al 1994: 735); escalating this to 1995 Rands, yields an amount of R412. This is very close to the value calculated here, and so the figures of R650 and R146 can be applied to calculate the cost of paraffin poisoning. The results of doing so are shown in Table 6.5.

Table 6.5 Valuation estimates for annual mortality and morbidity burden from paraffin poisoning (1995 Rm)

Health outcome	Low estimate	Central estimate	High estimate
Poisoning cases admitted	3.8	10.4	24.5
Poisoning cases in outpatients	0.8	2.4	5.5
Mortality due to poisoning	60.0	249.6	931.0
Total	64.6	262.4	961.0

the central estimate suggests that about one in 25 people will suffer from this long-term ailment, whilst in the case of acute bronchitis, approximately one in two people are likely to be afflicted every year.

The second relevant factor is that these calculations are based on assumed dose-response functions which were derived from North American studies. The limitations of transferring these functions to South African conditions were noted earlier, however, these problems become particularly acute when applied to the rural population. This is because assumed relationships between pollution exposures and medical treatment responses (admissions to hospital, visits to doctors or outpatient clinics, etc.) are less likely to be accurate in rural areas where incomes are often extremely low and health care facilities are remote. As a result, certain health outcomes - particularly those involving visits to doctors and hospitals for the treatment of relatively minor respiratory ailments - may be over-stated by the above calculations. On the other hand, however, it is likely that other health outcomes will be understated due to the absence of adequate treatment and health care services: notably mortality effects.

The net effect of these under- and over-statements is difficult to discern in the absence of empirical data on treatment ratios for various illnesses in South Africa. Consequently, no adjustments will be made to the above estimates. The effect of multiplying the above physical health outcomes by their economic costs, yields the results summarised in Table 6.4.

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Resp. Symptom day	12.1	74.1	211.4
Resp. Hospital adm.	0.6	3.1	8.4
Restricted activity	41.0	218.2	639.4
Total	674.3	3 020.8	8 128.0

It is clear from these results that the mortality effects dominate the total valuation of this externality, even though the costs of morbidity are in themselves not insignificant: for instance, the study by the MRC referred to above, found that the direct costs of treatment (that is, excluding transport costs and opportunity costs of absence from work) were of a similar order of magnitude to the costs of providing vulnerable communities with child-resistant paraffin closures (De Wet et al 1994: 735).

1.4 Valuation of effects of burns and fires

The average number of hospital admissions resulting from accidents with energy carriers was estimated in Chapter Five to be 8 736 per year; low and high estimates were 5 853 and 11 619 respectively. Admissions were estimated to have an average duration of 11.6 days. Furthermore, the number of deaths attributed to accidental fires was estimated to be in the region of 1 330 per annum, with low and high figures of 208 and 1 596 respectively.

Treatment for burns varies considerably, depending on the severity of the accident and the kinds of injuries sustained. In one study of burns sustained in primus stove accidents, Hudson et al (1994: 252) reported that average treatment costs were in the region of R500 per admission-day, including the costs of blood transfusions and theatre operations.[1] Excluded from this were costs of transport, absence from work, and pain and discomfort. When these are included, the central estimate of the treatment cost is in the region of R8 417 per case (see Appendix 4, Table B3).

The valuation results for this externality, based on the data described above, are summarised in Table 6.6.

Table 6.6 Valuation estimates for annual mortality and morbidity burden from fires (1995 Rm)

Health outcome	Low estimate	Central estimate	High estimate
Treatment of burns cases	49.3	73.5	97.8
Mortality due to burns	166.4	1 596.0	3 032.4
Total	215.7	1 669.5	3 130.2

[1] The figure in the paper was R1 200 (divided by 24 days equals R50 per day); but according to discussions with the author, this was a mis-print and should have read R12 000, yielding an average cost of R500 per day in 1992, which is more plausible.

It is evident that the scale of this external effect is highly material, particularly since it leads to a large number of deaths. Moreover, these calculations have not included the value of lost property and other assets in the same fires (a Class Two impact), and therefore understate the true scale of this problem. Its significance in relation to other externalities is summarised later in this chapter.

1.5 Valuation of wood scarcity on the social environment

The time spent by rural households (usually women) collecting wood for fires varies widely across the country, but usually falls within the range of 5.2 hours to 18.6 hours per week (refer Chapter Five). To the extent that fuelwood is collected from natural woodlands and by-passes commercialised markets, the time expended in its collection represents part of the social cost. To the extent that all or most of this social cost is not accounted for in any price relationship attached to fuelwood, it can be thought of as an external diseconomy or externality. At some point in history, the time required to collect wood was probably very small or negligible. As the supply of fuelwood has diminished, however, the opportunity cost of collecting it has increased, causing the social cost of wood collection to rise. It is this unaccounted-for cost which is under consideration: although this effect does not fall into the conventional conception of externalities (like air pollution from power stations), it is true that the social cost of wood collection diverges from its private cost, and this difference (the `externality') which needs to be quantified.

The valuation of this externality, however, is complex. Two approaches may be used in performing such a valuation (these were described briefly in Chapter Two). The first involves using surrogate markets to value the fuelwood which is collected from natural woodlands (not unlike the hedonic pricing method); the second entails imputing a shadow price for the time spent collecting wood, based on the opportunity costs of labour time.

With reference to the first valuation method, the valuation of fuelwood consumed from natural woodlands or communal sources will yield a wide range of values, because of the variability of wood prices from one place to another. Average wood prices reported in a number of rural energy use surveys conducted in recent years (summarised in Ward 1994), varied in a range from almost zero to R1.20 per kg; most commonly, however, prices fall into the range from R0.04 to R0.30 per kg (ibid, Gandar 1994, Viljoen 1994). In principle, these prices should include consumers' valuation of the labour time required to harvest and collect fuelwood, in addition to their valuation of the exploited natural resource, its transport and distribution. In practice, however, these differences cannot safely be assumed to reflect consumers' willingness to pay for the commodity based on their individual preference sets. In fact, these rural wood markets exhibit few of the qualities of an efficiently-operating market, and the prices are most likely related less to expenditure of labour time in wood collection, than to the abundance of wood in areas from which it is collected, and to the proximity of those supply sources to areas of demand. Thus this valuation method is not used for present purposes.[2]

Under the second valuation method, a shadow price is attributed to the labour expended by rural households on fuelwood collection. As noted earlier, Leach and Mearns have suggested that the central issue is not so much whether fuelwood is scarce, but whether labour used to collect it is scarce (1988: 17). This raises difficult questions about the marginal opportunity cost of (mainly women's) labour in rural areas. An appropriate measure in this regard could be average earnings of self-employed people working in the informal sector. A source of such data was a major World Bank-supported survey of over 8 000 households across the country (SALDRU 1995), from which it was estimated that 43.2% of self-employed persons earned between R0 and R150 per month, and a further 32.9% earned from R151 to R500 per month. These statistics, however, refer to both urban and rural households, and no breakdown for this split was available. Assuming that the self-employed were evenly distributed across urban and rural areas, and that earnings in rural areas were at the lowest end of the scale, then the earnings in the lowest band should approximate the earnings from self-employed rural people (about 45% of the population lives in rural areas - very close to the portion of self-employed in the lowest income band). Taking the mid-point of this income band, average earnings of rural households can be estimated at R75 per month.[3] Assuming an 8 hour day and 20 working days per month for waged farmworkers in the formal sector, then the shadow wage translates into an hourly rate of R0.47.

The time spent collecting wood, as noted above, is in the range of 5.2 to 18.6 hours per week, with an average of 11.9 hours. Further, if an average of one woman per rural household collects wood, this equates to about 3.2 million people. Based on these figures, the annual opportunity cost of fuelwood collection for the whole country is as summarised in Table 6.7.

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[2] For interest sake, this calculation would yield a valuation of R218 million to R1.6 billion per annum (based on national fuelwood consumption of about 5.45 million tons in 1980 (Aron et al 1991: 90)).

[3] By comparison, average agricultural wages on commercial farms were about R232 per month in 1991 (calculated from Central Statistical Services, in Horner 1994:11).

Table 6.7 Valuation estimates for social impacts of fuelwood collection (1995 Rm)

	Low estimate	Central estimate	High estimate
Labour time expended	406.7	930.7	1 454.7

These amounts represent between 4% and 10% of South Africa's commercial agricultural GDP (SAIRR 1994:236) - supporting the intuitive view that fuelwood collection plays an important role in the rural economy.

By accounting for the value of fuelwood collection in economic terms, it also becomes apparent that increasing scarcity of fuelwood supplies carries a significant marginal cost. If, for instance, the average time required to collect fuelwood increases by 30 minutes per week, this translates into an additional opportunity cost of about R39 million per annum (based on the central case). Essentially, this means that labour time to this value, has to be reallocated from agricultural or other activities to fuelwood collection.

1.6 Valuation of greenhouse gas emissions by households

It was established in Chapter Five that household emissions of carbon dioxide from coal and paraffin combustion are approximately 7.4 million tons per annum. Using the same damage cost estimates as in the case of electricity generation, that is, estimated costs of R5, R22 and R44 per ton of carbon dioxide for the low, central and high cases respectively, the results shown in Table 6.8 are produced.

Table 6.8 Valuation estimates for CO_2-induced climate change damages

	Low estimate (Rm)	Central estimate (Rm)	High estimate (Rm)
Emissions from coal	30.0	132.0	264.0
Emissions from paraffin	7.0	30.8	61.6
Total	37.0	162.8	325.6

The results shown in the table are summarised below, together with those of the other Class One externalities quantified in this chapter, both in total Rand amounts and as a cost per unit of energy delivered.

2. Summary of valuation results

The valuation results obtained in the preceding sections of this chapter are summarised in Table 6.9 in total Rand values; the table also includes a (subjective) assessment of the level of uncertainty associated with each estimate.

Table 6.9 Summary of valuation results for household sector's Class One and Class Two externalities in 1994 (Rm 1995)

	Level of uncertainty	Low estimate	Central estimate	High estimate
Air pollution from coal combustion	moderate	136.8	307.2	548.6
Air pollution from wood combustion	high	674.3	3 020.8	8 128.0
Paraffin poisoning	moderate	64.6	262.4	961.0
Effects of fires and burns[4]	moderate	215.7	1 669.5	3 130.2
Social impacts of wood collection	moderate	406.7	930.7	1 454.7
Greenhouse gas emissions	high	37.0	162.8	325.6
Total		1 535.1	6 353.4	14 548.1

It is evident from this comparison that the amounts involved are significant in absolute terms; the largest externality (in the central case) is caused by air pollution impacts from fuelwood combustion. Also highly significant are the effects of fires and burns, due mainly to the deaths which arise.

These amounts can also be expressed as an average external cost in relation to the units of energy which are delivered to consumers. Of the stages in the fuel cycle - primary energy, delivered energy and final energy - the relevant variable for present purposes is delivered energy: the quantity of energy which arrives at the door of the household, in other words after it has undergone initial transformation, but before it is used in any appliance. Data are required regarding the absolute amounts of energy which are delivered for each of the main

[4] This externality excluded the damages to physical property and assets of affected households.

externality-producing sectors described in this chapter. These data are summarised in Table 6.10.

Table 6.10 Quantities of delivered energy in respect of Class One externalities[5]

Externality	Number of households	Annual consumption per h/hold	Total annual consumption	Joules per unit[6]	Total Gigajoules per annum
Coal pollution[7]	1 million	3 300 kg	3.3 Mt	25.8 MJ/kg	85.1 * 10^6
Wood pollution[8]	3.2 million	3 531 kg	11.3 Mt	13.5 MJ/kg	152.6 * 10^6
Paraffin poisoning[9]	2.9 million	180 l	520 Ml	35.6 MJ/l	18.5 * 10^6
Fires and burns					
- candles	3.9 million	-	-	-[10]	-
- paraffin	2.9 million	180 l	520 Ml	35.6 MJ/kg	18.5 * 10^6
Wood collection	3.2 million	3 531 kg	11.3 Mt	13.5 MJ/kg	152.6 * 10^6
Greenhouse gases					
- paraffin	2.9 million	180 l	520 Ml	35.6 MJ/kg	18.5 * 10^6
- coal	1 million	3 300 kg	3.3 Mt	25 MJ/kg	85.1 * 10^6

Finally, these energy consumption data can be combined with the economic valuation estimates, to yield an average external cost per unit of delivered energy. These calculations are summarised in Table 6.11.

[5] Amounts may not add up in the table due to rounding effects.

[6] From National Energy Council (1990: 99).

[7] Coal consumption data from Palmer Development Group (1995).

[8] Wood consumption data from Gandar (1994: 9).

[9] Paraffin consumption data from McGregor (1994: 27, 28).

[10] The amount of energy produced by candles is negligible.

Table 6.11 Average external costs of household externalities in relation to quantity of delivered energy

Externality	Rands per Gigajoule		
	Low estimate	Central estimate	High estimate
Coal pollution	1.61	3.61	6.45
Wood pollution	4.42	19.80	53.26
Paraffin poisoning	3.49	14.18	51.95
Fires and burns			
- candles			
- paraffin	11.66	90.24	169.20
Wood collection	2.67	6.10	9.53
Greenhouse gases	0.36	1.57	3.14
Total	24.21	135.50	293.53

The values in this table represent the average external cost of each of the Class One externalities analysed in this chapter; expressed in this form, they can be compared with the external costs of electricity generation: this is done in the next chapter. Chapter Seven also includes an assessment of the limitations and weaknesses inherent in the present analysis.

Chapter Seven

Conclusions and policy implications

One of the key questions posed at the beginning of this thesis concerned identifying the energy-environment problems which result in the most significant costs to society. It was hypothesised that the costs in the household sector may be the most significant. Having undertaken an empirical estimation, in the previous four chapters, of a number of external costs in the household consumption sector and the electricity generation sector, it is now possible to provide an answer to this question. This is done in the next section of the chapter. Having done so, the subsequent section steps back from the details of the quantitative analysis and considers some of the limitations of the analysis, from two perspectives: the first is a technical one, whilst the second involves a more philosophical critique of the environmental economic analysis constituting the body of the thesis. The latter perspective therefore addresses the second question posed at the beginning of this thesis, namely, how useful is a conventional environmental economic analysis in developing energy-environment policies? Finally, the chapter draws out some implications of the preceding analysis for South Africa's energy and environment policy.

1. A comparison of the external costs of electricity generation and household energy consumption

Both the electricity generation and the household energy consumption sectors are responsible for the production of significant external costs under current conditions. These costs can be evaluated on the basis of total, average and marginal costs, which is done below. In each case, the base year for present purposes is 1994.

1.1 Total external costs

The total external costs of the Class One impacts quantified in previous chapters are presented in Figure 7.1 in total 1995 Rands (for the central estimate).[1]

[1] Where reference is made in this chapter to `total costs' or `total external costs', this refers to the `total' only of *Class One* externalities quantified in this study - thus it omits certain costs not quantified here.

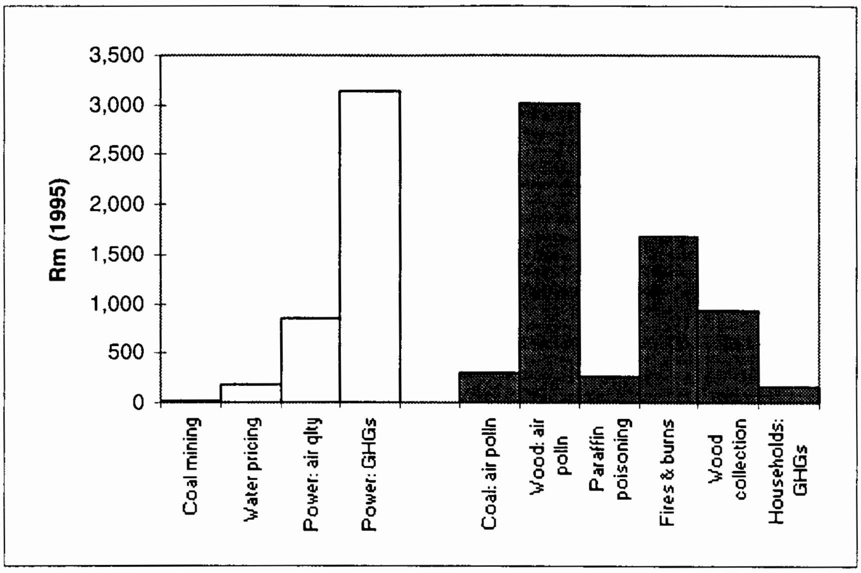

Figure 7.1 Central estimates of total external costs in the electricity and household sectors in 1994, Rm 1995

Several observations may be made regarding this comparison. Firstly, the most significant externality of those quantified is the estimated damage from greenhouse gas emissions by the electricity generation sector. The central estimate of damage costs taken from the literature was R22 per ton of carbon dioxide emitted in 1994, which yielded total damages of some R3.1 billion for 1994: this represents 20% of Eskom's total revenue for 1994 (Eskom 1995a: 41) - a highly material amount. Implications of this for policy are considered later.

Secondly, the relative significance of greenhouse gas damages are the virtual inverses of each other: in the electricity generation sector, GHG damages are by far the most significant externality relative to the others, whereas in the household sector, the costs of GHG damages are amongst the smallest in relation to other externalities in that sector. This suggests, at the broadest level, that the degree of control or internalisation of non-GHG externalities is relatively greater in the electricity supply industry than it is in the household consumption sector.

Thirdly, the total costs shown in Figure 7.1 give no indication of the range of uncertainty inherent in the quantification process. It is important that this not be overlooked, given the number of assumptions and simplifications which have been made along the way. An indication of the spread of total costs per annum is shown in Figure 7.2. The figure shows the aggregate of Class One externalities for the two energy sectors.

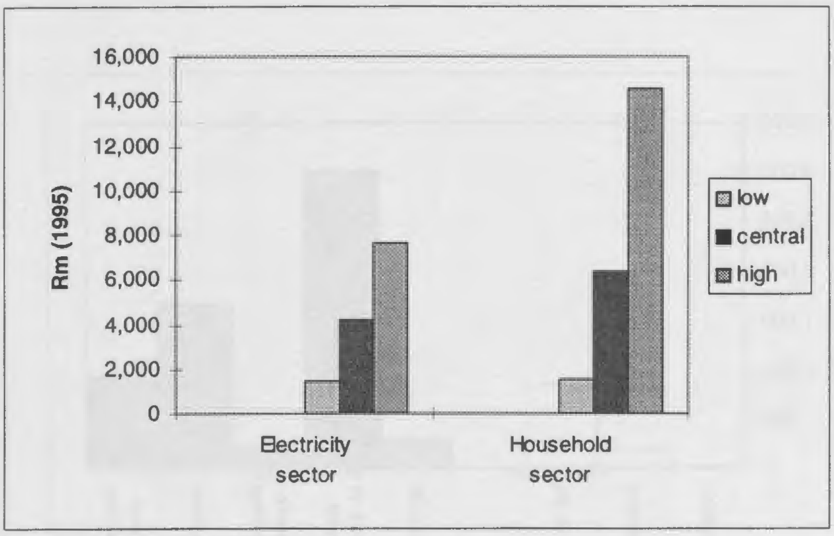

Figure 7.2 Low, central and high estimates for total external costs in the electricity and household sectors, total Rm 1995

It is evident that the spread of external costs in each sector, as well as the absolute orders of magnitude in each case, are broadly similar. In general, external costs in the household sector are higher than in the electricity sector for each of the three estimates. In the central estimate, in particular, external costs are in the region of R6.4 billion in the household sector, some 51% in excess of those in the electricity sector (R4.2 billion). This provides an answer to the question posed at the beginning of the thesis, and taken at face value, confirms the first hypothesis presented then: that is, external costs *are* highest in the household sector.

In addition, the spread of costs in the household sector is wider, as shown below:

	low	central	high
electricity sector	35%	100%	181%
household consumption	24%	100%	229%

These are significant deviations, which underline the uncertainties inherent an economic analysis of this nature. This point will be addressed again shortly.

The first observation which was made in relation to Figure 7.1 above, was related to the significance of damage costs attributable to GHG emissions from the electricity sector. The aggregation of this externality with the others has the effect of distorting the overall picture, especially in view of the higher level of uncertainty implicit in existing valuation studies of GHG damages, as well as the unique policy environment surrounding the climate change issue. Consequently, it is important to evaluate the external cost estimates also without GHG damages. Figure 7.3 presents each of the remaining Class One externalities in total Rand values for the base year.

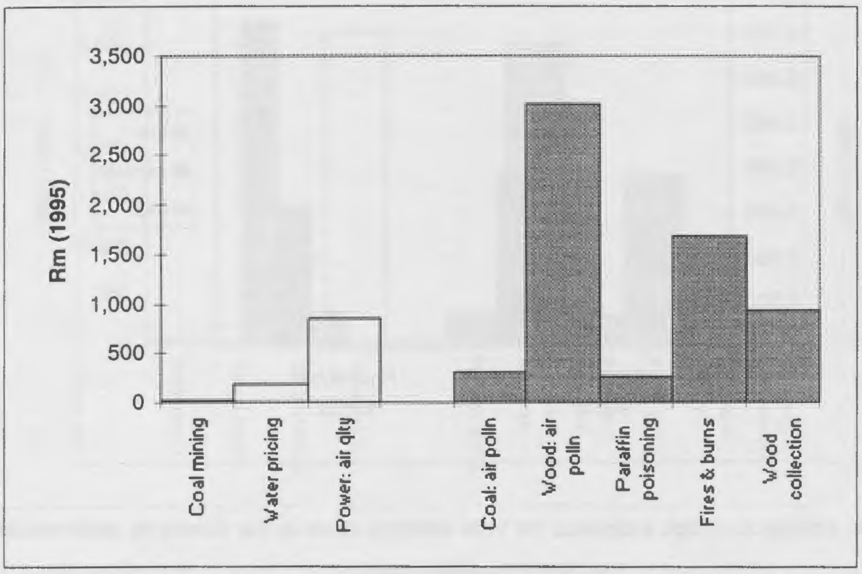

Figure 7.3 Central estimates of total external costs in the electricity and household sectors, excluding GHG damages, Rm 1995

The picture when external costs arising outside of South Africa are excluded is substantially different from Figure 7.1 which included GHG damages.[2] It is evident that the four largest externalities are now as follows: first, air pollution costs from wood combustion in rural households; second, the health and opportunity costs of fires and burns from accidents with candles and paraffin, third, the opportunity costs of fuelwood collection, and fourth (just lower than the previous impact), the air pollution costs from power station emissions. This has important policy implications, and in some respects constitutes a second answer to the question posed at the beginning of this thesis: of the external costs imposed on the country by the energy sector, the three most significant are related to the consumption of wood, candles and paraffin by low-income households; collectively, these are far in excess of the external costs arising from the power sector. The relative significance of household and electricity sector externalities is shown graphically in Figure 7.4, which shows the aggregate of Class One external costs for the low, central and high estimates, both without GHG damages.

[2] Of course a very small portion of the estimated GHG damage costs may be incurred by South Africa, so this geographical distinction is not strictly correct; nonetheless, this does not substantially detract from the point being made.

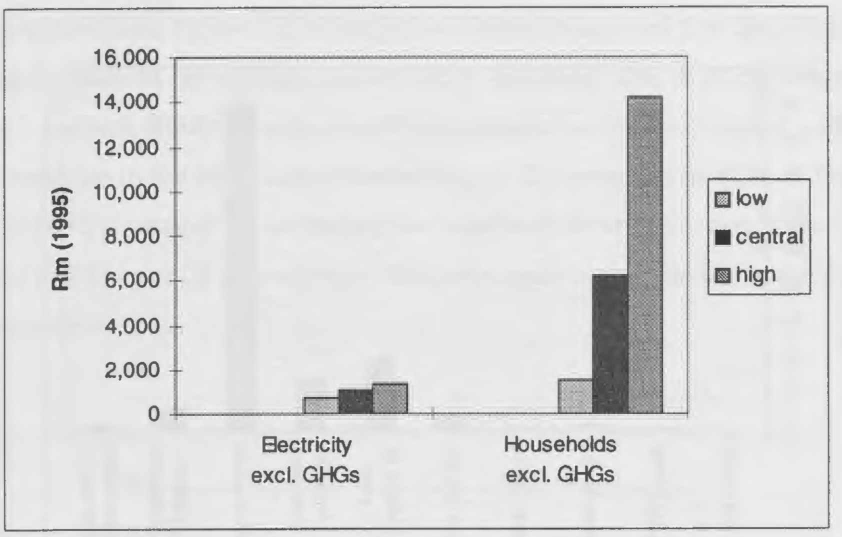

Figure 7.4 Low, central and high estimates for total external costs in the electricity and household sectors, excluding GHG damages, total Rm 1995

It is apparent that when GHG damages are excluded, the absolute scale of total external costs for the base year (1994) in the electricity production sector are dwarfed by those in the household consumption sector, by a factor of almost six in the central case. Even the lowest estimate of damages in the household sector exceeds the highest estimate of damages from electricity generation by approximately 12%. This suggests that as a first priority, energy and environment policy-makers should seek to identify abatement options in the low-income household sector, since the benefits of abatement are potentially much greater than in the electricity industry. Thus, when GHG impacts are excluded, the first hypothesis posed in Chapter One is confirmed even more strongly: namely that the total external costs are substantially higher in the household sector than the electricity sector. For practical purposes, the treatment of GHG impacts is dependent on the international policy context, which at present does not require the internalisation of these costs. Nonetheless, the scale of potential costs is sufficient to underline the importance of this area for policy-making; some policy implications of these conclusions are addressed in more detail later in this chapter.

1.2 Average external costs

The comparison of total external costs, as above, is useful for purposes of identifying the environmental impacts with the greatest social cost, but is constrained by the fact that there is no common physical denominator in the comparison. It is therefore useful to calculate average external costs, in other words, to compare the external costs in relation to a common unit, namely delivered energy (in Gigajoules). This comparison is shown in Figure 7.5.

A final observation from Figure 7.5 is that on the whole, it appears that the average costs are higher in the household sector than the electricity industry. This is more directly apparent from Figure 7.6 which shows average costs in aggregate for the low, central and high cases. The central estimate in the electricity sector is R8 per GJ compared to R136 in the household sector, and if GHG damages are excluded, the relative difference is even greater: just under R2 compared to R134 per GJ respectively. This once again highlights the relative significance of household externalities.

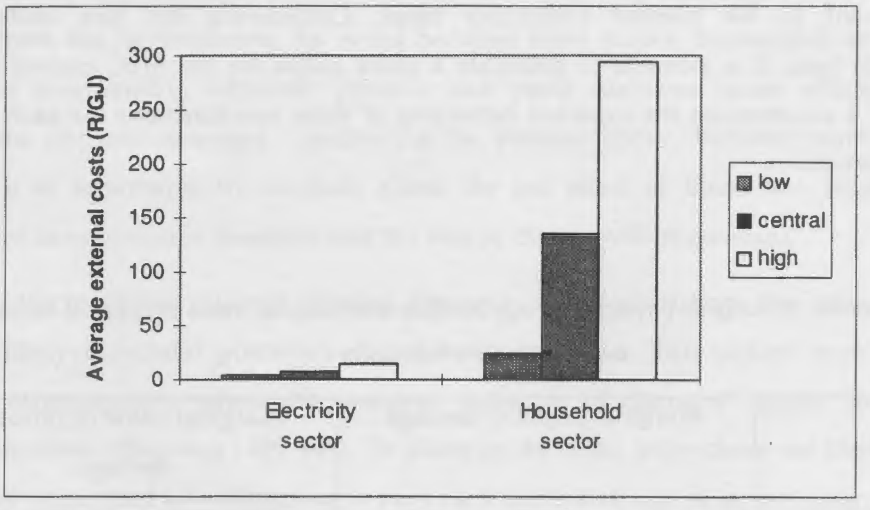

Figure 7.6 Average external costs for all Class One externalities in aggregate, in Rands per Gigajoule

Not to be overlooked in this analysis, is the wide range of quantitative results obtained in preceding chapters, and this is again evident from Figure 7.6, which shows that the range of uncertainty is wider in the household sector than in the electricity sector. The ratios of low, central and high estimates, based on average costs, are as follows:

	low	central	high
electricity sector	35%	100%	181%
household consumption	18%	100%	217%

1.3 Marginal external costs

Finally, it is important for purpose of economic decision-making to consider how these costs are likely to behave at the margin. Since the estimates in this study have yielded only a single estimate of total and average costs, corresponding to actual energy production and consumption in 1994, there is little empirical basis for making assumptions about the likely slope of a marginal external cost (MEC) function. Nonetheless, some inferences can be made with a view to identifying the approximate shape and slope of the MEC curve. This is perhaps more appropriate, once again, for purposes of making comparisons between these sectors, than for purpose of comparing with abatement options.

The MEC curve is conventionally assumed to be a positive strictly convex function of the level of pollution, usually reflecting two variables: marginal physical damage and marginal value of physical damage, both of which usually increase (Burrows 1995: 245). Although several cases exist in which non-convexities are likely to exist (ibid: 250-262), those are not directly relevant to the present externality cases. Considering the two underlying assumptions in turn, it is possible to postulate a likely shape for the MEC curves in this study. Table 7.1 summarises the expected behaviour of these two functions for each of the Class One externalities.

Table 7.1 Behaviour of marginal physical damage function and marginal value of physical damage for each Class One externality

	Marginal physical damage	Marginal value of physical damage
Coal mining	positive, constant	positive, constant
Water pricing	positive, constant	positive, constant, step increases
Power: air quality	shape of DR function?	positive, constant
Power: GHGs	positive, constant	positive, constant ?
Coal: air pollution	shape of DR function?	positive, constant
Wood: air pollution	shape of DR function?	positive, constant
Paraffin poisoning	positive, constant	positive, constant
Fires & burns	positive, constant	positive, constant
Wood collection	positive, increasing/decreasing?	positive, constant/increasing?
Households: GHGs	positive, constant	positive, constant

In most cases, it can be reasonably expected that these functions will be linear with a positive slope. Dealing first with the marginal physical damage functions, some exceptions to this general observation arise. In the case of the air pollution impacts from coal and wood combustion, and from power stations, the marginal physical damage function will be dependent primarily upon the shape of the underlying dose-response (DR) function. As noted in Chapter Three, there is little information on the likely shape of the dose-response function at relatively high levels of pollution, such as those encountered in South Africa, and therefore it was assumed to be linear. In the case of the physical damage from wood collection, two trends are likely to occur. Firstly, as fuelwood supplies become more scarce, women and children are likely to expend more time in the collection of given quantities of fuelwood. Since physical damages are measured, for this externality, in terms of labour time expended to collect wood, the damage function would have an increasing slope. On the other hand, physical damages from fuelwood collection may decrease as more wood is removed from the environment. As wood becomes more scarce, households are likely to respond by economising wherever possible and using fuelwood more efficiently, thus reducing the physical damages - countering the increase above. Without more empirical evidence, it is impossible to conclude about the net effect of these two tendencies. A conservative assumption is therefore that the rate of change will be constant.

Turning to the marginal value of physical damages, it is evident from the table that most cases are likely to exhibit positively-sloped linear functions. This differs from valuation results in other studies, where the marginal valuation of damages usually increases as pollution increases (Burrows 1995: 245). To illustrate the latter, individuals are likely to value more highly (measured by willingness to pay) each successive unit of air pollution they have to breathe. Whilst this condition may hold in instances where valuation methodologies are based on individual preference measures, this does not hold where valuation of outcomes is primarily based on opportunity costs. The opportunity cost of an additional case of bronchitis, for instance, is similar to the previous case.

The only exceptions to this general conclusion of a positively-sloping linear function arise, firstly, for water consumption in power stations, where step increases in the marginal economic cost will occur as the supply capacity of existing dams and water schemes becomes fully utilised and new schemes have to be constructed. The second possible exception arises in the case of fuelwood collection, where the opportunity cost of labour expended on fuelwood collection might increase over time, as more labour is expended on fuelwood collection and less is available for other productive uses: effectively, there may be increasing labour scarcity. However, this effect, if it occurs at all, is likely to be very small, since general levels of labour scarcity in rural areas are low.

The effect of this discussion is that the marginal external cost curve will have a positive slope, and is likely to be almost linear, with perhaps a slightly convex shape. Consequently, approximate MEC curves can be derived, as a function of units of energy produced or consumed. Indicative MEC curves for the central estimate in the two energy sub-sectors are shown in Figure 7.7.[3]

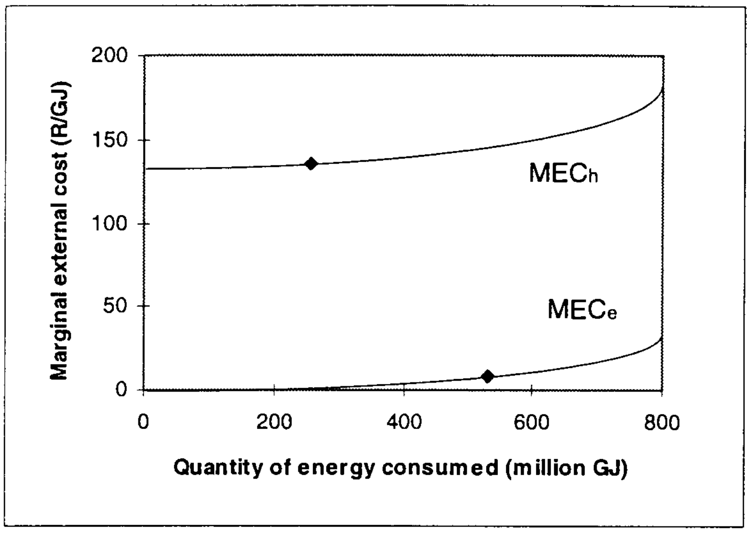

Figure 7.7 Indicative marginal external cost curves for electricity (MECe) and household (MECh) sectors

These MEC curves are indicative only, nevertheless they show clearly that the marginal external costs in the household sector are considerably higher than in the electricity sector. For policy-making purposes, these MEC curves also represent the marginal abatement benefits of intervention options which might reduce the external costs. Although this is beyond the scope of this thesis, these MEC (or marginal abatement benefit, MAB) curves could be compared against the marginal abatement cost (MAC) curves for alternative intervention options, so that the optimal mix of abatement alternatives could be selected.

This analysis represents further corroboration of the first hypothesis discussed previously: namely that the potential economic gains from focusing on household externalities are greater than those in the electricity sector. Furthermore, to the extent that a substitution occurs within low-income households, of electricity for coal, wood, paraffin and candles, then net environmental and economic benefits are likely to accrue. This is not to suggest that

[3] These curves are indicative only, especially as there is no data set corresponding to other quantities of energy consumption. Microeconomic theory suggests that the MEC and AEC curves would intersect at the minimum turning point of AEC. In the absence of data to the contrary, it can be assumed that this is the level of current output.

electricity is a substitute for all of these sources of energy in low-income households, but it does point to the additional benefits which will be associated with electrification.

2. A critical review of the present analysis

The preceding analysis has been directed towards assessing where in the energy sector the social costs are greatest. Subject to various limitations which will be described shortly, the hypothesis presented in Chapter One, namely that external costs are greatest in the household energy consumption sector, has been confirmed by the preceding empirical analysis - and even more so when damages from GHG emissions are omitted.

The second question which was posed in Chapter One, was whether the environmental economics discipline can make a meaningful contribution to the analysis of energy-environment policy options, and it is to this question that the discussion now turns. It was hypothesised that an environmental economic analysis can make an important contribution to the development of sound policy. The results produced by the preceding analysis have, it is suggested, already demonstrated the value of such an analysis. Nonetheless, it is important that its limitations are made explicit. This leads to the caveat attached to the second hypothesis: namely that there are a number of limitations inherent in an analysis of this kind, both in the South African context and in developing country contexts more generally.

This section presents a critique of the preceding analysis, using two categories of limitations: firstly, those concerning data weaknesses and technical constraints on the economic assessment of external costs, and second, more fundamental criticisms of environmental economic philosophies and methodologies. Whilst the former category essentially represents a set of caveats and provisos to the analysis, the latter is more challenging since it goes to the heart of some of the methods adopted. Each of these is considered in turn.

2.1 Technical limitations of the analysis

An inevitable component of quantitative externality studies is a certain level of uncertainty, and it is important in such analyses to make explicit all such weaknesses. At the most basic level, the calculated economic values are as good only as the input assumptions and information from which they are derived. The numerical results obtained and reported above are inevitably subject to biases, of which there are two main causes: omissions and uncertainties.

2.1.1 Omissions in the quantitative analysis

Given limited time and resources, as well as diminishing returns as the scope of a quantification exercise is widened, this study developed a classification system to make explicit which externalities were being excluded and why. Only those impacts which were potentially significant and for which sufficient information existed to perform an economic valuation, were quantified (Class One impacts). Consequently, a number of externalities, including some which are potentially serious, were reviewed but not quantified in this study.

Thus there is an unavoidable source of bias due to the omission of the Class Two and Three impacts. In the case of the electricity sector, these externalities included the following:

- chronic and acute illnesses experienced by workers on coal mines supplying Eskom;
- impacts of air and water pollution emitted by coal mines supplying the power stations;
- impacts on human health of air pollution originating from coal power stations' ash dumps;
- impacts of coal power station emissions and resultant acidic deposition, in terms of human health, damages to crops, forests, water supplies and other physical assets in the Mpumalanga Highveld and neighbouring regions;
- impacts of coal power station emissions on visibility conditions, particularly in the Mpumalanga Highveld;
- impacts of coal power station emissions into water courses;
- impacts of nuclear power stations on environmental quality and human health.

It is possible that if data existed to permit their economic valuation, some of these externalities would be significant. This is perhaps especially the case with the acidity impacts of power station emissions, which has usually been found to be a large component of total external costs in many other externality studies (see for example, the reviews in Lee 1995 and Van Horen 1995).

In the household energy consumption sector, there were also a number of omissions:

- impacts of air pollutants other than particulate matter (such as oxides of sulphur and nitrogen) on human health;
- impacts of all air pollutants in terms of reduced visibility, contribution to acidification and consequent damage to property and vegetation;
- impacts of fires, caused by accidents with candles, paraffin and the like, in terms of damages to houses, assets and physical property of affected people;
- impacts of fuelwood collection on the natural environment, through loss of biodiversity, contribution to soil erosion and depletion of natural capital.

It is entirely speculative to make any inferences as to the significance of these externalities in the absence of empirical data; nonetheless, indications are that these omissions introduce a bias into the valuation results which may not be immaterial.

2.1.2 *Uncertainties in the quantitative analysis*

With respect to uncertainties in evaluation of external costs in the electricity sector, there are several important considerations:

- Firstly, in the case of injuries and deaths occurring in the coal mining sector, data for the coal mining industry as a whole had to be apportioned between the main consumers, since data were not available specifically for the coal mines supplying Eskom. The effect of this may have been either to understate or overstate the results.

- Secondly, there is uncertainty over the applicability of dose-response functions derived in North America to South African populations. No epidemiological studies have yet derived these relationships for South Africa, yet these variables constitute an essential step in the damage function approach to valuing health effects. Any under- or over-statements in these dose-response functions would be compounded throughout subsequent steps in the valuation exercise.

- With regard to the valuation of health impacts of air pollution, there is some uncertainty regarding the atmospheric modelling approach used. The atmospheric models employed (contained in the EXMOD model and described in Appendix Three) used a Gaussian plume type of dispersion model to approximate the dispersion of emissions from power station chimneys. According to Eskom's air quality experts, actual conditions on the Highveld are not especially well-represented by this kind of model due to the presence of the inversion layer and mixing patterns (Turner 1995). However, there are no publicly-available atmospheric models designed specifically for South African conditions, and so this level of uncertainty is unavoidable.[4]

- Fourthly, there is a high level of uncertainty regarding the future global impacts of anthropogenic greenhouse gas emissions in the economic, social and environmental spheres, and the valuation estimates for GHG damages which exist in the literature are heavily qualified (and indeed, contested).

- Finally, there have been no previous economic valuation studies of environmental and health issues in South Africa, from the pricing of water to the value of human health and

[4] Eskom is in the process of developing such atmospheric models, but they are not yet complete.

mortality; consequently there is a shallow empirical base upon which this thesis has drawn.

Likewise, in the household sector, several uncertainties could not be avoided in the analysis:

- As in the case with power station emissions, there is uncertainty over the applicability of the dose-response functions to South African populations.
- Secondly, there is uncertainty inherent in describing `typical' cases of various health outcomes, both in terms of their likely treatment scenarios, and in terms of the costs associated with each scenario. The data used in this study were collected from and corroborated by health specialists working in the field, but there remains some uncertainty in generalising those observations to a national scale.
- Thirdly, the externality concerned with air pollution from rural wood-consuming households is based (apart from international comparisons) on only one study with a small sample of participants; thus its results may not be representative of conditions in other parts of the country.
- Fourth, as with the electricity sector, there are considerable uncertainties surrounding the future impacts of climate change.

These uncertainties have been accommodated, to an extent, by utilising a range of estimates rather than a single estimate, and by allocating an appropriate weighting to these estimates where relevant, so as to reflect the inherent uncertainty.

The possible effects of these omissions and uncertainties on the reported economic valuations, that is, whether the direction of the resultant bias will be to understate or overstate the externality valuations, are summarised in Table 7.2.

With the exception of the omitted Class Two and Three externalities, the overall effect of these biases is difficult to assess, since any one of them (for example, GHG impacts) could be large enough to more than offset all others. Nonetheless, it is important to note in all cases in this study, the chosen route has been to err on the side of understating external effects rather than the opposite. Thus there is a reasonable level of confidence that the range of quantified externalities does not overstate the minimum value of externalities in the South African energy sector.

Table 7.2 Summary of potential biases in this study due to omissions and uncertainties

Uncertainty or omission	Direction of bias on externality valuations		
	Under-stated	Unknown	Over-stated
Class Two & Three impacts omitted (see above)	x		
Coal mine accident rates: industry average vs. Eskom suppliers rates		?	
Atmospheric dispersion modelling		?	
Extrapolating results of only one study of wood pollution		?	
Dose-response functions: North American vs. South African data[5]		?	
Future impacts of GHG emissions		?	
Valuation of environmental & health impacts		?	
Categorisation of `typical health outcomes'		?	

2.2 A critique of the environmental economic framework

The rapid emergence of the environmental economics discipline in the 1980s and 1990s as an influential body of intellectual thinking, does not mean that all has been plain sailing in theory and practice. On the contrary, a number of important challenges to the discipline have appeared in the literature, particularly from the political economy tradition. Although not explicitly addressed thus far, many elements of these critiques have lain beneath the surface of the environmental economic analysis employed in this thesis (and at times, may have surfaced). Therefore for purposes of assessing the integrity and usefulness of the environmental economic framework, it is necessary to give consideration to these criticisms in this discussion. The critique is subsumed under the following four points, each of which is addressed in turn below:

- First, environmental economic analysis is too often divorced from the political and social contexts in which the problems arise.
- Second, policy prescriptions are reformist and managerial, treating the symptoms of environmental stress rather than the causes.

[5] The last four uncertainties apply to both the electricity and household sectors.

- Thirdly, environmental economic analysis is not especially well-equipped to accommodate distributional or equity considerations.
- Fourthly, the appearance of `scientific objectivity' in quantitative analyses can be misleading for policy-making purposes.

2.2.1 *The political and social context of environmental problems*

A relatively broad but fundamental criticism of conventional environmental economic analyses is that they generally take little account of the wider political and social contexts. As political economists sometimes phrase this, environmental problems are `socially constructed' - in other words, the environment is socially and politically intertwined with the development process, and conversely, with a process of underdevelopment (Redclift 1984: 130, Adams 1990: 83). Likewise, Dasgupta and Maler have highlighted the almost complete `disconnection' of development economics from the environments of `people about whom development economics revolves' (1995: 319). Linking environmental problems with their political contexts is a strong theme in a portion of the development literature (see, for example, Adams 1990, Redclift 1984, 1987, Blaikie 1985, Blaikie & Brookfield 1987).

The fairly general point made above can be given more substance by referring to practical examples of certain environmental problems discussed in this thesis. The question might be posed: accepting for now that the external costs of low-income household energy use are high (and certainly higher than in the electricity sector), *why* are these costs so high? If these costs were so significant, *why* is it that these costs have been incurred for several decades, without much effort by government to ameliorate them? Are these costs evenly *distributed* across society, and how do social power relations affect the distribution of these costs?

If these questions are considered at all, a conventional analysis would investigate the preference sets of those households, and particularly their willingness to pay for improved environmental conditions, in relation to their demand for other goods and services. It might be found that households' preferences lay elsewhere and that, because their WTP for environmental improvement was lower than the cost of supplying that improvement, it did not take place. Thus, for instance, it might be concluded that electrification (to the extent that this would reduce the rate of accidents caused by candles and paraffin) did not occur because even though households would have liked to reduce the risk of such accidents, their WTP for an electricity supply was insufficient to justify that investment.

Whilst this logic is not inconsistent, it provides only a partial answer to the questions above. A much fuller understanding can be gained by considering the role of political and social forces in causing and perpetuating these environmental problems. Thus national and local government politics in the apartheid era can be seen as a fundamentally important factor which perpetuated the poor environmental conditions in historically-black residential areas.

Several factors conspired to deny many households the option of securing an electricity supply. Firstly, responsibility for the supply of electricity to household consumers rested with local authorities rather than a public utility (such as Eskom). One of the cogs in the apartheid machinery was racially-separate local government, and given the lack of political legitimacy and the financial incapacity of black local authorities, this meant that local government was poorly positioned to respond to demand for electricity supplies or to environmental problems of the kind described earlier. Secondly, national government and public sector institutions such as Eskom served the interests not of disenfranchised black households, but of enfranchised white households, commercial farmers, industry and mines.[6] The result was that whether or not the demand for electricity supplies by low-income households was high, the politics of apartheid interfered to suppress this demand. One consequence of this suppression (of both economic demand and human rights) was the existence and persistence of high environmental costs in low-income black residential areas.

Thus one underlying reason for the high external costs in the household sector rests with the politics of apartheid. It is also instructive to consider why these costs (excluding GHG damages) are so much higher than the costs in the electricity sector. Again, this can be only partially explained by a higher demand for clean air (and clean electricity) by those affected by power station emissions. Another factor relates to the relatively greater political power held by those affected by power station emissions - commercial farmers, tourists in Mpumalanga province, even environmental non-governmental organisations - than consumers of polluting and hazardous energy sources in black townships. As suggested earlier in this thesis, concerns over pollution from power stations have enjoyed a much higher profile in the national press and policy-making sector than the `hidden' problems afflicting low-income households. The failure of environmental economic analyses either to ask `why' the present state of affairs exists, or if the question is asked, the failure to look at wider political and social causes, certainly limits the approach's analytical power.

Further limits also arise in the policy directions which are suggested by a conventional analysis. Typically, these involve `getting the prices right', often through the design and imposition a pollution tax to bring the private cost closer to the social cost, *a la* Pigou (1920). In the case of the household coal market, for instance, it might be suggested that polluting bituminous coal should be subjected to a pollution tax of sorts, to provide consumers with

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[6] This is evidenced perhaps most acutely by the policy through which Eskom was given subsidies to connect remote (white) rural farmers to the national grid, whilst not making connections to the homes of (black) workers on those same farms (Eberhard & van Horen 1995: 26).

the correct signals as to the real resource and health costs of their consumption patterns. In principle, the amount of this tax would be equal to that portion of the external costs which are not borne directly by the person buying the coal. Again, by itself, this logic is not faulty, but does not necessarily provide a policy-maker with the kind of proposals he or she seeks.

To simply impose a pollution tax on conventional coal in the household sector may correct for one cause of market failure by getting the price right, but could have inequitable social outcomes if no alternatives are introduced at the same time. It has been well-established in the literature that there are numerous economic, social and cultural reasons why households prefer to consume coal in the colder regions on the South African Highveld (refer to Chapter Five, Van Horen 1994: 72). Levying a pollution tax on this commodity when households have few alternatives which are affordable and which provide these multiple services,[7] would simply place an additional burden on the already-stretched budgets of low-income households, without necessarily engineering the shift away from coal to a cleaner fuel. In other words, a technically-correct solution (a pollution tax) would have inequitable results, without necessarily improving environmental conditions. Unless, therefore, the strategies suggested by a conventional analysis are measured against the social (and, similarly, the political) contexts into which they might be introduced, the policies run the risk of failing, or of conflicting with social and political objectives.[8]

Thus, if one is to extract something constructive from the criticism discussed in this section, then it would be that sound policy-making should rest not only upon the analysis and prescriptions of a conventional environmental economic approach, but that such an approach should be enhanced and strengthened by giving explicit and direct attention to the prevailing political and social contexts. This does not mean that economic realities should be overlooked, ignored or diluted, but that they should not ignore other realities which can also contribute to sound policy-making.

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[7] Electricity, for instance, is considerably more expensive when one considers the multiple energy services provided by a coal stove (refer to Chapter Five), not to mention the social and cultural utility it provides.

[8] Thus, for instance, an appropriate strategy may be to introduce a pollution tax on conventional coal simultaneously with the introduction of a competitively-priced low-smoke coal which can substitute for normal coal and still satisfy the social and other needs referred to earlier (see, for example, Van Horen et al 1995).

2.2.2 *The reformist, managerial nature of environmental economics*

A second criticism of conventional prescriptions for dealing with environmental problems which merits consideration, is that they are reformist attempts at achieving `technical fixes'. These prescriptions and approaches are often given the collective label of `environmental management' strategies, and include traditional Cost Benefit Analysis (CBA), environmental impact assessments (EIA), and decision rules such as the `safe minimum standard' tool.[9]

Criticisms of environmental management approaches focus on the question of causation. For instance, Redclift (1988: 638) argued that, in contrast with political-economic approaches which focus on the structural and underlying causes, the `reformist' approach tackles the symptoms rather than the causes of environmental degradation, using an armoury of techniques and management tools, such as EIAs, CBA, pollution taxes and the like.

In some respects, this criticism follows on from the previous one, insofar as many strategies such as pollution taxes and tradable permits have, in the final analysis, relatively modest impacts on the level of environmental degradation. This should be read in the context of the source of this critique, which generally comes more from the ecological end of the spectrum of social and natural scientists than is the case with economists. Inherent in this criticism of environmental economics is often a sense of frustration that it sets out to `solve' the environmental problems, but ends up suggesting only how to achieve the optimal level of pollution in the most efficient manner. Inevitably, at this economic optimum, pollution is non-zero, which is clearly less than satisfactory for ecologically-inclined analysts. Hence their criticism that these technical fixes simply reform the operation of the economic machine, without changing its direction or basic design.

In some respects, the criticism is unfair, since environmental economics usually does not pretend to achieve anything as grand as solving or eliminating a given externality or environmental problem. To this extent, the discipline is therefore being judged against criteria which it has not set for itself.

In other respects, however, it has to be acknowledged that environmental economic analysis is inherently constrained in its possible contribution to the formulation of sound policy. By placing individual utility at the centre of its model, other considerations become lost or considerably simplified. For instance, notwithstanding the considerable attention recently devoted to evaluation of choices regarding tropical forests, it is a gross simplification to reduce to a single (monetary) variable, natural processes such as the complex interplay

[9] This method uses the minimax loss principle from game theory, to ensure that a minimum environmental standard (e.g. genetic pool) is maintained, given high levels of uncertainty and risk at the time that a decision is made (Tisdell 1988, Randall & Farmer 1995).

between organisms in richly-diverse biotic environments evolving over decades and centuries. Given the neo-classical framework into which environmental analysis perforce must fit, this kind of simplification is unavoidable and should therefore not be overlooked.

2.2.3 *Accommodating equity and distributional concerns*

At the heart of this set of criticisms are such questions as `*who* bears the external costs and *who* will enjoy the benefits of abatement measures?'; `are these groups the same?'; `will policies make the distribution of income more equal or less?'

Whilst the question of income distribution has received considerable attention in the environmental economics and welfare economics literature, it remains a difficult area in practical applications of the discipline. There is nothing in environmental economic or cost-benefit analyses which will necessarily lead to a more equal distribution of income. Indeed, a Pareto improvement is one in which, *for a given distribution of income*, someone can be made better off without making anyone else worse off. Further, in terms of the Hicks-Kaldor criterion, it need only be *possible* for compensation to be paid by those left better off to those left worse off (Layard & Glaister 1994: 6). Thus there is nothing in conventional analyses which will of necessity bring about more equitable income distributions, and so equity considerations have to be taken account of (if at all) in one of two explicit ways:

- through lump-sum transfer payments to the intended beneficiaries - generally the theoretically preferred option because it is the most direct means of improving the income distribution; or
- by applying weighting factors to the valuations of different income groups (or other social groups for that matter) in order to account for the varying marginal utilities of income of those groups (Layard & Glaister 1994: 47).

In practice, however, it is acknowledged that lump-sum transfers are not feasible or, at least, are very costly (Layard & Walters 1994: 197), thus some weighting system is required to adjust for the obverse consequences of relying on individual preferences to estimate society's valuation of environmental goods and bads.[10] However, there are often serious data constraints regarding income levels of different social groups, especially in developing countries, with the effect that there are few successful applications of weighting adjustments in practice (Munasinghe 1993a: 26). In any event, the introduction of weights into valuation

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[10] Clearly, the marginal utility of income differs widely between rich and poor persons, hence the need to adjust for this when aggregating individual WTPs (or willingness to accept compensation).

exercises is of necessity a subjective exercise, which leads immediately into the arena of political and social policy.

One of the areas in which equity considerations become most acutely apparent is in the valuation of global damages due to greenhouse gas emissions. In the work of the Intergovernmental Panel on Climate Change (IPCC) Working Group III, for instance, which was tasked with assessing possible economic and social impacts of climate change scenarios, the vexing question of value of lost lives in different countries came to the fore. Using conventional individual preference valuation approaches, the value of a lost life in an industrialised country was held to be in the region of $1.5 million, and with relatively arbitrarily-set values (since there were no valuation studies from developing countries) of $300 000 for middle-income countries and $100 000 for low-income developing countries (after Fankhauser 1992: 15). Whilst these values were not inconsistent with conventional economic theory (subject to the data limitations), they raised serious questions from an equity perspective: to take an extreme example, the costs of climate change abatement measures (such as combating sea level rise) could conceivably be unjustifiably high if they were to prevent 1 000 deaths in low-lying Bangladesh, but could be justified if they would prevent 1 000 deaths in the low-lying Netherlands. In economic terms, this would reflect the higher value `society' would place on the life of a productive Hollander than a subsistence farmer in Bangladesh; clearly, however, the implications of this analysis raise particularly difficult prospects from an ethical and equity perspective. This was reflected in the acrimonious debate which followed the release of the IPCC Working Group III's draft report in 1995.

Likewise, equity concerns underlie much of the debate around joint implementation (JI), a debate which is conducted between those on the one hand, who point to the lower costs of following the route of investing in least-cost GHG abatement options regardless of their location (Markandya 1991: 53), and those on the other hand who are concerned that the interests of poorer developing countries may be compromised by ceding their lowest-cost abatement options to industrialised countries (Agarwal & Narain 1991).

The point therefore is that equity considerations are not necessarily taken care of by conventional analyses, and that explicit attention should be given to these issues. In South Africa, where equity goals enjoy a high political priority, this is especially pertinent.

Finally, it is worth considering why it is that equity issues do not usually feature prominently in environmental economic analyses. An important reason stems from the context in which the discipline was developed intellectually, which was almost exclusively in North America and Europe. Their social contexts differ markedly from those of developing countries: in the latter, markets are often poorly developed, wide disparities in income and wealth exist, individuals frequently have little experience of valuation and bidding processes, public-interest institutions are poorly developed, economic data are weak or non-existent, and the nature of environmental problems is often more diffuse than in industrialised countries (Convery 1995: 4, Redclift & Goodman 1991: 5). As a result, the discipline was first developed in industrialised economies and then followed by attempts to apply the existing framework in the different contexts of developing countries. Had the order been reversed, it is possible that the body of environmental and development economics literature would have looked considerably different today.

As noted in Chapter Two, many environmental economists remain fairly optimistic that the existing framework can be applied without serious difficulties in developing countries (Convery 1995: 4, Pearce & Warford 1993: 108, Munasinghe 1993a: 62). Most are cautious in this optimism, however, recognising that the challenges posed by equity and other goals are significant.

2.2.4 *The appearance of scientific objectivity*

This criticism concerns the impression sometimes left by environmental economic analyses that the results obtained are objective, scientifically sound and defensible, and not open to subjective judgements or political manipulation. In this way, the analysis is elevated to a supposedly higher level, and its prescriptions accepted as the truth. An example of the somewhat naive acceptance of the merits of applying the disciplinary tools - and its inherently ideological nature - is embodied in the Department of Environmental Affairs and Tourism's (DEAT) documents proposing the use of environmental economic analysis. One of their research reports, for instance, begins with a overly-simplistic juxtaposition of `central planning' against `free market' alternatives, as the two possible `solutions to the rationing problem', and espouses `the great merit of the market system' as `consistently provid[ing] what central planning has consistently been unable to provide - an allocation system that works' (DEAT 1993: 1). Taking this as its point of departure, the report proceeds to describe how the use of fiscal incentives in South Africa's environmental management systems can deliver the optimal level of social welfare.

Consumers of the results of these analyses easily lose sight of the assumptions and limitations in the exercise and sometimes place too much weight on the results and too little on the way in which they were derived. Stereotypically, the tendency is to focus narrowly on a single numerical result: `the environmental costs of power station emissions are R*x*, no more and no less'. Clearly, the discipline does not embody this level of scientific precision in its workings.

In some respects this is a criticism less of the intellectual discipline and its fundamental workings, than of some practitioners and advocates of the approach who hold a naive belief in the framework as the provider of all solutions. The risks of doing so are that the limitations inherent in the analytical framework, some of which have been outlined above, can be overlooked, to the detriment of public decision-making.

This point is less likely to be applicable in situations where there is a longer tradition of utilising environmental economic tools, and of exposing those applications to critical and public evaluation. Advocates of the discipline's strengths are also aware of criticisms and go to some lengths to respond to these (for example, Pearce et al 1991: 3-9). In South Africa, by contrast, this is not the case, as evidenced by the DEAT's `Environmental Resource Economics' programme. It is evident from the analysis in this thesis that numerous judgements are required, and whilst these can be made with reference to the underlying theory of neo-classical economics, in many cases the results embody a high range of uncertainty. In this context, an analysis which omits to state its assumptions and to highlight the limitations it encountered, can be faulted for contributing to a false impression of scientific objectivity. This is likely to be particularly important in the coming decade or so in South Africa, as the number of environmental economic analyses proliferate.

3. Directions for energy and environment policy

What does the preceding analysis mean for South Africa's energy and environment policy? In some respects, not much can be said in a definitive way, since the assessment of current external costs is but a first step in a more comprehensive policy-making process. Nonetheless, the exercise has been sufficiently comprehensive to provide clear indications of the directions in which policy-making might proceed.

Firstly, the quantification of external costs in the energy sector means that there is now some basis for making comparisons between the costs and benefits of abatement options. Without an indication of the scale of external costs, it is impossible to assess the likely benefits (avoided costs) of abatement options and, therefore, to say whether the costs of abating those impacts would be justified in economic terms. Thus, for example, it is conceivable that a scenario might have prevailed in which public pressure on Eskom to reduce its pollution impacts, would have induced it to invest in flue gas desulphurisation for its existing power stations, notwithstanding that the costs of doing so are high, and the consequent pollution reductions may be modest. In other words, the analysis in this thesis provides some basis for making more efficient use of scarce environmental resources, without which a sub-optimal allocation could ensue.

Secondly, at the broadest level, the orders of magnitude of the estimated external costs in the energy sector suggest that the greatest direct environmental improvements may be achieved by focusing on the low-income household sector. It is there, rather than in the electricity generation sector, that external costs are most significant. It is sometimes suggested that the national electrification programme will merely replace one set of environmental costs for another, and that this does not lead to any major shift towards sustainable development. However, the analysis in this thesis indicates strongly that this is a positive trade-off to make: in other words, a shift from non-electric sources of energy to electricity will deliver significant external benefits. Whilst this is an important conclusion, it needs to be qualified on two accounts: first, net GHG effects may increase, and second, electricity is not a substitute for certain existing household fuels (for example, coal and especially wood). Nonetheless, on the whole, net external benefits will result from the electrification of low-income households - these effects have not been factored into the financial and economic analyses undertaken to-date.

Significantly, this suggests that there will be convergence between environmental, efficiency and equity goals by focusing on these household-scale problems. This conclusion certainly requires more investigation of the benefits of environmental abatement strategies (they could be extremely expensive) before it can be accepted as conclusive, but the absolute scale of external costs in the household sector suggest that the benefits of intervening here may be significant. To the extent that this is the most cost-effective area in which to achieve environmental improvements, clearly efficiency and environmental goals are served simultaneously. Likewise, given that these external costs are borne primarily by households at the lower end of the income distribution, and since successful interventions would improve their quality of life, equity benefits will result. This is an important conclusion, given the high priority attached by South Africa's democratic government to efficiency and equity goals.

Thirdly, having made the above general observation, it is important to note that the assessment of externalities and abatement options should be carried out for each case individually. As a matter of principle, the appropriate level of analysis is around specific projects, programmes or interventions - the costs and benefits of which can be quantified, political, social and distributive effects considered, and so on. Some indication of the more pertinent policy implications for each Class One externality is provided below.

With respect to the quantified externalities in the electricity sector, the following policy implications can be inferred:

- *Coal mining occupational health effects:* the rates of injury and mortality on South African mines were found to be higher than in the country's competitors, which prompted the Leon Commission to pronounce them `unacceptably' high (refer Chapter Three). From a narrow economic viewpoint, the costs of these health impacts should be compared with the costs of improved health and safety practices on the mines, and the latter would be implemented if they were cost-effective. In addition, it might be suggested by government that greater weighting be given to the health and safety of miners in order to redress their historically poor working conditions, which would shift the balance in the cost-benefit calculation in the direction of increased safety measures. In any event, the Leon Commission made numerous practical recommendations regarding improved health and safety practices and suggested that these could be achieved at relatively low cost.

- *Water pricing in power stations:* the simplest economic response would be to `get the water prices right', and then allow Eskom to decide on its technological and managerial responses. For instance, if it builds another coal power station early in the next decade, it might be prompted by this pricing signal to install dry-cooling systems rather than once-through wet cooling systems. From a practical point of view, water price increases could be negotiated and phased in so as to reduce any price shock effects for electricity consumers.

- *Air pollution from power stations:* the health effects of power station emissions were estimated to result in external costs equivalent to about 0.6 cents per kWh, or 4.6% of current average electricity prices. The costs of retrofitting flue gas scrubbers to all existing power stations, on the other hand, are likely to be in the range of 10% to 49% of current prices (King & Rodseth 1993:20, Petrie et al 1992: 434), thus apparently outweighing the possible benefits of their installation. However, this conclusion should not be drawn so quickly, given that important external effects are not included in this calculation (amongst others, acidification and visibility impacts). What is apparent, is that a methodology exists which can and should be used in the evaluation of technological alternatives, particularly when decisions are made around future supply options - which include the commissioning of another coal power station in South Africa, importing hydro-electricity from Mozambique, Zambia, Zimbabwe and ultimately Zaire, gas from Mozambique or Namibia, or investments in energy efficiency programmes. The appropriate comparison should be between the full social costs of these alternatives.

- *Greenhouse gas emissions:* The international governance of greenhouse gas emissions and mitigation of their effects is a rapidly evolving arena. The current situation is that South Africa, if and when it ratifies the Framework Convention on Climate Change (FCCC), will not face any specific emission reduction targets or obligations. For the present, and at least for the next few years, these commitments are likely to fall only on industrialised countries. Thus there is no immediate necessity for South Africa to introduce GHG mitigation measures at its own expense,[11] nor is there any question, for the present, of introducing a carbon tax or similar externality `adder' on South Africa's carbon dioxide emissions.

This does not mean, however, that the climate change issue can be ignored altogether. Given that the country was the eighteenth largest source of GHG emissions in 1988, and one of the largest on a per capita basis, and since it is not unlikely that middle-income developing countries such as South Africa will face stricter commitments at some point in the not-too-distant future, it is prudent to factor into current and future investment decisions, their climate change implications. This factor could be highly consequential when decisions are taken around the country's next bulk supply option, given that coal is the most carbon-intensive energy source.

Clearly, many abatement options in the electricity sector will have the effect of increasing the price of electricity. Notwithstanding the fact that this runs counter to current price trends and is contrary to Eskom's vision of being the world's lowest-cost producer of electricity (1995a: i), this is precisely what should happen. Electricity consumers should be aware of the resource and environmental implications of their consumption and, moreover, it is correct that the burden of external costs should be shifted from the victims of pollution, accidents and so on, and borne instead by those ultimately responsible for those effects.[12]

Turning to the policy implications in the case of household externalities, several points can be made:

- *Air pollution from coal combustion:* the central estimate of damages due to pollution from household coal combustion was R307 million per annum. Two interventions offer hope of ameliorating these damages: electrification and low-smoke fuels. Although electrification was initially thought to be *the* solution to township air pollution problems (see, for example, Turner & Lynch 1992), the experience over the past decade has

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[11] A range of international funding sources exist to assist developing countries achieve reductions in their GHG emissions, for example, the Global Environment Facility (GEF) administered by the World Bank, United Nations Development Programme and United Nations Environment Programme, and unilateral funding from (inter alia) Germany and the United States. These sources are available, at present, to assist developing countries to fulfil their reporting requirements in terms of the FCCC, as well as to support projects that have benefits for the global environment.

[12] This is not to deny the secondary effects of higher electricity prices throughout the economy.

disproved this view (Eberhard & van Horen 1995: 166). Indeed, electrification is no longer seen as a substitute for coal in most conditions, and so attention has shifted to the second alternative, namely the production of low-smoke fuels (Dickson et al 1995).[13] In the presence of external costs of the order of R307 million, there is considerable scope for government support for low-smoke fuels which are, at present, more expensive than conventional bituminous coals. Assuming that low-smoke fuels would reduce pollution levels to 10% of present levels, then, crudely speaking, up to R270 million could be devoted towards facilitating the substitution of low-smoke fuels for normal coal, with a benefit-cost ratio of greater than one. Given a household coal market of just over 3 million tons per annum, this equates to price support of up to R90 per ton, which would be more than sufficient to shift the price advantage in favour of low-smoke fuels (Van Horen et al 1995). Current policy attention is focusing on a suite of interventions, which might include a mixture of price support, a pollution tax on normal coal, concessionary finance for low-smoke fuel producers, and enforcement of smoke control legislation.

- *Air pollution from wood combustion:* the external costs of pollution emissions were estimated to be highly material in absolute terms, thus making a strong case for effective abatement strategies. This is an area of policy, however, in which improvements are notoriously difficult to realise in practice. Fuel switching to relatively cleaner options such as electricity, gas and paraffin, is not a practical option, since income levels of rural households are generally far too low to allow them to use these commercial fuels instead of wood for energy-intensive services such as cooking and heating. Thus fuel switching will become an option only if income levels in rural areas increase substantially. Other options such as the promotion of efficient wood stoves with chimneys, have had little success to date in South Africa, and are notoriously difficult to gain acceptance amongst rural consumers. Consequently, the abatement of pollution in rural homes is an area urgently requiring further investigation.
- *Paraffin poisoning:* two sets of abatement options exist. Firstly, the production and use of child-resistant paraffin containers (and/or lids for commonly-used containers) is an option which offers considerable potential. A study by the Medical Research Council in the early 1990s suggested that these could be produced an equivalent cost to the direct costs of treating poisoning patients. As noted earlier in this thesis, it is likely that the benefit-cost equation would shift further towards the favourable end of the scale if related opportunity costs (such as travel costs, lost productivity, and so on) were

[13] In 1995, the Department of Mineral and Energy Affairs set the ambitious goal of facilitating a complete substitution of bituminous coals in the household sector by the year 2000.

accounted for. At present, this strategy is being pursued by certain suppliers (oil companies), with little active support from the government.

The second abatement option is electrification, since households generally substitute electricity for paraffin to a relatively large extent. Insofar as electrification reduces the amount of paraffin consumed, and the consequent rate of accidents, it will deliver an important benefit in the form of reduced external costs.

- *Burns and fires:* the main causes of these costs are accidents with candles and paraffin. Since electrification brings about an almost complete substitution of these fuels, these costs can be expected to decline as the electrification programme proceeds. Indeed, the scale of external costs is of a similar order of magnitude to the annual capital investment requirements of the electrification programme (Eberhard & van Horen 1995: 118). To the extent that electrification is proceeding without any external fiscal subsidy at present (although there is a significant cross-subsidy from other electricity consumers to newly-electrified low-income consumers), it does not appear that specific strategies are required to mitigate this external effect. The benefits of electricity, in its cleanliness and lower risk of fires, are probably already embodied, to some degree, in the price which consumers are willing to pay for the product. Nonetheless, it is important that these external costs not be overlooked altogether, since there may be a point at which some kind of transfer payment is justified to support continued electrification of households, assuming that their willingness to pay is constrained by low income, in order that these external benefits can be captured.
- *Fuelwood scarcity:* a range of policies are under consideration by various government departments at present.[14] These include better management of natural woodlands which supply the bulk of wood resources to households, supply increases through woodlots, social forestry and small-grower schemes, and fuelwood conservation (Eberhard & van Horen 1995: 155-160). Whilst such programmes generally have high capital and operating costs, and struggle to compete with other sectors for fiscal resources, it is important that the opportunity costs of fuelwood scarcity are factored into assessments of the costs and benefits of intervention options.
- *Greenhouse gas emissions:* in the same way as there is no question of a carbon tax being imposed on the combustion of fossil fuels in the electricity sector, this is also not relevant to the household sector. Of more importance, perhaps, are possibilities of securing

[14] Fuelwood supply and management falls under the responsibility of several government departments at all tiers of government, as well as under traditional leaders in rural areas. This institutional fragmentation makes effective management especially difficult.

concessionary international finance for strategies which deliver global benefits. An example of this is a proposal which was developed for support from the Global Environment Facility (GEF) to finance the incremental costs of improved thermal performance in low-cost housing (described in Van Horen & Simmonds 1995). Projects such as these which would abate or avoid emissions of GHGs, are eligible for concessionary GEF financing and would simultaneously deliver local benefits.

Clearly, each of these areas merits detailed consideration, both with respect to their economic costs and benefits, and with reference to the political and distributional imperatives impacting upon each. Although the preceding discussion has provided an indication of policy directions which might be pursued, a full analysis of each area is beyond the scope of this thesis.

4. Conclusion

In the first two years of democratic government in South Africa, there has been much activity in the policy arena. In several sectors, however, policy formulation is still in its infancy - one such area is at the interface of energy and environment policy. It was in this context that this thesis posed its first question: namely, which are the energy-environment problems with the greatest costs to society, and which therefore warrant attention for policy-making purposes? Since there are numerous points of contact at this interface, the thesis focused on two: the effects of electricity generation, and the effects of household energy consumption.

Using an environmental economic framework of analysis, and specifically the theory around externalities and their valuation, an empirical study of external costs in these two sectors was undertaken. Total external costs, average costs and marginal costs were estimated for a range of externalities in each sector, using a classification system developed for this study. The results of the valuation exercises in the two sectors were compared, from which it was apparent that external costs, measured on any of these bases, were significant in both sectors when damages attributed to greenhouse gas emissions were included. When the latter were excluded, the external costs in the household sector were substantially higher. Thus the first hypothesis presented at the beginning of the thesis was confirmed. In undertaking this analysis, this study is unusual amongst developing countries where there has been relatively little consideration of these kinds of externalities.

The second question addressed in this thesis concerned whether the environmental economics discipline could be embraced for purpose of such evaluations in a developing country context. Having utilised the framework in this study to yield plausible quantitative results, it is clear that the discipline has much to offer policy-making in contexts such as that of South Africa. At the same time, though, a number of limitations of conventional analyses such as this were highlighted, ranging from technical constraints of poor input data, to concerns over the political context in which policy is formulated and over the distributive effects of externalities and abatement options. By giving consideration to such factors, the overall analysis is likely to be enhanced and, it is anticipated, better policy-making should result.

Taken as a whole, the analysis in this thesis has yielded an important conclusion: namely that there are potentially significant gains to be made, from both an efficiency and an equity perspective, if policy-making targets the environmental problems in the low-income household sector. This convergence between efficiency and equity goals is especially important in the context of South Africa's infant democracy, in which these goals are frequently in conflict with each other, and in which difficult trade-offs have to be made between competing demands on limited national resources.

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Appendix 1

Technical and geographic data on power stations

Table A1 Eskom's coal power stations: location, output and capacity, 1994

	Province	Latitude (degrees S)	Longitude (degrees E)	Altitude	Units produced (GW)	Net capacity (MW)
Arnot[1]	Mpumalanga	25.9500	29.8000	1 700 m	4 557.5	1 955
Duvha	Mpumalanga	25.9667	29.333	1 596 m	21 970.0	3 450
Hendrina	Mpumalanga	26.0333	29.6000	1 000 m	11 871.2	1 900
Kendal	Mpumalanga	26.1000	28.9667	1 610 m	19 396.8	3 840
Kriel	Mpumalanga	26.2500	29.1833	1 619 m	13 394.5	2 850
Lethabo	Free State	26.7333	27.9667	1 460 m	17 863.1	3 558
Matimba	N. Prov.	23.6667	27.6167	880 m	22 685.2	3 690
Matla	Mpumalanga	26.2833	29.1333	1 620 m	18 737.2	3 450
Tutuka	Mpumalanga	26.7667	29.3500	1 625 m	17 522.9	3 510

[1] Only two of Arnot's six sets were in operation in 1994.

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Table A2 Eskom's coal power stations: stack dimensions and flue gas data

	Stack height (m)	Internal diameter (m)	Aggregated diameter (m)[2]	Ave flue gas velocities (m/s)[3]	Flue gas temp (degrees C)
Arnot	193	8.00	11.31	4.0	138
Duvha	300	7.13	10.08	21.0	141
Hendrina	110	13.00	18.38	8.5	128
Kendal	275	7.46	10.55	21.0	127
Kriel	213	13.82	19.55	11.2	110
Lethabo	275	7.46	10.55	23.5	135
Matimba	250	7.52	10.63	19.5	130
Matla	278 & 213	6.32	8.94	9.3	130
Tutuka	275	6.45	9.12	31.0	135

[2] Aggregated diameter for two chimneys calculated by multiplying single diameter by square root of 2.

[3] Averaged, where necessary.

Appendix 2

Cost of illness data

Table B1 Cost of illness data for pneumonia, asthma and chronic obstructive pulmonary disease

	Pneumonia			Asthma			COPD		
	mild	mod	severe	mild	mod	severe	mild	mod	severe
% of cases[1]	70%	25%	5%	70%	25%	5%	70%	25%	5%
Outpatient visits per case per year	0.5	0.5	0.5	2	6	12	2	6	12
GP visits per case per year	0.5	0.5	0.5	2	6	12	2	6	12
Days in general ward per case	0	12	8	0	0	5	0	0	5
Days in ICU per case	0	0	7	0	0	1	0	0	1
Drug costs p.a.[2]	incl.	incl.	incl.	R180	R720	R1440	R180	R720	R1440
Cost per outpatient	R30	-	-	R30	R60	R90	R30	R60	R90
Cost per GP visit	R100	R100	R100	R100	R100	R100	R100	R100	R100
Cost per general ward day[3]	-	R200	R200	-	-	R200	-	-	R200
Cost per ICU day[4]	-	-	R1616	-	-	R1616	-	-	R1616
Transport costs per case[5]	R10	R130	R320	R40	R120	R360	R40	R120	R360
Persondays work missed per case[6]	1	13	16	4	12	30	4	12	30
Cost per workday missed[7]	R126	R126	R126	R126	R126	R126	R126	R126	R126
Total	R201	R4233	R15313	R1104	R3792	R15228	R1104	R3792	R15228

[1] Most of these data are based on discussions with Drs George Swingler and Anne Robertson, Red Cross Hospital, Cape Town, and Dr Rodney Ehrlich, Department of Community Health, UCT.

[2] Based on estimated cost per inhaler of R100 for private care and R50 for public care, and a split of 20:80 for private:public care, giving a weighted average of R60 per inhaler. Mild cases will use 4 per annum, moderate will use 12 per annum and severe cases 24 per annum.

[3] Based on analysis of hospital costs in former Cape Province, supplied by Dale McMurchy, Health Economics Unit, UCT.

[4] Based on standard rates for general ICU according to Cape Medical Plan. Specialised ICU rates are R3034 per day, but these have not been used in this study.

[5] Transport costs assumed to be R10 per person per return visit to the hospital, clinic or GP. For patients admitted to hospital, it is assumed there will be one visitor per admission day.

[6] In the case of ICU admissions, it is assumed that one care-giver will miss work for each day the patient is in hospital.

[7] Based on 1993 average wage in non-agricultural sector of R2 527 per month (South African Statistics 1994: 4.14) and a 22 day working month; inflated by 10% to give 1994 daily wage of R126.

Summary of cost estimates

1. Respiratory hospital admissions (RHA, for pneumonia, asthma, COPD): weighted average cost per admission = (25/30 * R4233) + (5/30 * R15313) = R6080 per admission for pneumonia, and for asthma and COPD = (25/30 * R3792) + (5/30 * R15228) = R5698 per admission. Assume each accounts for a third of RHA, then weighted average cost of RHA is R5825 per admission.
2. Emergency room visits (outpatients) for pneumonia, acute bronchitis: average cost per visit = R201.
3. Asthma attacks (mild): average cost per occurrence-day = R1104/4 = R276.
4. Persons with mild COPD (e.g. chronic bronchitis): estimated cost per year = R1104.
5. Lifetime cost of mild COPD = R1104 pa at a real discount rate of 2% over 40 years = R30200, rounded down to R30000.

Table B2 Cost of illness data for paraffin poisoning cases

Cases admitted overnight	
X-rays, medications and other direct costs[8]	R10
Hospital admission cost per day[9]	R200
Days in hospital	2.4
Transport costs per case[10]	R34
Person days work lost per case	1
Cost per workday missed[11]	R126
Total cost per case admitted	R650
Cases treated as outpatients	
X-rays, medications and other direct costs	R10
Transport costs per case	R20
Person days work lost per case	1
Cost per workday missed[12]	R126
Total cost per outpatient case	R146

[8] These costs may include, in addition to X-rays and antibiotics, oxygen assistance and bronchodilators (cost estimates from Dr. George Swingler, Red Cross Children's Hospital).

[9] As per Table B1.

[10] As in Table B1, it is assumed that the cost of one return trip to the hospital is R10; in addition to the patient's travel, it is assumed that each child will have one visitor per day in the hospital.

[11] As per Table B1.

[12] As per Table B1.

Table B3 Cost of illness data for burns injuries

Hospital admission cost per day[13]	R666
Days in hospital	11.6
Transport costs per case[14]	R126
Person days work lost per case[15]	2.9
Cost per workday missed[16]	R126
Rehabilitation costs[17]	R200
Total cost per case	R8 417

[13] As per Hudson et al (1994: 252), escalated to 1995 Rands; includes direct costs of medication, blood transfusion and other treatment.

[14] As before, it is assumed that the cost of one return trip to the hospital is R10; in addition to the patient's travel, it is assumed that each patient will have one visitor per day in the hospital.

[15] It is assumed that one work-day will be lost for every four patient-days in hospital.

[16] As per Table B1.

[17] These include time spent in convalescence homes, as well as follow-up plastic surgery and rehabilitation treatments.

Appendix 3

The EXMOD externalities model

1. Background[1]

In response to an order from the Public Service Commission in New York state, United States, a consortium of agencies commissioned a major study to assist in quantifying the environmental externalities in the power sector.[2] The aim of the project was to produce a methodology and user-friendly computer model which could be applied by regulatory authorities and utilities to estimate new and relicensed electric resource options in New York state. The results of the project, which was completed in early 1995, were summarised in four comprehensive volumes (Rowe et al 1994a, 1994b, Bernow et al 1995a, 1995b).

The project had several distinct components: firstly, the development of a methodology which could be applied to externality evaluation, secondly, the development of a computer modelling tool, called EXMOD, which performed the calculations required by the methodology and which reported their results, and thirdly, the testing of the model through various case studies in New York state.

2. Overview of the model

The study used the damage function or impact pathway approach, as described in Chapter Three of this thesis, and displayed in Figure 3.1. For purpose of quantifying externalities in terms of this approach, data were required for each of the following main steps in the impact pathway:

- facility and site characteristics (such as geographic co-ordinates, elevation, generation capacity, capacity utilisation, fuel characteristics, chimney height and diameter, and exhaust gas temperature and velocity);
- pollution emissions from facilities (for pollutants such as sulphur dioxide, nitrogen oxides and particulates);

[1] The description in this Appendix draws from Bernow et al (1995a) and Rowe et al (1995).

[2] The funding agencies were the Empire State Electric Energy Research Corporation, the New York State Energy Research and Development Authority, the New York Department of Public Service, and the Electric Power Research Institute, and the two organisations undertaking the work were the Tellus Institute, Boston, and RCG/Hagler Bailly, Boulder.

- dispersion and deposition of pollution after its emission (this was modelled by air dispersion models which are described below);
- receptors on the ground which may suffer damage from pollution (such as human populations, crops and buildings);
- dose-response functions (that is, the response of the receptor to an given exposure of a pollutant);
- physical impacts (such as physical number of health outcomes);
- valuation in economic terms to yield estimated damage costs.

For each of these categories, the EXMOD model contained default assumptions which reflected the results of literature reviews undertaken as part of the study, as well as demographic and other conditions in New York state where the project was undertaken.

The air quality modelling component of EXMOD comprised three air quality models (called ISC2, SLIM3 and SCREEN2) which were selected from amongst existing models in North America. They were integrated into EXMOD so that the dispersal of emissions from specified facilities were modelled, using long-term wind data, yielding estimates of changed ambient air concentrations. The three models covered various ranges: < 50 km, > 50 km, and 80 km to 500 km respectively, and were based on simple Gaussian plume dispersion models.[3]

EXMOD contained a number of so-called `end-point' data sets: in other words, the model could calculate external costs for many effects over-and-above air pollution impacts: for example, water quality damages (due to changed fishing productivity), visibility impacts, valuation of nuclear radiation, and so on.

3. Application of EXMOD in South Africa

The application of EXMOD in South Africa for purpose of this thesis was the first time the methodology had been applied outside of New York state. The model was used exclusively to calculate health impacts of air pollution emissions, and only for three pollutants for which data were available in this study: SO_2, NO_x and particulates. In other words, other end-point sets were not used in this study since insufficient local data exist to substitute for default (American) data.

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[3] As noted in Chapter Seven, these models do not necessarily best represent actual dispersal conditions on the Mpumalanga Highveld, although no better models are yet available.

All relevant default data were replaced with South African data, with the exception of dose-response functions (as discussed in Chapters Four and Seven). Specifically, the following data were collected and entered to run the model for local conditions:

- facility and site characteristics for each of Eskom's nine coal power stations (these data are described in Appendix 1);
- pollution emission factors for each power station (this is described in Chapter Three);
- long-term average wind data from 15 monitoring stations in South Africa (and 2 in Namibia) were entered for purpose of the air quality models;
- demographic data were entered, in the form of population numbers in each magisterial district in South Africa, using census sources; because they were not Class One externalities, no data were entered in respect of crops and buildings;
- valuation estimates for the main health outcomes predicted by EXMOD were derived and entered (these are described in Appendix 2).

Since EXMOD is specific to each site and facility, the model was run nine times, once for each of the coal power stations. The results of the model, in terms both of physical health outcomes and of economic damages, were then aggregated in a separate spreadsheet (these are reported in Chapter Four). The resulting values were summarised in total Rand values and in mills per kilowatt-hour.

The critical component of EXMOD therefore, which necessitated its use in this thesis, was the modelling of the air pollutants' transport after their emission from power stations, until they were deposited or distributed to human populations. Without this, it would not have been possible to use the damage function approach since this essential link in the impact pathway would have been missing, or would have necessitated several crude assumptions. Problems with the application of EXMOD's air quality models to South African conditions remain, and are discussed in Chapter Seven.

Appendix 4

Calculation of external costs of household coal and wood consumption

This appendix summarises the calculations of external costs in the household coal and wood sectors, which were made using a simple spreadsheet model. Table D1 contains input data for each step in the impact pathway, as well as results for the coal externality, while Table D2 contains the same for the wood externality.

Table D1 Calculation of health costs from coal use (dose response functions, exposed population, population with asthma, pollution exposures, physical health outcomes, unit costs and total costs)

Dose-response functions		Low	Central	High
Mortality >65	>65	0.000000101	0.000000169	0.000000254
Mortality <65	<65	0.0000000014	0.0000000023	0.0000000035
Chronic bronchitis	>25	0.00003	0.000061	0.000093
Resp. hosp. adm.	all	0.000000018	0.000000033	0.000000048
Outpatients visits	all	0.00000032	0.00000065	0.00000097
Asthma attacks	asthma sufferers	0.00009	0.00016	0.00054
Restricted activity day	>18	0.00008	0.00016	0.00025
Resp symptom day	all	0.0002	0.00046	0.0007
Children bronchitis	<18	0.0008	0.0016	0.0024

Exposed population			6,000,000
Age group	0-9	17%	1,020,695
	10-19	16%	945,985
	20-29	20%	1,228,821
	30-39	18%	1,097,148
	40-49	13%	781,358
	50-59	8%	504,194
	60-64	2%	148,589
	65+	5%	273,210

Population with asthma 15%

PM10 exposures (µg/m3) 272

Physical health outcomes		Low	Central	High
Asthma attack	Occurrence-day	22,032	39,168	132,192
Acute bronchitis	Person	427,950	855,899	1,283,849
Chronic bronchitis	Person	32,912	66,921	102,027
Outpatient/GP visit	Visit	522	1,061	1,583
Mortality >65	Death	8	13	19
Mortality <65	Death	2	4	5
Resp. Symptom day	Occurrence-day	326,400	750,720	1,142,400
Resp. Hospital adm.	Admission	29	54	78
Restricted activity	Occurrence-day	87,765	175,530	274,266

Rand values per outcome	Low	Central	High
Asthma attack	207	276	345
Acute bronchitis	201	201	201
Chronic bronchitis	828	1,104	1,380
Outpatient/GP visit	151	201	251
Mortality >65	800,000	1,200,000	1,900,000
Mortality <65	800,000	1,200,000	1,900,000
Resp. Symptom day	8	10	13
Resp. Hospital adm.	4,369	5,825	7,281
Restricted activity	95	126	158

Economic value of health outcomes	Low	Central	High
Asthma attack	4,560,624	10,810,368	45,606,240
Acute bronchitis	86,017,950	172,035,699	258,053,649
Chronic bronchitis	27,251,136	73,880,784	140,797,260
Outpatient/GP visit	78,822	213,261	397,333
Mortality >65	6,400,000	15,600,000	36,100,000
Mortality <65	1,600,000	4,800,000	9,500,000
Resp. Symptom day	2,448,000	7,507,200	14,280,000
Resp. Hospital adm.	126,701	314,550	567,918
Restricted activity	8,293,793	22,116,780	43,196,895
	136,777,026	307,278,642	548,499,295

Table D2 Calculation of health costs from wood use (dose response functions, exposed population, population with asthma, pollution exposures, physical health outcomes, unit costs and total costs)

Dose-response functions		Low	Central	High
Mortality >65	>65	0.000000101	0.000000169	0.000000254
Mortality <65	<65	0.0000000014	0.0000000023	0.0000000035
Chronic bronchitis	>25	0.00003	0.000061	0.000093
Resp. hosp. adm.	all	0.000000018	0.000000033	0.000000048
Outpatients visits	all	0.00000032	0.00000065	0.00000097
Asthma attacks	asthma sufferers	0.00009	0.00016	0.00054
Restricted activity day	>18	0.00008	0.00016	0.00025
Resp symptom day	all	0.0002	0.00046	0.0007
Children bronchitis	<18	0.0008	0.0016	0.0024

Exposed population			17,000,000
Age group	0-9	17%	2,891,970
	10-19	16%	2,680,289
	20-29	20%	3,481,658
	30-39	18%	3,108,586
	40-49	13%	2,213,848
	50-59	8%	1,428,550
	60-64	2%	421,003
	65+	5%	774,095

Population with asthma		15%	
PM10 exposures (µg /m3)	474	947	1421

Physical health outcomes		Low	Central	High
Asthma attack	Occurrence-day	108,783	386,376	1,956,717
Acute bronchitis	Person	2,113,001	8,443,088	19,003,635
Chronic bronchitis	Person	162,502	660,146	1,510,210
Outpatient/GP visit	Visit	2,579	10,464	23,432
Mortality >65	Death	37	124	279
Mortality <65	Death	11	35	81
Resp. Symptom day	Occurrence-day	1,611,600	7,405,540	16,909,900
Resp. Hospital adm.	Admission	145	531	1,160
Restricted activity	Occurrence-day	433,340	1,731,531	4,059,705

Rand values per outcome	Low	Central	High
Asthma attack	207	276	345
Acute bronchitis	201	201	201
Chronic bronchitis	828	1,104	1,380
Outpatient/GP visit	151	201	251
Mortality >65	800,000	1,200,000	1,900,000
Mortality <65	800,000	1,200,000	1,900,000
Resp. Symptom day	8	10	13
Resp. Hospital adm.	4,369	5,825	7,281
Restricted activity	95	126	158

Economic value of health outcomes	Low	Central	High
Asthma attack	22,518,081	106,639,776	675,067,365
Acute bronchitis	424,713,201	1,697,060,688	3,819,730,635
Chronic bronchitis	134,551,656	728,801,184	2,084,089,800
Outpatient/GP visit	389,429	2,103,264	5,881,432
Mortality >65	29,600,000	148,800,000	530,100,000
Mortality <65	8,800,000	42,000,000	153,900,000
Resp. Symptom day	12,087,000	74,055,400	211,373,750
Resp. Hospital adm.	633,505	3,093,075	8,445,960
Restricted activity	40,950,630	218,172,906	639,403,538
	674,243,502	3,020,726,293	8,127,992,480