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EVALUATING THE COSTS AND BENEFITS ASSOCIATED WITH THE REDUCTION IN SO_2 EMISSIONS FROM INDUSTRIAL ACTIVITIES ON THE HIGHVELD OF SOUTH AFRICA

by

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SUMMARY

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Air quality management in South Africa has undergone significant legislative reform in recent years, aimed at improving air quality within South Africa. The National Environmental Management: Air Quality Act 39 was promulgated in 2005 and makes provision for various instruments to improve ambient air quality. One such instrument is minimum emission standards, limiting the emissions from industrial point sources. The standards were first set in 2010, but were subsequently amended in 2013. These standards have been set for various industry sectors and require facilities to comply with one set of emission standards by 2015 and a stricter set of standards by 2020, dependent on the commissioning date of the facility.

In setting the standards, the costs of compliance and expected ambient benefit of compliance were only indirectly assessed. Subsequent to the promulgation of the standards, various industries have indicated that not all of the standards were feasible through various applications for the postponement of these standards. The effect of such a

postponement will be to delay compliance with the emission standards, resulting in the intended ambient air quality benefits not being realised.

The purpose of the investigation was to assess the desirability of implementation of the standards from an environmental as well as economic point of view. Due to the scope of such a study and the variety of industries involved, the investigation was limited to a single set of standards. The research assessed compliance with the Category 1.1 standards (solid fuel combustion installations used primarily for steam raising or electricity generation with a design capacity greater than 50MW) for sulphur dioxide (SO_2) emissions as the majority of installations falling into this category have indicated that compliance with the SO_2 emission standard will not be achieved within the required timeframe.

A review of environmental evaluation techniques was done in order to determine the most appropriate method to assess the economic desirability of the legislation, taking into consideration the expected benefit of implementation as well as the costs and impacts of implementing the required abatement technology to reach the standards.

The study used a bottoms-up or impact pathway approach to analyse the impact of emission reduction. The costs and benefits associated with the implementation of an SO_2 point source standard of 500 mg/Nm^3 for solid fuel combustion installations (Category 1.1 sources) was evaluated to determine the net present value of SO_2 regulation on the Mpumalanga Highveld of South Africa. All category 1.1 sources within the study area expected to have a significant impact on ambient SO_2 concentrations were included in the study. An evaluation of the likely technology to be implemented to reach the new plant (2020) SO_2 emission standard of 500 mg/Nm^3 was conducted and the installation of wet flue gas desulfurisation (FGD) was determined to be the technology of choice. In order to conduct the economic valuation, the costs and benefits associated with the installation of FGD was identified and ranked into four categories, based on the expected impact and the availability of information. All costs and benefits that could be quantified and monetized (Category 1 impacts) were included in the evaluation.

While the methodology followed is widely used and well documented, uncertainty is introduced into the evaluation at each step of the analysis, requiring assumptions to be made which could significantly impact the result of the economic analysis. To a certain extent, uncertainties can be lessened by well-considered and valid assumptions and further reduced by sensitivity analysis, yielding a range of potential values that could realise for sensitive inputs. A sensitivity analysis was conducted on the costs and benefits with the largest impact on net present value (NPV) or the largest uncertainty associated with the calculation to determine a range of feasible values. Site specific information was used where available, supplemented by benefit transfer where local data was not available.

The impact of reduced ambient pollutant concentrations on human health was identified as the only quantifiable benefit. To evaluate the expected health benefit, the change in ambient pollutant concentrations was modelled using dispersion modelling. The health impacts associated with reduced SO_2 and secondary sulfate concentrations was calculated using concentration response functions and population data obtained from the South African 2010 census. The impact on premature adult mortality was found to be the most significant benefit and dependent on the concentration response function selected and sensitive to the value of a statistical life (VSL) estimate used (high R115 billion; low R36 billion). The choice of appropriate dose-response functions and the applicability thereof in the South African context are important considerations, likely requiring further study. Potentially significant impacts such as the ecological impact of acid deposition were identified by the study, but were not included in the NPV calculation as they could not be adequately quantified and monetised. Due to the long timeframes associated with the recovery of ecosystems and international experience with improved ecosystem functioning resulting from pollution reduction initiatives, further study is recommended prior to the inclusion of the benefit in this study.

The most significant costs associated with the implementation of FGD were the capital cost of installation, and operating costs comprising of water and lime costs. The capital cost of FGD installation was found to be the most significant cost and was sensitive to the evaluation method (central R187 billion; high R306 billion; low R80 billion). In this case study, the operating costs of the abatement equipment was of the same order of magnitude as the health benefits. If only the capital cost of a regulation is included in an analysis, the cost implications could be significantly underestimated. Although relatively better defined than the evaluation of benefits, assumptions can also influence the calculation when evaluating the cost side of the NPV equation and availability of these resources and logistics associated with the provision of these resources has to be considered.

A comparative analysis was conducted on the station with the highest health impact valuation to determine whether partial implementation of FGD could be considered viable. The results indicated that, due to the high capital cost of FGD installation, the NPV for the particular installation was negative under all scenarios.

A project such as a regulatory intervention is generally considered economically viable if the benefits expected exceed the cost of regulation. The results indicate that, given the information currently available, it is unlikely that the benefit of reducing SO_2 emissions to the required standard outweighs the cost of implementation.

LIST OF ABBREVIATIONS

ACS	American Cancer Society
AQA	Air Quality Act
CBA	Cost-benefit analysis
CFB	Circulating fluidised bed
CI	Confidence interval
COI	Cost of illness
CO_2	Carbon dioxide
DEA	Department of Environmental Affairs
DEAT	Department of Environmental Affairs and Tourism
DME	Department of Minerals and Energy
DMR	Department of Mineral Resources
DWA	Department of Water Affairs
DWAF	Department of Water Affairs and Forestry
ERF	Exposure response function
FGD	Flue gas desulfurisation
GDP	Gross domestic product
HEI	Health Effects Institute
HPA	Highveld Priority Area
LHWP	Lesotho Highlands Water Project
MES	Minimum emission standards
NPV	Net present value
NAAQS	National ambient air quality standards
PM	Particulate matter
PPP	Purchase power parity

RIA	Regulatory impact assessment
RIS	Regulatory impact study
RR	Relative risk
SO_2	Sulfur dioxide
US-EPA	United States – Environmental Protection Agency
VSL	Value of a statistical life
VOLY	Value of a life year
WHO	World Health Organisation
WTA	Willingness to accept
WTP	Willingness to pay
YLL	Years of life lost

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1 INTRODUCTION

1.1 PROBLEM STATEMENT

1.1.1 Context of the problem

Air quality management in South Africa has undergone significant legislative reform in recent years, aimed at improving air quality within South Africa. The National Environmental Management: Air Quality Act 39 (AQA) was promulgated in 2005 and makes provision for various instruments to improve ambient air quality. One such instrument is minimum emission standards, limiting the emissions from industrial point sources. The standards were first set in 2010, but were subsequently amended in 2013. These standards have been set for various industry sectors and require facilities to comply with one set of standards by 2015 and a stricter set of standards by 2020, dependent on the commissioning date of the facility.

In setting the standards, the costs of compliance and expected ambient benefit of compliance were only indirectly assessed. Subsequent to the promulgation of the standards, various industries have indicated that not all of the standards were feasible through various applications for the postponement of these standards. The effect of such a postponement will be to delay compliance with the emission standards, resulting in the intended ambient air quality benefits not being realised.

1.1.1 Research gap

The purpose of air quality legislation is to improve or prevent deterioration of ambient air quality. In order to assess whether a legislative requirement is sustainable, an economic analysis can be performed to evaluate whether the intended benefit justifies the implementation cost of the regulation. While previous studies have been conducted assessing the costs and benefits of pollution reduction, the intent of this study is to extend the analysis to include all quantifiable externalities to determine the true cost and benefits associated with the legislative requirement.

1.2 OVERVIEW OF STUDY

The purpose of the investigation is to assess the desirability of implementation of the standards from an environmental as well as economic point of view. Due to the scope of such a study and the variety of industries involved, the investigation will be limited to a single set of standards. The research will assess compliance with the Category 1.1 (solid fuel combustion installations used primarily for steam raising or electricity generation with a design capacity greater than 50MW) standards for SO_2 emissions. The Category 1.1 standard includes limits for SO_2, NOx and PM. The study will focus on the standard pertaining to SO2 emissions, as the majority of installations falling into this category have indicated that compliance will not be achieved within the required timeframe. The cost and externalities associated with the reduction of PM and NOx to the required standards are less significant than for SO_2 (Sasol 2014; Eskom 2014) and various installations have indicated the intention to comply with these requirements. PM and NOx compliance was therefore not included in the study.

A review of environmental evaluation techniques will be done to determine the most appropriate method to assess the desirability of the legislation, taking into consideration the expected benefit of implementation as well as the costs and impacts of implementing the required abatement technology to reach the standards.

The study is intended as a practical implementation of economic analysis of a specific regulation, taking South African-specific information into consideration.

This report is structured in three parts:

- Part A provides the background to environmental evaluation economics and evaluation techniques and the practical implementation thereof in the international as well as South African context.
- Part B details the practical application of the theory discussed in Part A as applicable to the case study of SO_2 reduction on the Highveld of South Africa.
- Part C discusses the conclusions of the case study as described in Part B. The study limitations and potential refinements for future studies are discussed.

2 PART A

2.1 CHAPTER OBJECTIVES

The purpose of Part A is to provide the background to environmental evaluation economics and evaluation techniques and the practical implementation thereof in the international as well as South African context.

2.2 ECONOMICS

The purpose of this section is to provide a high-level overview of the basic economic principles underpinning the theory behind environmental economics. The application of the economic theory, valuation techniques that can be used and important considerations related to economic evaluation are discussed.

2.2.1 Markets and market failures

In an ideal economy, resources are efficiently allocated to maximize the overall welfare of all participants in the economy. A perfectly efficient market exists when the supply of a good equals the demand of that good (Viljoen 1998). This can be illustrated in Figure 1 below:

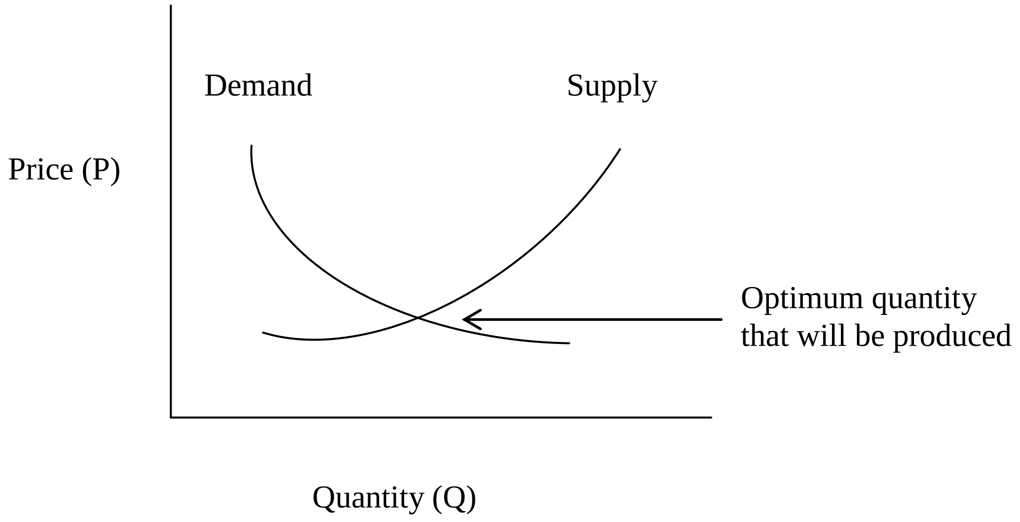

Figure 1: Efficient market conditions exist when supply equals demand.

When a market does not allocate resources efficiently, a market failure occurs. A market failure occurs where, for any reason, the quantity supplied by the market is not equal to the demand. This implies an economic inefficiency whereby resources can be better allocated to increase overall welfare. One reason for a market failure is the presence of externalities. An externality occurs when the costs of impacts, either positive or negative, are not reflected in the market relationship between a producer and its consumers (Viljoen 1998; Van Horen 1996).

In the presence of externalities, the supply curve is an inaccurate reflection of market equilibrium, as the cost of production is artificially low, due to the non-inclusion of externality costs that are not borne by the producer, but rather by society. Should these costs be included in the costs of production, the supply curve would move to the left from S1 to S2, as shown in Figure 2 below:

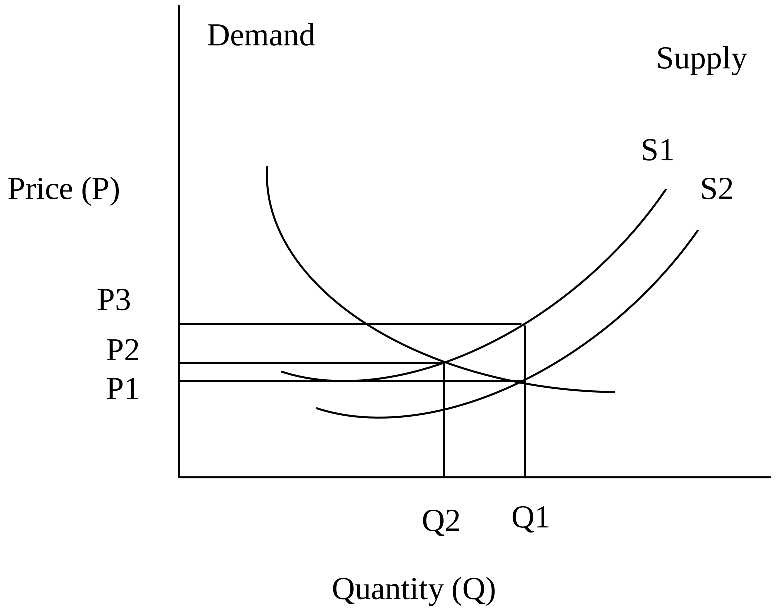

Figure 2: Impacts of externalities

When externalities are accounted for, the price moves from P1 to P2, with quantities demanded moving from Q1 to Q2. At quantity Q1, the actual price of the commodity, taking externalities into account, should be P3 in an efficient market. The welfare loss of the externality is therefore reflected by the differences in these prices.

Where externalities exist, the market will not produce an efficient result and intervention is required to correct the market to more efficient and sustainable levels (Van Horen 1996). Various types of interventions can be employed to correct the market failure, such as:

- free market bargaining options;
- environmental standards and regulations;
- pollution taxes and subsidies; and
- marketable emission permits.

Free market bargaining suggests that an efficient market can be obtained by bargaining between polluter and society, without any direct government involvement (Van Horen 1996). In practice, however, this does not occur and some form of intervention is required.

Environmental standards and regulations can take various forms, such as placing a limit of amount of pollution emitted, specifying control technologies, ambient concentrations limits and so forth. These interventions are relatively simple and are often the first tools to be utilized by governments (Van Horen 1996). Depending on the manner in which the standards are set, they may not achieve an economically efficient outcome. Pollution taxes and subsidies underlie the polluter pays principle (Van Horen 1996). The benefit of such taxes is that it encourages the reduction of pollution and can have social benefits if taxes are not seen as a source of income, but rather utilized to enhance the system of environmental regulation and enforcement (Van Horen 1996).

Emissions trading involve the issuing of permits for a maximum amount of pollutant that can be emitted, which is traded in the market (Van Horen 1996). In theory, this should result in the most cost-efficient market. Furthermore, the regulator can buy up permits to further reduce pollution levels. In a monopolistic industry structure, the use of permits may not yield efficient results, as hoarding of permits could occur (Van Horen 1996), which would be the case if few emitters emit the largest tonnages of pollutants.

In the context of this study, the intervention employed by the Regulator was the setting of emission limits as described in Section 3.2.1.

2.2.2 Practical implementation of economic principles

Vilfredo Pareto described in 1909 whether or not a change improves overall welfare. The Pareto principle states that a change is Pareto efficient if at least one person is better off and no person is worse off (Stavins 2007; Viljoen 1998). Once a position has been reached whereby one person's position can no longer be improved without worsening the position of somebody else, a Pareto-optimal allocation of resources has been achieved (Viljoen 1998). Very few public policies, however, meet this criterion.

In order to formulate a more pragmatic approach, the Kaldor-Hicks criterion was postulated in 1938 and states that a change is welfare-improving if those who gain from the change could, in principle, compensate the losers with at least one gainer still being better off (Stavins 2007). This test forms the basis for modern cost-benefit analysis, whereby total benefits should exceed total costs, with the most efficient allocation of resources occurring where the difference between the costs and benefits is maximised.

In theory, a government possesses many instruments with which it seeks to maximize overall welfare. In practice and even in theory, this would be too complex to undertake and legislative impacts are assessed separately and not in its entirety. The purpose of a cost-benefit analysis can serve as a tool to partially solve the maximization problem by determining whether a project or intervention has the potential to positively contribute to social welfare (Staehr 2004).

The purpose of a cost-benefit analysis therefore is to ensure a more efficient allocation of resources, as shown in Figure 3 below.

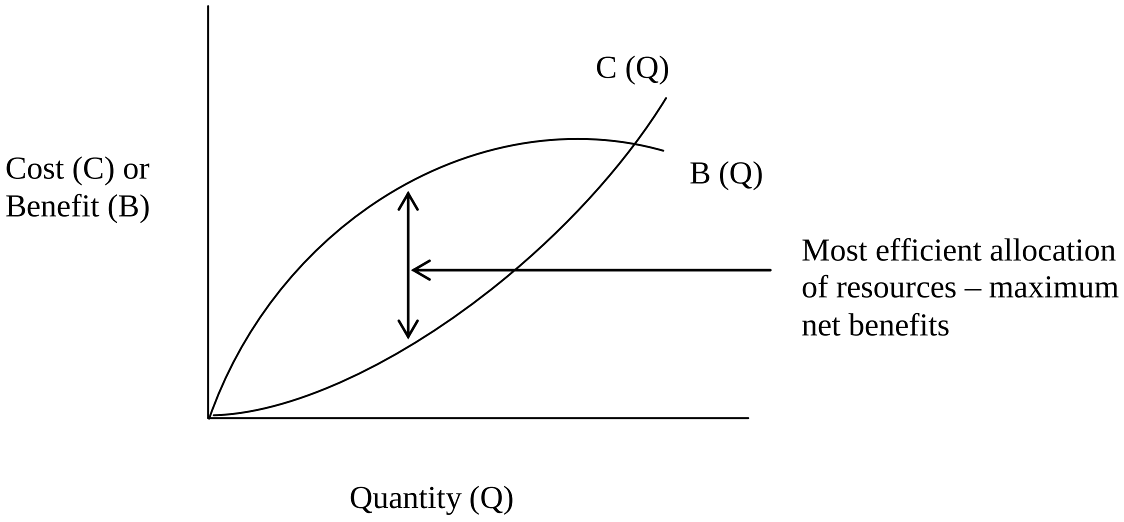

Figure 3: Aim of a CBA – a more efficient allocation of resources (adapted from Boardman, Greenberg, Vining & Weimer (2014))

Various approaches can be utilized to assess the desirability of a particular project or regulatory intervention. Depending on the type of analysis required, costs and benefits can be fully monetized, as is the case with a full cost-benefit analysis, projects with the same benefit can be ranked according to cost without monetising the benefits, or hybrid methods can be utilised when not all impacts can be reduced to monetary terms. In these cases, the monetisation of costs and benefits is done as far as practically possible, and non-monetised benefits can be scored and weighted. The difference between all monetised costs and benefits can also be determined and compared to the remaining non-monetised benefits.

Methods such as benefit-cost ratios and cost-effectiveness ratios can be used to rank projects from most to least beneficial. These are useful when decisions regarding competing projects need to be made. Cost-effectiveness analysis is useful when competing projects are evaluated that achieve a defined desired outcome. The projects are ranked on the basis of cost. Since a fixed benefit has been identified, the benefit does not require being monetised. Costs are valued with these methods, but benefits are not quantified in monetary terms (DEAT 2004).

A full cost-benefit analysis (CBA) requires monetisation of all costs and benefits. A full cost-benefit analysis is the methodology of choice for the present analysis as it is a widely accepted methodology for assessing regulatory impacts and is also referred to by the National Framework for Air Quality Management in the Republic of South Africa (DEA Government Notice 919, November 2013). The framework defines a cost-benefit analysis as "the process that involves weighing the total accepted costs against the total expected benefits in order to choose the best option".

Extensive literature exists that describes CBA methodology is available and the reader is referred to these volumes for a full description (US-EPA 2010; Heinzerling & Ackerman 2002; Jalaludin, Salked, Morgan, Beer & Bin Nisar 2009). Boardman *et al.* (2014) set out the steps in CBA as follows:

a) Identify scenarios
b) List which costs and benefits will be evaluated
c) Identify impact categories and select measurement indictors
d) Predict the impacts quantitatively
e) Monetize all impacts
f) Discount costs and benefits to obtain present values
g) Compute net present value for each alternative
h) Perform a sensitivity analysis
i) Recommendation.

In a manual compiled for cost-benefit analysis in the South African water sector, the steps in the execution of a CBA are set out by Mullins, Mosaka, Green, Downing & Mapekula, 2007. The steps are similar to the steps set out by Boardman *et al.* (2014). A similar process is set out in the South African Department of Environmental Affairs CBA Guideline (DEAT, 2004). Jalaludin *et al*. (2009) conducted a review of European, US and Australian cost-benefit analyses conducted to determine the appropriate methodology for calculating health benefits associated with reductions in ambient pollutants levels.

A cost-benefit analysis can either utilize a top-down approach or a bottom-up approach. The choice of method will depend on the purpose of the study, the availability of relevant input information as well as the level of site-specific resolution required (Van Horen 1996).

A top-down approach aggregates data and assigns an average cost to emissions based on health impacts, for example by assigning a monetary value to a ton of emissions (Van Horen 1996). This approach can be utilized where data is scarce, as the data requirements are not as intensive. The approach does not take site-specific information into consideration when assigning a monetary value to emissions and may not be reflective of actual damage costs if the local factors are different from those on which the aggregate data is based (Sunqvist 2004). The process is shown in Figure 4.

Figure 4: Steps in a top-down CBA

A bottoms-up approach follows a type of impact pathway analysis and calculated damages based on the ambient impact of the emissions (Van Horen 1996). Such an approach is data intensive as information is required for all stages of the analysis: emissions data, data on dispersion of pollutants, number of receptors, concentration-response information to calculate the economic impact of the emissions (Van Horen 1996; Sunqvist 2004). Site-specific factors such as dispersion of pollutants and size and sensitivity of affected populations can therefore be taken into consideration. This approach has the disadvantage of being data intensive and may not be appropriate where significant data gaps exist (Van Horen 1996). The steps in the analysis are shown in Figure 5.

Figure 5: Steps in a bottoms-up CBA

Generally, the cost of regulation is taken to include only the direct cost (Jalaludin *et al.*, 2009). However, in deciding whether a regulation is preferable or not, consideration can be given to whether the risks disproportionately affect a certain population group, such as the elderly, very young or minority or disadvantaged groups. Further considerations are: the impact of the risk, dreaded risks and risks that are not well understood. Regulators may attach a weight to these impacts when determining the desirability of a regulation Jalaludin *et al.*, 2009).

2.2.3 Valuation methods in environmental economics

In order to be included in a cost-benefit analysis, an impact or externality has to be expressed in monetary terms. Where market values for an impact exist, these should be used in the analysis. In the case of externalities, markets often do not exist and estimates are required.

In the analyses of costs, markets often exist to value an input. However, the market value may not necessarily be reflective of the true cost of the input. As an example, consider the cost of water or electricity in South Africa. Since the market prices for electricity and water resources in South Africa are based on historical costs and not necessarily reflective of the real cost of supply, alternative valuations should be used to monetize such impacts (Mullins *et al.*, 2007). Since the economic value of water is higher for industrial and domestic use than for agricultural use, water can be assumed to be diverted from agriculture in favour of these uses in a water-scarce environment (Mullins *et al.*, 2007). The cost of water should then at least be valued at the opportunity cost of the water from reduced agricultural or irrigation (Mullins *et al.*, 2007).

The analysis of benefits often requires the use of non-market valuations. The benefits people enjoy from environmental protection can be related to health, ecological impacts or material damage (Stavins 2007). Monetisation of impacts can be done by either direct or indirect methods as indicated in Figure 6 below. Different externalities arising in a study may require different approaches to quantify the externality and one single approach may not be adequate.

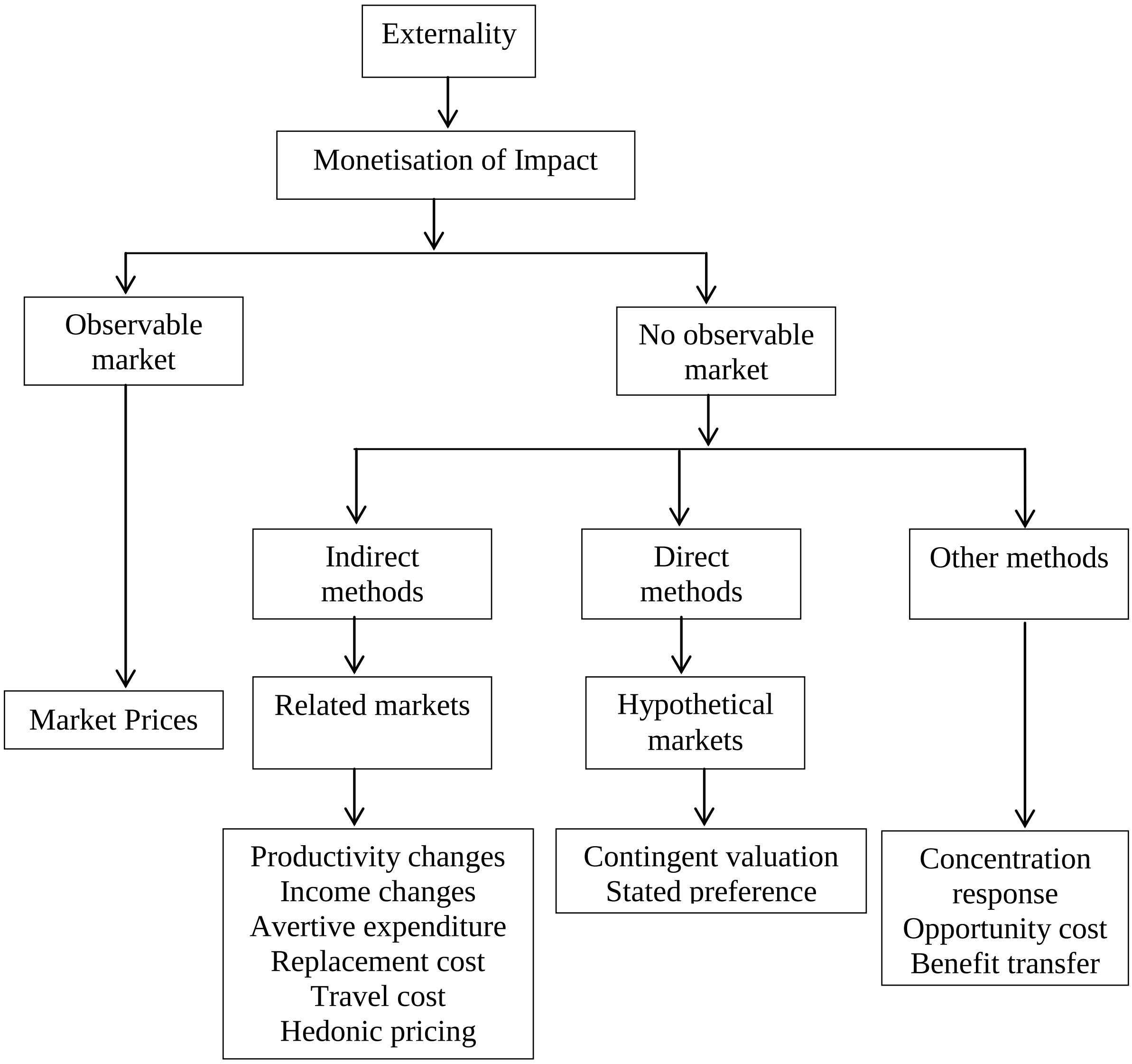

Figure 6: Techniques for the monetisation of benefits. Source (adapted from Sundqvist 2000)

The methods for the monetisation of impacts are discussed in the sections below.

a. Abatement cost approach

The abatement cost approach uses the cost of abatement of an environmental damage as an implicit value for the damage avoided (Sundqvist 2000) as a regulatory revealed preference or shadow pricing. This approach assumes that the cost of abatement was seen as reflective of the value society places on the benefit. This further implies that the abatement required was set at the level where marginal cost of abatement was equal to marginal cost of the damage. If the standard set by the regulators is not set at this most efficient point, the use of abatement costs is not meaningful. Practically, the abatement cost is not related to the benefit realized and merely informs the one side of the cost-benefit analysis (Van Horen 1996).

c. Stated preference methods

Stated preference methods question the monetary value respondents would assign to an externality. The techniques consider the willingness of a respondent to accept compensation for a loss, willingness to accept (WTA) the damage or willingness to pay (WTP) to avoid the damage. The researcher creates an artificial market for an externality. Contingent valuation methods require respondents to trade something for an environmental change, often involving the use of questionnaires (Van Horen 1996).

The welfare of a person can be stated in terms of Utility (U), which is dependent for present purposes on a vector of consumption possibilities (q) and an environmental quality index (z) (Sundqvist 2000) as shown in Equation 1.

$$\boldsymbol{U_i = U_i(q; z)} \qquad \text{Equation 1}$$

A change in environmental quality from z_0 to z_1 can be described by Equation 2:

$$\boldsymbol{U_i(q_0, z_1) - U_i(q_0, z_0)} \qquad \text{Equation 2}$$

The compensation for a change in welfare due to the change in an environmental quality index can be described in Equation 3:

$$U_i(q_0, z_0) = U_i(q_0 - CV, z_1) \qquad \text{Equation 3}$$

CV is the amount of money associated with a move from z_0 to z_1, and therefore the amount of money an individual is willing to pay for an improvement in quality or the amount an individual is willing to accept as compensation for a deterioration (Sundqvist 2000).

How a society values a particular resource depends on the intrinsic value that the resource has for that society and may differ between different income groups and socio-economic groups. Furthermore, such studies aggregate the responses from study participants without necessarily adjusting for differences in income levels (Van Horen 1996). This is particularly challenging in the South African context as a certain monetary value assigned by a lower income individual could have a higher intrinsic value to that person than the same monetary value assigned by a higher income individual.

The willingness to pay or accept will differ based on the individual's willingness to accept risk. Furthermore, the willingness to accept risk may be dependent on whether the risk is internal (by choice of the individual) or external (risk due to the actions of another party).

d. Revealed preference methods

The willingness to pay to avert a negative environmental impact can be estimated from people's behavioural response to change. This is also referred to as the averting behaviour method. Actions are taken by individuals to lessen an environmental impact. From these actions, a willingness to pay can be derived (Stravins 2007).

The value of resources may be estimated by an individual's decision to visit that resource. An individual may pay to visit a resource (willingness to pay) or expend money to travel to a resource (travel costs), also named the travel cost method. These models can be used to estimate use value of a resource, but not non-use value (intrinsic value) of a resource. Furthermore, the opportunity cost of leisure time (visiting a resource) is not easily calculated (Van Horen 1996).

e. Hedonic pricing methods

Hedonic pricing methods derive valuations for environmental factors from markets in related goods or services, such as wages and property prices (Van Horen 1996). The increased price that a consumer is willing to pay for a cleaner environment is then used to calculate the benefit of such a cleaner environment. In the context of a developing country, property prices are often not indicative of environmental factors, as housing markets are either non-existent or informal for a large portion of the population (Van Horen 1996).

The hedonic wage method utilises the increased compensation offered for employment that carries higher risk. By calculating the increased wages as a result of increased risk, a person's willingness to accept compensation for risk can be estimated through benefit transfer. From this the value of a statistical life can be calculated and utilised for environmental studies (Stravins 2007). This method does have limitations, such as the difference between actual risk and perceived risk, differing risk appetites of individuals and imperfections in the labour market, which may skew the results.

This method assumes that labour markets function efficiently in that wages are reflective of risk (Van Horen 1996). This may not be the case in a country with high unemployment rates, such as South Africa, where labour supply far outweighs demand.

f. Opportunity cost

This method values an environmental impact by calculating the loss in income resulting from an environmental impact. (Van Horen 1996). As an example, the loss in agricultural production due to pollution of an irrigation source can be used as a proxy for the damage caused by the pollution. Similarly, the loss in income resulting from death or illness due to air pollution can be used to determine the damage cost of the air pollution.

g. Benefit transfer

Benefit transfer is not a valuation method in itself, but makes use of monetary estimates of previous studies and adapting the values for present purposes. The values are adjusted for use in the context of the study (Sunqvist 2000). This method involves the use of economic, health, engineering or environmental data from a site and applies the data to a different site (Burtraw, Krupnick & Sampson 2012). Benefit studies are useful when insufficient data exist for a study area or insufficient resources are available for a primary study. Furthermore, benefit transfer can take advantage of the expertise embedded in such studies (Burtraw *et al.* 2012). The value of a benefit transfer will be dependent on the similarities between the two sites, for example similarities in populations, epidemiological relationships, dispersion characteristics, etc., depending on the data transferred (Burtraw *et al*. 2012).

Benefits (B) can be adjusted for differences in income levels using the per capita income (I) and income elasticity (γ) as shown in Equation 4:

$$\boldsymbol{B_a = B_b \left(\frac{I_a}{I_b}\right)^{\gamma}} \qquad \text{Equation 4}$$

2.2.4 Considerations in economic evaluation

The valuations of impacts necessarily involve attaching a monetary value to non-market goods such as human health and ecosystem services. Practically, many of these valuations can only be done in an imprecise manner, some far into the future with complex cause-effect relationships, which introduces risks and uncertainties into the analysis (Staehr 2004). In cases such as the valuation of human health and mortality, these estimates can be contentious. Other factors such as the choosing of which benefits to assess, the timeline of the assessments as well as the timeframe of the impact have to be taken into consideration when conducting a cost-benefit analysis. The most significant of these considerations are discussed in this section.

a. Risk and uncertainty

In order to address risk and uncertainty within the analysis, the CBA can be adjusted in various ways. One way to address risk is to introduce sensitivity analysis. Gross sensitivity analysis is conducted by evaluating the impact of a change in one variable on the NPV (Staehr 2004). A stress test can be used whereby best/worst case scenarios are calculated, also known as analysis of extremes (Staehr 2004). The variables can then be ordered so that the variables are arranged from top to bottom with the largest negative impact on NPV at the top (Staehr 2004). This can indicate variables that can be considered a cause for concern, but does not indicate the likelihood of the specific outcome. Analysis, taking into consideration the likelihood of an outcome, can provide more information than a stress test or a gross sensitivity analysis, such as a Monte Carlo simulation by which a distribution of NPVs is calculated using likely distributions of the input variables (Staehr 2004).

b. Choice of benefits to assess

The benefits to be assessed may include direct as well as indirect benefits. Care should be taken when including indirect benefits and opportunity costs in the analysis. To ensure a robust result, the multiplier effect of benefits should not be included unless there is an indisputable benefit. By adding sufficient multiplier benefits or secondary benefits, an unfeasible project may appear to be feasible. Costs and benefits should be clearly identified and double counting of these should be avoided (DEAT 2004). Sunk costs should not be included, only costs that are relevant to the project. Costs that would have been expended regardless of the project should therefore not be included. Similarly, costs that have already been internalized should not be included in the analysis, as these costs have been included in market prices.

e. Economic impact

In order to assess the economic impact of a regulation, partial equilibrium models can be used where the regulation affects a single sector or a small number of sectors and general equilibrium analysis should be conducted when a large number of sectors can be impacted US-EPA 2004). The estimation of total costs should include the opportunity cost of capital along with direct costs, however care should be taken to not double count impacts (US-EPA 2004). In instances where the regulation impacts the larger economy, or where long-term impacts should be analysed, general equilibrium models should be used (US-EPA 2004). In order to assess the distributional effects of a regulation, a further economic impact analysis can be conducted (US-EPA 2004).

In a manual developed for cost-benefit analysis in the South African water sector, Mullins *et al*. (2007) note that while it is not the purpose of a CBA to determine the macroeconomic impact of a project, the measurement of secondary effects using a general equilibrium approach could be done. In the South African context, large differences exist between the incomes of high- and low-income groups, which may result in analyses using weighting factors. Mullins *et al.* (2007) recommend that first-round analyses should be conducted without weighting and that subsequent analyses can then use weighting systems to demonstrate the impact on lower income groups.

f. Use of discount rate for evaluation of benefits

The timelines of these benefits and costs of a project, or intervention, may range from short term (daily or annual) to long term (>10 years). The evaluation of environmental projects is complicated by the fact that costs and benefits may have significantly different timelines; for example capital expenditure may be a short-term cost in relation to the long-term benefit of preserving the environment (US-EPA 2004). Furthermore, the availability of scarce environmental resources can be expected to decline, placing a higher long-term value on these resources (DEAT 2004).

In order to be directly comparable, benefits and costs have to be discounted to their present value. The present value of net benefits (PVNB) is defined as shown in Equation 5:

$$PVNB = \sum_{t=0}^{T}\left((B_t - C_t)(1+r)^{-t}\right) \qquad \text{Equation 5}$$

Where B_t (or C_t) is the benefit (or cost) at time t and r is the discount rate, with T the terminal year of the analysis. Future impacts are thus translated to present values (Stavins 2007).

Choosing a discount rate should be done with care, as it can greatly influence the outcome of the analysis. Theoretically, the discount rate used should be based on the aggregate individual time preference rates of all individuals affected by the change (Stavins 2007). Practically, preferred discount rates are dependent on whether a gain or a loss is evaluated, the size of the impact and the time span involved (Stavins 2007). Discounting of future generations' rights necessarily has attached a long timeline, which, if discounted at too high a rate, may be insignificant if discounted to present values, contrary to the principle of sustainable development (Mullins *et al*. 2007).

Boardman *et al.* (2014) note that the choice of discount rate can be contentious and a sensitivity analysis on the discount rate is recommended, but nevertheless recommend a discount rate of 3.5% for projects with a lifetime of <50 years and the use of a time-declining discount rate for intergenerational projects. Discount rates ranging from 0 to 6% were used by ExterE, while Rowe *et al.* (1995) used a discount rate of 3% (Burtraw *et al.* 2012).

Mullins *et al.* (2007) note that due to South Africa's low savings rate, the discount rate can be assumed to be low. The interest rate was noted to not be an accurate indicator of the true shortage of investment capital in South Africa. The manual notes that high discount rates (10 – 12%) discount long term environmental impacts into insignificance and are therefore nor appropriate. The study recommends using a discount rate of 8% to assess environmental benefits specifically and using sensitivity analysis at lower rates.

Intragenerational discounting refers to projects with timeframes that can extend to decades, but do not impact further generations, while intergenerational discounting applies to projects with long timeframes where the costs or benefits are expected to impact on future generation (US-EPA 2004). Discounting has a greater impact on projects where the timing of costs and benefits are significantly different and has a lesser impact where costs and benefits are along the same timeframes, where costs and benefits are largely constant over time or do not change over time (US-EPA 2004). Therefore, policies that will realise a benefit in the long term, such as climate change, groundwater pollution or biodiversity improvements, will be more sensitive to discounting and the discount rate used (UP-EPA 2004). Costs and benefits should be discounted in the same manner (US-EPA 2004). It is generally recommended that sensitivity analyses be conducted using different discount rates (DEAT 2004; US-EPA 2004; Mullins *et al.* 2009; Jalaludin *et al.* 2009).

e. Valuing health and life

To assess the impact of pollutants, concentration-response techniques calculate the impact of pollution on a receptor and assign a value to the response of the receptor. The relationship between a pollutant and a specific outcome is used to evaluate the impact. A cost is then assigned to the impact. This is the approach often used in health impact studies (Ostro 2004).

Cost-benefit analyses have been conducted in developed countries to evaluate the desirability of policies for many years (Rowe, Lang, Chesnut, Latimer, Rae, Bernow & White 1995; Holland, Watkiss, Pye, de Oliveira & van Regemorter 2005; De Mocker 2011). In the United States, the dominant policy approach until the 1980s was to value benefits of reduced mortality in terms of lost earnings and medical expenses (Viscusi & Aldy 2002). Subsequently, it has become the norm to utilize the value of a statistical life

(VSL) in economic evaluations. VSL estimates are based on cost of death, which is much higher than using lost earnings (Viscusi & Aldy 2002). The VSL is not equivalent to the actual value of a person's life, but is calculated based on the WTP of WTA a marginal increase in risk (Viscusi & Aldy 2002). A monetary value can then be assigned to mortality risks.

The VSL is often determined by using hedonic wage or hedonic price models. In hedonic wage models, the VSL is determined by analyzing the compensation in the form of higher wages that an individual is willing to accept in exchange for increased on-the-job risk (Viscusi & Aldy 2002). Various errors and uncertainties are recognized using these models, for example the failure to capture all the determinants that influence an individual's wage. One example is that risky jobs may also be undesirable for other reasons, such as unpleasant working conditions or physically demanding jobs (Viscusi & Aldy 2002).

The VSL so determined is reflective of the preferences of workers in a given sample and transferring of such estimates should take into consideration that different populations have different preferences in terms of risk (Viscusi & Aldy 2002). Viscusi & Aldy (2002) note that careful consideration should be given to the challenges of applying an estimate from literature to a potentially different risk or population. The evaluation of 60 studies from 10 countries undertaken by Viscusi & Aldy (2002) indicated that estimates from the US and Canada were similar, with estimates from the UK being significantly higher and estimates from developing countries lower. The lower values can potentially be explained by the fact that safety acts as a normal good and poorer counties will therefore have a lower VSL estimate (Viscusi & Aldy 2002). The study further notes that VSL has generally been shown to decrease with age, but that the value of an additional life year increases (Viscusi & Aldy 2002). It would therefore be possible to calculate an age-adjusted VSL by using life expectancy and discounting (Viscusi & Aldy 2002).

Assigning a monetary value to mortality risk can be controversial. While concerns can be noted, it should be recognized that individuals make risk trade-offs in everyday life, as more fully discussed in Viscusi & Aldy (2002), and as can be generally observed, for example the wearing of a seat belt or use of a car seat for infants. Criticism in terms of discounting of VSL is equally not grounded, as it is not lives that are discounted, but society's willingness to pay for future risks (Viscusi & Aldy 2002).

Where country-specific VSL estimates are available, these should be used (Jalaludin *et al.* 2009). The review by Jalaludin *et al.* (2009) further recommends valuing mortality in terms of life years lost rather than in terms on the number of premature deaths, where possible, and further recommends using sensitivity analysis to determine the differences using the two approaches.

h. Reversibility of the impact

The value of ecosystems is complex and ecosystem loss may be an irreversible loss. Individual preferences are subjective and may not be reliable indicators to assess the importance of valuing such an environmental externality. The role of the decision maker is then to oversee that rights are protected. This is particularly important where the impacts are future impacts, which may not be adequately quantifiable in the present, such as ecosystem losses. When damages are considered irreversible, the precautionary principle is often applied whereby projects that may reduce the risk of damage are implemented. The calculation of a correct NPV is, in this case, dependent on appropriate information. The project can then either be undertaken in accordance with the precautionary principle, or can be delayed in order to source the required information. Within this context, projects can also be seen as being irreversible, as not undertaking the project can have irreversible negative consequences.

2.3 INTERNATIONAL STUDIES

The methodology used in selected international studies relating to air quality is summarized in this section. The studies reviewed were selected as they are applicable to the current study and reflect different methodologies that can be utilized. Even though most studies considered the impacts of multiple pollutants, the review below focuses on the SO_2 impact.

2.3.1 Empire State Study (Rowe *et al.* 1995)

The study by Rowe *et al.* (1995) aimed at estimating the true cost of electricity generation over 23 new and relicensed electricity generation sites. The study investigated sites in New York State, USA, and included impacts from nuclear, coal, oil, gas, biomass incineration and wind generation.

The study adopted a bottoms-up approach to develop a model (EXMOD) for estimating environmental externalities associated with electrical resource options in New York State. The following steps were followed Rowe *et al.* (1995):

- Emissions and resource use were quantified.
- Changes in environmental quality were predicted.
- Environmental and social impacts were assessed.
- Changes in well-being or damages were quantified and measured by willingness to pay.
- Damages were aggregated across effects, individuals and time.

Externalities were screened and grouped according to their impacts and availability of information relating to the externality. Externalities were categorised as follows (Rowe *et al.* 1995):

- Category 1: Externalities for which damage functions were developed (these were included in the model)
- Category 2: Externalities that were assigned a $0 value, because they were considered to have a small impact
- Category 3: Externalities to be developed at a later stage

- Category 4: Externalities for which insufficient data existed to develop a method of defensible quality

The emissions of greenhouse gasses, for example, were included in Group 4, due to the significant uncertainties in developing a damage function (Rowe *et al.* 1995).

The dispersion models utilized to determine the changes in environmental quality were annual average or simple peak models, as annual average damages over multiple-year time horizons were computed and utilized in the study (Rowe *et al.* 1995).

Key assumptions made in the study include assumptions regarding populations, thresholds under which no health effects were assumed, and environmental impacts. The study assumed that overall populations were fixed and populations were not expected to grow significantly. Mortality risk valuations were based on wage risk trade-offs (hedonistic model) and all mortality risks were evaluated using the same valuation (Rowe *et al.* 1995). Non-use values were considered to be more significant for resources that are unique or cannot be replaced compared with use values of these resources. In this study, an assumption was made that the value of ecological benefits is based on direct effects on individuals. Due to limited information on non-use value for many impacts, non-use values were only included for visibility impacts and for groundwater contamination (Rowe *et al.* 1995).

For this study, positive discount rates were used. Freeman, quoted by Rowe *et al.* (2005), recommends a discount rate of 2-3% based on evidence from financial markets, recommendations by Congressional Budget Office and studies into the trade-off between higher wages and loss of future life through the increased risk of accidental death on the job (Rowe *et al.* 1995). The range of 2-3% is particularly appropriate for assessing impacts within the same generation, but is more controversial when impacts are cross generational. A discount rate of 3% was used in this study and the rate fixed for all externality endpoints. Since most of the impacts were within 40 years of exposure, this was considered to be within one generation (Rowe *et al.* 1995).

Sensitivity analyses were done for selected parameters. Uncertainties were specified for key assumptions and model components within each externality endpoint computation. Uncertainty was reflected in the model as *high*, *central* and *low* values. Central values are considered best estimates with high and low values plausible alternatives. Uncertainty in the model accumulates as individual components are added to estimate endpoints. Uncertainty computations were then aggregated for individual externalities to estimate total damages.

A sensitivity analysis was conducted on the use of threshold values for ambient concentrations of pollutants (Rowe *et al.* 1995). Health impacts at lower concentrations were assumed to only occur above a certain threshold, below which effects were assumed not to occur. Above the threshold, known concentration-response functions were extrapolated from the threshold value. Linear approximations to non-linear concentration-response functions were used in this study. The results of the study indicated that the use of a health effects threshold was found to have a significant effect on the results.

In terms of SO_2, externalities related to morbidity, impact on vegetation and impacts on groundwater, and were considered to have a very small damage associated with them (Rowe *et al.* 1995).

2.3.2 Cost-benefit analysis of policy option scenarios for the Clean Air for Europe Programme (Holland *et al.* 2005)

The purpose of this study was to assess the costs and benefits associated with pollution-reduction policies and to conduct an analysis of the scenarios generated by the Clean Air for Europe (CAFÉ) Programme (Holland *et al.* 2005). The project was conducted for the 25 EU countries. Policy scenarios ranged from current legislation (CLE) to maximum technically feasible reduction (MTFR) with scenarios (A, B and C) constituting increasing degrees of control in between. The study considered $PM_{2.5}$, SO_2, NO_x, VOCs, and NH_3. A bottoms-up approach was followed.

Direct and indirect impacts were considered. The direct impacts considered included tropospheric ozone formation (effects on health, crops, materials and ecosystems), health impacts from primary and secondary pollutants, ecosystem acidification, ecosystem eutrophication, and damage to buildings and other materials. The indirect impacts included changes in greenhouse gas emissions, wider social and economic impacts and measures for their control. Not all of these impacts were valued, however some, such as change in greenhouse gas emissions, were quantified, but not monetized. The impacts on cultural assets, visibility, social and altruistic impacts of health impacts and the direct effects of SO_2 were not included in the study.

The EMEP model was used to model the impacts across Europe on a 50 x 50 km grid. The study envisaged that a more detailed analysis can be done on finer scale (city specific, for example). To evaluate the health impacts and ecological impacts, the RAINS model was used (Holland *et al.* 2005). The RAINS model was used to quantify critical load exceedances, but these impacts were not monetised. Only the health impacts of PM and ozone were considered (Holland *et al.* 2005).

In order to evaluate the economic impacts of each scenario, a General Equilibrium Model (GEM – E3) for the EU member states was used. The model indicated that the macro-economic cost is limited when compared to health and ecosystem benefits. With stricter control, the benefits do not justify the impact on GDP (MTFR scenario). The impact of competitiveness was limited, as all states participate and the price effect is small. A discount rate of 4% was used to annualise investment costs (Holland *et al.* 2005).

The study found that health impacts, damage to crops and damage to buildings can be valued and that impacts such as acid deposition relative to critical load can be quantified, but not monetized (Holland *et al.* 2005). In order to monetize the health impacts, VSL and VOLY were used (based on EU WTP estimates) for mortality assessments. It was recommended that adjusted VSL be used for infant mortality (one, one point five and two times the adult VSL). Cost of illness (COI) plus cost of absenteeism were used for morbidity assessments (Holland *et al.* 2005).

The results indicated that the direct impact of ozone on agricultural production was likely to be small. Significant health impacts were associated with PM. The health impacts of SO_2 were not considered significant in comparison to health effects associated with PM and ozone (Holland *et al.* 2005).

The study identified impacts that were regarded as likely significant in the European context: effects of ozone, acidification and eutrophication on forests and biodiversity, acidification and impacts on invertebrates and fish (Holland *et al.* 2005). The following impacts were identified as potentially significant in the European context: social and altruistic impacts of air pollution on health, chronic health effects of ozone, impacts on cultural heritage, interactions between pollutants and pests/pathogens (Holland *et al.* 2005). The following impacts were identified as unlikely to be significant in the European context: visibility, effects of acidification on groundwater, visible injury to crops, indirect impacts on livestock, effect of SO_2 on morbidity and direct effects of VOCs (Holland *et al.* 2005).

In all scenarios, the benefits were found to outweigh the costs, although benefit cost ratios deteriorated as stricter controls are added (moving further from CLE to MTFR) (Holland *et al.* 2005).

The study methodology was peer reviewed and reviewed by other stakeholders (Krupnick, Ostro & Bull 2004; Gephart, Lewis, Roberts, Salter & White 2005). The peer review concluded that the methodology was sound, but made recommendations specifically related to the use of a single VSL for adults and children as well as the appropriate method for using VOLY (Krupnick *et al.* 2004). CONCAWE made further recommendations with regard to VOLY vs VSL (Gephart *et al.* 2005).

2.3.3 Baltic States Study

The study assessed the externality costs of electricity generation in Lithuania, Latvia and Estonia (Streimikiene, Roos & Rekis 2008). The study evaluated the damage costs of the effects of emissions on human health and on crops and materials, using an impact pathway or bottoms-up approach (Streimikiene *et al.* 2008). The study selected a bottom-up methodology, as the impact of emissions is seen to be very site dependent (Streimikiene *et al.* 2008). For crops and materials, market prices were used to calculate damage costs. For human health, fatal and non-fatal outcomes, WTP or WTA methodology was used. The standard price approach was used and improved to determine the costs of environmental externalities such as eutrophication and acidification, for which an existing software package, EcoSense, was used. The study assessed the impacts of PM_{10}, $PM_{2.5}$, VOCs, NO_x, NH_3 and SO_2. The study found that high externality costs are associated with the use of high sulfur fuels due to the impact on human health, loss of biodiversity and impact on materials (buildings) (Streimikiene *et al.* 2008).

2.3.4 Muller study

The study aimed to estimate air pollution damages for various industries in the United States (Muller, Mendelsohn & Nordhaus 2009). The value-add of the industry, based on revenue, was used to determine the ratio of external costs to value-add. The industries were ranked according to this ratio. The study found that several industries, including coal-fired power generation, have external damage costs higher than their value-add. This does not necessarily indicate that the industry is undesirable, but rather indicates that additional output from that industry would result in damage costs higher than its revenue. Understanding the externality costs of an industry can be useful to inform environmental regulation to ensure that sources with highest damage costs can be prioritized (Muller *et al.* 2009).

The model took into consideration emissions of six criteria pollutants and their impact on human health, decreased agricultural yields, reduced visibility, accelerated depreciation of materials and reduction in recreational facilities. For the power generation industry a sensitivity analysis was done with and without the external costs of CO_2 emissions.

The study acknowledges the uncertainty of valuing impacts on human health and particularly mortality (Muller *et al.* 2009). The study compared two methods of calculating mortality costs. The first method was a uniform valuation method. This valuation method uses the VSL at a fixed cost, regardless of the age. The second method, preferred by the authors, was to differentiate the VSL according to age. Premature mortality is then treated in terms of life years lost and is the preferred method as younger people will forego more life years on premature mortality than older people whose life may be shortened by relatively fewer years. This method places a larger value on young relative to older people due to higher life expectancy. For the evaluation based on foregone life years, the VSL was used to calculate the value per life year. A discount rate of 3% per year was used. The economic cost for each age group was calculated based on the discounted value for their remaining life years weighted by the probability of each age group surviving to the next age group (Muller *et al.* 2009).

The study used a VSL of $6 million per statistical life (2 000 dollar value), based on the mean of 26 studies (Viscusi 1992), as used by the US-EPA in determining the cost benefit of the Clean Air Act.

The study showed that evaluating mortality costs using a single VSL value not adjusted for remaining life years showed a higher external damage cost than the adjusted life years approach (Muller *et al.* 2009).

The study noted three areas of uncertainty namely the method to calculate the value of life or life years, the impact of PM emissions on mortality and the value of CO_2 emissions (Muller *et al.* 2009).

2.3.5 Regulatory Impact Statement: New South Wales (NSW 2010)

A regulatory impact study (RIS) was conducted by the New South Wales Government (Environment, Climate Change and Water Department) prior to the promulgation of the Protection of the Environment Operations (Clean Air) Regulation 2010 (POEO). A RIS is required when regulation is to be remade (with or without amendments) that examines the economic and social costs and benefits of regulatory proposals and their alternatives. NSW has a developed system of environmental regulation and has the largest population and the largest proportion of industrial activity in Australia (NSW 2010).

The study considered all sources under regulation in NSW and compared regulation under POEO Act with a no-regulation scenario. Under a no-regulation scenario each facility would have to comply with the emission limits that were set in their licenses prior to the commencement of the regulations (NSW 2010). Costs and benefits were calculated per "activity" including vehicle emissions, domestic burning (wood heaters) as well as industry (NSW, 2010). An impact assessment was conducted for each part of the regulations. Part 5 of the regulations govern emissions from Industrial activities. Part 5 sets out maximum emission levels plants have to achieve, according to the age of the facility. The maximum emission limits are supplemented by a licensing mechanism that can require facilities to comply with the maximum standard in Part 5, or more stringent standard should local conditions require. At the time of the study, an estimated third of all licenses contained site specific emission standards (NSW 2010).

The impact statement focussed on NO_x and PM emissions, as ambient concentration of ozone and PM are of greatest concern when compared with levels required by the ambient air quality standards (NSW 2010). According to the study, industry contributes the largest proportion of NO_x and a significant portion of PM emissions and therefore these pollutants have been selected as the basis to calculate the benefits of the regulation in terms of health costs. The regulations do not prescribe limits for SO_2, instead requiring the use of low sulfur fuels. The impact statement acknowledges the additional benefits achievable by regulating other pollutants, particularly due to their toxicity levels, but states that it is not possible to separate out the benefits of their regulation due to (NSW 2010):

- similar health impacts that could lead to double counting;
- these pollutants are already being regulated under site-specific standards, and benefits specifically allocated to the proposed regulation would be difficult to allocate; and
- occupational health and safety standards may require the reduction of some pollutants, such as acid mist, H_2S and F.

The following were mentioned as unquantifiable benefits of the regulations (NSW 2010):

- decreases in offensive odours;
- avoidance of damage to vegetation, crops and buildings;
- improved visibility; and
- improved human health from reduction of pollutants not quantified in the study.

The study indicated that negative health and environmental effects from SO_2 emissions are more limited in NSW than is the case internationally due to the regulation of fuel sulfur content (NSW 2010). However, the largest impact from SO_2 was found to be coal-fired power stations. The study highlighted the following negative impacts from increased ambient SO_2 levels (NSW 2010):

- direct impacts of SO_2 on human health;
- secondary particulate formation as well as the adsorption of SO_2 onto fine particulate matter and the associated impact of the resultant $PM_{2.5}$; and

- impact on vegetation – plants become less productive, lose foliage or die prematurely (concentrations at which sensitive species show signs of damage were quoted).

The study used a top-down approach and used the health damage costs associated with each additional ton of pollutant released (NSW 2010). This approach assumes that the site forming the basis of the study is not significantly different from the site from which data is obtained. The benefit transfer approach was used in this study, as the bottom-up approach requires significant amounts of data, much of which was not available at the time of study (NSW 2010). Damage costs estimates from the US and EU as well as damage costs estimates from local health impact studies for the reduction of transport-related air pollution were considered for this study. The Australian damage costs for the health impacts of transport related air pollution were used to estimate damage costs in this study. These damage costs were found to be comparable with the EU and US numbers (NSW 2010).

The study found that the benefits of regulation exceeded costs and recommended that the regulations be implemented due to the large benefits already monetised, as well as the additional unquantifiable benefits (NSW 2010).

2.3.6 Clean Air Act Impact Assessment

The US-EPA undertook a cost-benefit analysis of the Clean Air Act amendments (CAAA) put in place in 1990. The study analysed the costs and benefits of the change in regulations between 1990 and 2000 across 48 US states (De Mocker, 2011). Two scenarios were compared – one without CAAA scenario in which pollution controls were frozen at 1990 levels and a second with CAAA scenario which includes changes to the regulations. The "without CAAA" scenario assumed that pollution levels would be kept at 1990 levels and that more stringent abatement would not be voluntarily implemented or required by local authorities. The study assumed that the benefits would be realised over time, as the regulations take effect. The study further assumed no difference in the demographic patterns over time and no difference in the spatial pattern of economic activity (De Mocker 2011).

The study utilised a bottoms-up approach, whereby the impacts of emissions were quantified using dispersion modelling, after which a monetary benefit was attached to health and environmental variables where sufficient information exists (De Mocker 2011). The study considered the following pollutants: PM_{10}, $PM_{2.5}$, SO_2, NO_x CO, VOCs and Ozone. The study used the Community Multiscale Air Quality (CMAQ) model to model ambient impacts. Concentration-response functions were used to translate ambient concentrations into physical outcomes using BenMap (De Mocker 2011). Where sufficient information does not exist to do a quantitative analysis, a qualitative assessment of the outcomes was done. The study notes that the modelling step is a critical component of the analysis, as the valuation of the benefits is based on these results (De Mocker 2011). The difference in incidence rates between the two scenarios was calculated.

At each phase of the analysis, a sensitivity analysis was done, with the aim of testing alternative methods, models or assumptions. The study divided uncertainty into two categories – potentially significant uncertainty where the effects are greater than 5% of the total effects and probably minor effects where the impact is less than 5% of the total benefit (De Mocker 2011).

To assess the impact of the regulations on markets and transactions in the economy, a Computable General Equilibrium (CGE) model was used. The model predicts the impact on Gross Domestic Product (GDP), prices and sector shifts in production. Non-market effects such as a person's willingness to pay to avoid the effects of disease were not included in the study. The model further indicates how benefits such as increased labour supply would impact on social welfare (De Mocker 2011).

The health impact assessment calculated the difference in health impacts between the two scenarios, rather than an absolute estimation of the incidence of health impacts associated with each scenario. Due to the spatial differences in ambient impact, the change in concentration at each grid cell was matched to the size of the population within that grid cell (De Mocker 2011). Base incidence and prevalence rates for health outcomes were sourced from a variety of databases, including the CDC, National Centre for Health Statistics, as well as the American Lung Association. From base incidence rates the decrease in additional cases can be calculated using the reduction in ambient pollution levels. The cost of illness was used to analyse morbidity outcomes, however the study cautions that this is an underestimation as it does not consider the utility derived from improved health status (De Mocker 2011). In analysing mortality benefits, a VSL value of

$7.4 million (2006 USD) was used. This VSL value was calculated from the mean of the Weibull function of 26 estimates (Viscusi & Aldy 2002). The study further used a 20-year lag distribution structure for reduction in risk to human health, under which 30% of the reductions occur in the first year, 50 % occur from years two to five and the remaining 20% from years six through to 20. A 5% discount rate was used in these calculations (De Mocker 2011). Alternative distributions for the health impact assessment were tested for the purpose of sensitivity analysis. The results of the sensitivity analyses indicated that the impact of the lag function chosen was lower (-22% to + 16%) than the sensitivity of selecting a different concentration response function used (-41% to +40%). The choice of concentration-response function was found to have a significant impact on the results (De Mocker 2011).

The results indicated a wide gap between the costs and the benefits of the regulations and found it highly unlikely that the costs would outweigh the benefits of the regulations (De Mocker 2011). The study indicated that the bulk of the benefits realised were related to the reduction in premature mortality, mostly due to the high value attached to the mortality outcome. Mortality outcome calculations were based on PM and ozone concentrations, as literature more strongly relates these pollutants to mortality outcomes. The VSL estimate used was also found to have less of an impact than the concentration-response function (De Mocker 2011).

The study noted that, although the CAA is meant to be protective of human health as well as the environment, human health impacts have been better quantified (De Mocker 2011). The National Acid Precipitation Assessment Program (NAPAP) study analysed the impact of SO_2 on building and infrastructural material such as carbonate stone, galvanised steel, carbon steel and painted wood. In such an analysis, a national inventory of sensitive materials is required. Concentration-response functions relating material damage to SO_2 concentrations are required. Material damage can then be valued. The study found that the benefits related to material damage were relatively small when compared with other benefits. The long-term impact of ecological damage was not quantified by this study. The study notes that these chronic impacts are difficult to observe and poorly understood and may be difficult to separate from other influences. This may lead to an underestimation of the benefits (De Mocker 2011). Since the impacts calculated for benefits related to nonhealth impacts were small, uncertainty analyses were not done as they would be proportionally small (De Mocker 2011).

2.3.7 Thai Study

A study conducted by Sakulniyomporn, Kubaha & Chullabodhi (2011) evaluated the costs associated with increased morbidity and mortality rates associated with power generation in Thailand for the period 2006 to 2008. The study was selected for evaluation as it was seen as reflective of a developing country study. The study compared various types of electricity generation units (71 gas-fired, two coal-fired and 10 lignite-fired, the rest oil- and diesel-fired plants).

The study used impact-pathway analysis to evaluate health costs in a bottom-up approach. Emission of criteria pollutants SO_2, NO_x, PM and secondary particulates (sulfate and nitrate) were included in the study, which excluded the impact of CO_2 emissions. The ambient impact of the emissions was modelled using the Calmet/Calpuff modelling system and Mesopuff II was used to account for chemical transformations (Sakulniyomporn *et al.* 2011).

The emission standards applicable to existing facilities are more lenient than those applicable to new facilities (facilities built after 1996). Most of the facilities evaluated were fitted with abatement equipment, most notably, flue gas desulfurisation. All the facilities evaluated comply with the Thai emission standards (Sakulniyomporn *et al.* 2011).

The study used relative risk to determine exposure response functions (ERF). These ERFs were assumed to be linear without threshold values. In addition to increased mortality risk, nine health endpoints of morbidity were included in the study. Where available, epidemiological studies conducted in Thailand were used to determine ERFs (Sakulniyomporn *et al.* 2011). Hospital admission statistics were used to determine baseline values for morbidity.

To convert health endpoints to costs, the willingness to pay (WTP), value of a statistical life (VSL) and cost of illness (COI) were used. The study did not find any local valuation for VSL and benefit transfer was used, using a US study adjusted for the purchase power parity (PPP) of the two countries as shown in Equation 6:

$$U_{v(Thailand)} = U_{v(Reference)} \times \left(\frac{PPP_{Thailand}}{PPP_{Reference}}\right)^{\gamma} \qquad \text{Equation 6}$$

With γ the income elasticity, in this study assumed to be 1.

The study found that 93% of the total costs were due to costs associated with the mortality outcome (Sakulniyomporn *et al.* 2011). The study estimated an additional mortality impact of approximately 354 deaths annually (95%CI: 82-630 deaths) associated with power-generation activities. The study further found that 83% of the total costs were associated with SO_2 emissions (Sakulniyomporn *et al.* 2011).

The study found that the factors that affected the outcome of the study most significantly were associated with the location of the facility and size of surrounding exposed population, meteorological conditions, quality of the fuel and fuel type as well as abatement installed (Sakulniyomporn *et al.* 2011). The study predictably found largest costs associated with plants that do not have abatement equipment installed. Larger costs were also associated with facilities near more densely populated areas (Sakulniyomporn *et al.* 2011).

2.3.8 EEA Technical Report 2014 (EEA 2014b)

The purpose of this report was to assign a damage cost to emissions from European industrial facilities over a five-year period from 2008 to 2012. The study included the impact of main air pollutants (ammonia, NO_x, PM, SO_2, NMVOCs) as well as heavy metals, organic pollutants and CO_2. The study did not include diffuse sources such as vehicles and households (EEA 2014b).

The study included the impact of emissions on crops and building materials, but did not quantify the impact of emissions on biodiversity and ecosystem services. The study did note that, although not included in the study due to insufficient data for their valuation, the value of certain ecosystem services could be substantial (EEA 2014b).

The study included the damage cost of carbon emissions, but notes that there is not as yet an established methodology as is the case for the main air pollutants such as PM. The study notes that there are significant uncertainties and limitations associated with the various approaches to quantify carbon damage costs (EEA 2014b. The study used the range of values based on the EU Emissions Trading System carbon price forecasts.

Due to the large number of sources that have to be assessed, average, country-specific damage costs per ton of pollutant were calculated. To account for differences in release height, stack exit temperatures and factors that affect dispersion, damage costs were calculated per sector using dispersion modelling (EEA 2014b). Emissions data for each facility was then multiplied by the sector and country-specific calculated damage cost. Country-specific damage costs were seen to vary significantly due to factors such as size of affected population, differences in dispersion patterns and atmospheric chemistry, as well as dispersion to sea (EEA 2014b).

In order to determine the damage costs of emissions, the analysis was run for a base year using dispersion modelling. The subsequent run adjusted the emissions by 15% for each pollutant, for each country. Each 15% reduction increment was then subtracted from the baseline to provide a difference in concentration at each grid cell. The concentration was then multiplied by the population in that specific grid cell to calculate a population-weighted average change in concentration. The change in concentration per ton emissions was calculated and health concentrations-response functions used to calculate impact. The damage costs assigned a dollar per ton of pollutant value to a specific sector. Damage costs were calculated by multiplying the emissions data from each facility with the value calculated (EEA 2014b).

The report notes that although health effects caused by direct SO_2 and NO_x emissions could not be ruled out, the health impacts of these emissions were quantified only in terms of their contribution to secondary particulates (EEA 2014b). The report notes that including both effects may overestimate the impact of such emissions. The impact of direct emissions of SO_2 and NO_x was seen as likely to be negligible and was assumed to not impact significantly on the damage costs per ton of pollutant. The study further notes that, in economic terms, the impact of PM on mortality and morbidity was the greatest effect of concern (EEA 2014b).

A sensitivity analysis was also conducted using two approaches to value mortality outcomes – value of a statistical life (VSL) and the median value of a life year (VOLY) (EEA 2014b).

2.3.9 Discussion

The review of international studies included studies using a bottoms-up approach (Rowe *et al.* 1995; Sakulniyomporn *et al.* 2011; De Mocker 2011) and studies using a top-down approach (NSW 2010). A bottoms-up approach is favoured when the impacts are likely to be site dependent (Streimikiene *et al.* 2008; Sakulniyomporn *et al.* 2011) while a top-down approach is preferred when sufficient data is not available (NSW 2010).

In most studies, sensitivity analyses were conducted for key variables (Rowe *et al.* 1995; DeMocker, 2011; Sakulniyomporn *et al.* 2011).The discounting of impacts was also done (Rowe *et al.* 1995; Muller *et al.* 2009).

Important factors highlighted by these studies include:

- Health effect thresholds affect the results of the study (Rowe *et al.* 1995) and should be used with care.
- Where local values are available, these are preferable (Sakulniyomporn *et al.* 2011; NSW 2010), and not locally available values were adjusted using Purchase Power parity (PPP) (Sakulniyomporn *et al.* 2011). Benefit transfer was widely used in the studies reviewed.
- The size and the location of the facility have an important impact on the impact of the facility (Sakulniyomporn *et al.* 2011; Streimikiene *et al.* 2008).
- The value of a statistical life (VSL) can be used as a fixed valuation, or adjusted for age to obtain an estimate of years of life lost (YLL). YLL estimates were preferred in some studies as it places more emphasis on the loss of a younger life (Muller *et al.* 2009), but was found to yield lower costs than using VSL (Muller *et al.* 2009; EEA 2014b).
- The addition on the impacts of various pollutants may lead to overestimation of the total impact (NSW 2010; EEA 2014b).
- The impact of SO_2 emissions on health, building, vegetation and crops was highlighted by a few studies (NSW 2010, Sakulniyomporn *et al.* 2011, Streimikiene *et al.* 2008).

- Many studies found that the majority of the impact was related to mortality impacts (De Mocker 2011, Sakulniyomporn *et al.* 2011), with impacts such as material damage being relatively small (DeMocker 2011).

- Due to a lack of information, some studies did not include ecological impacts (De Mocker, 2011; NSW 2010; EEA 2014b).

While some studies indicated that SO_2 had a significant negative impact (Sakulniyomporn *et al.* 2011; Streimikiene *et al.* 2008), other studies indicated a relatively small impact when compared with other pollutants (Rowe *et al.* 1995), while still others did not consider SO_2 due to a small perceived impact (NSW 2010). Differences in local conditions, local fuel use, ambient SO_2 levels and location of large populations relative to the facility could explain this difference (NSW 2010; Sakulniyomporn *et al.* 2011).

2.4 SOUTH AFRICAN STUDIES

The largest and most comprehensive studies relating to air quality in South Africa are reviewed in this section.

2.4.1 Van Horen Study (1994)

The purpose of this study was to assess the true cost of electricity generation by comparing actual cost of producing energy using different energy sources and assigning an R/kW-generated true cost of production. The study assessed the entire electricity generation value chain. The sources included in the study were: 12 Eskom coal-fired power stations; one nuclear station; two gas turbines; two hydroelectric; and two pumped storage units, all with a combined generation capacity of approximately 37840 MW (Van Horen, 1994). The study assessed the impacts of PM, SO_2, NO_x and CO_2.

The Van Horen study used a bottom-up approach to conduct a cost-benefit analysis of the electricity generation sector in South Africa. The study considered the entire generation value chain, including coal mining and transportation. The study categorized impacts as follows (Van Horen, 1994):

- Class 1 impacts – impacts that were deemed to be potentially significant which had sufficient data for their analysis;
- Class 2 impacts – impacts that may be significant, although insufficient data existed for their evaluation; and
- Class 3 impacts – impacts that have already been internalized or impacts that were likely to be less significant than Class One or Class Two impacts.

Firstly, the impact of coal mining was assessed. The occupational impact of coal mining related to injuries and death were found to be well documented and were considered a Class 1 impact for the purposes of the study. Occupational exposure to pollutants in the mining environment was considered to be a Class 2 impact, as insufficient data existed to quantify the economic impact thereof. The impact associated with water and air pollution from coal mining was considered a Class 2 impact, as, although potentially significant, the degree of uncertainty regarding their impact rendered these impacts unsuitable for economic analysis at the time. Other impacts were assigned to the Class 3 impact category, as their impact has been internalized through environmental management procedures (Van Horen 1996).

Next, the impact of electricity generation activities was assessed. The study noted that the water price was considerably lower than the marginal cost to provide additional water, as the infrastructure required to provide water had already been capitalized (Van Horen 1996). This was considered a Class 1 impact, as sufficient information existed to calculate the cost of this water. The impact of water pollution from power stations, on the other hand, was considered to be a Class 3 impact, as this impact was not considered significant, specifically when compared to the impact of water pollution caused by coal mining activities (Van Horen 1996).

The impact of power-generation emissions on health was considered to be a Class 1 impact and although local data was not available for evaluating these impacts, sufficient information existed to utilize existing models. Therefore, these impacts were considered Class 1 impacts (Van Horen 1996).

The impact of emissions on acid deposition was considered a Class 2 impact, as insufficient information existed to calculate the economic impact thereof. The study noted that the soils in the Mpumalanga area have varying degrees of sensitivity and that the Atmospheric Deposition Risk Advisory System (ADRAS) could be used to identify sensitive soils where damage could be significant. The study also noted that, although anecdotal evidence suggests increase corrosion rates in Mpumalanga, no data existed that could prove that this was the case (Van Horen 1996).

The study noted that visibility may be impaired on the Highveld, but that attributing the reduced visibility to a specific cause was not easy. The high degree of uncertainty as well as the non-transferability of the valuations done elsewhere meant that visibility impacts were considered to be Class 3 impacts for the purposes of the study (Van Horen 1996).

The impact of greenhouse gas emissions was considered to be a Class 1 impact, as sufficient information existed internationally to permit some quantification of the impacts (Van Horen 1996).

The following Class 1 impacts were therefore identified (Van Horen 1996):

- Coal mining: morbidity and mortality
- Impact of generation on water consumption
- Impact of generation on air pollution and associated health impacts
- Impact of generation on greenhouse gas emissions.

In evaluating morbidity risks, the opportunity cost approach was used. The evaluation of health impacts included actual expenditure on healthcare, foregone income, transport costs, medication costs, and so forth. This approach is termed the “cost of illness” approach. To evaluate the economic consequence of premature death was found to be difficult as no such studies had been conducted in South Africa. Furthermore, due to sharp income inequalities in South Africa, differential evaluations for different income groups would be problematic, and the study used a consistent evaluation for premature mortality (Van Horen 1996). The study used the data from the study by Rowe *et al.* (1994) and adjusted the data to reflect

South African income levels by using the per capita GDPs of the respective countries (Van Horen 1996).

In determining the mortality impact of coal mining, average mortality rates at coal mines were calculated and the number of deaths associated with mining calculated from the amount of coal mined (Van Horen 1996). To evaluate the economic impact, the adjusted data from the Rowe study was used (Rowe *et al.* 1994). For the evaluation of morbidity, medical costs were found to vary widely, and a range of estimates was used in the study. For lost productivity, the estimated time away from work based on the injury type, was multiplied by the average wage rate applicable to coal mines. Compensation in terms of the Compensation for Occupational Injuries and Diseases Act (COIDA) was calculated and added to the productivity and medical costs to calculate a total cost for morbidity. The valuation of morbidity outcomes was found to be small in terms of cost per kWh generated, but the study noted that these impacts could be significant on a local scale (Van Horen 1996).

In order to evaluate the cost of water consumption, estimates of the cost of supplying water from the Lesotho Highlands project were used, with high and low estimates (Van Horen 1996).

In order to evaluate the impact of emissions on health, the EXMOD model, as developed as part of the Rowe study (Rowe *et al.* 1994) was used. Technical and emissions data from Eskom's power stations were used as input to the model. Demographic data from the 1991 census data for each magisterial district was collected and included data on total population, age distribution and geographical coordinates and altitudes of districts (Van Horen 1996). Surface data from 15 local monitoring stations was used as input to the model. The impact of PM, SO_2 and NO_x was included in the health evaluation portion of the study. Concentration-response data was adjusted to reflect a larger uncertainty in the South African context (Van Horen 1996). The EXMOD model uses a zero threshold for impacts. The study found a higher number of mortality cases modelled in the <65 year age group, mostly due to the bottom-heavy age distribution of the South African population.

Mortality valuations used an averaged VSL estimate from US and EU studies to calculate high, central and low estimates (Van Horen 1996).

The study found that the economic impact of greenhouse gas emissions was highly significant, due to its regional and long-term impact and was calculated to be 10 times larger than health impact valuation when using values obtained from the IPPC (Van Horen 1996). Cost of electricity generation was found to be significantly higher than electricity price. The study recommended that the assessment of external costs should be considered in future policy decisions (Van Horen 1996).

2.4.2 Fund for Research into Industrial Development Growth and Equity Study (FRIDGE 2004)

A study carried out for the Fund for Research into Industrial Development, Growth and Equity (FRIDGE) of the National Economic Development and Labour Council (NEDLAC) in 2004 examined the social and economic impact of air pollution from the use of combustible fuels. The study considered emission sources such as domestic fuel burning, emission from industries and power generation ("dirty fuels") as well as vehicular emissions (FRIDGE 2004). Due to the episodic nature and variable, uncertain duration, the impact of biomass burning emissions was excluded from the study. The study assessed international best practice in air quality management, identified the air pollutants and sources from the categories of fuel combustion listed above, quantified the environmental benefits of selected reduction initiatives and assessed the socio-economic impact of these reduction initiatives (FRIDGE 2004).

The study focused on several geographical areas, with the emphasis on sources in each area that could affect the health of surrounding communities. The study areas were: the City of Ethekwini, including the Durban South Basin, Highveld area of Mpumalanga, the City of Cape Town, the City of Johannesburg, the City of Tshwane and the Vaal Triangle. The study included impacts from PM, SO_2 and NO_x, benzene and 1,3 butadiene, and formaldehyde, as well as a limited assessment of Pb (FRIDGE 2004).

The study only quantified health-related benefits from air pollution reduction and did not consider ecological impacts, impacts on vegetation or material damage. Concentration response functions for these last-mentioned impacts are not readily available and were found to not necessarily be applicable to local conditions. The benefits considered in the study were limited to reductions in morbidity and mortality and to improvements in worker productivity associated with pollution reduction (FRIDGE 2004).

Dispersion modelling was utilised to assess the impact of emission sources using Calpuff and the potential for health impacts established using concentration-response relationships. Health impact studies were not available in South Africa at time of the study and international studies were utilised to quantify the benefits of pollution reduction impacts. Benefit transfer was used and costs adjusted for income (FRIDGE 2004).

The spatial population data was obtained from the 2001 South African census, which distinguishes populations by age group. Predicted pollution concentrations were overlaid over the spatial population data to obtain predicted health effects using the above functions (FRIDGE 2004). The study assumed children to be <5 years of age, adults to be >20 years of age and economically active individuals to be between 20 and 65 years of age (FRIDGE 2004).

Medical costs for respiratory diseases and burns were not available from the South African Department of Health and medical aid data was utilised to determine health costs (FRIDGE 2004).

The study found that the most significant health impacts could be attributed to the following activities, across all study areas (FRIDGE 2004):

- Domestic fuel burning (69% in 2003)
- Coal-fired boilers (4% in 2003)
- Vehicle emissions (12% in 2003).

The high impact of domestic emissions could be attributed to the fact that these emissions occur in densely populated areas, are low-level emissions and that peak emissions coincide with times of the day when poor dispersion conditions exist (FRIDGE 2004).

In terms of the Highveld specifically, the study identified power generation and the industrial fuel burning, including the Sasol complex, as the most significant sources of SO_2 (FRIDGE 2004). Due to the relatively high stacks of these facilities, long-range transport of pollutants can be expected. The power stations Sasol Secunda complex and Highveld Steel and Vanadium were identified as the main industrial contributors to ambient pollution concentrations. The contribution of vehicle emissions was found to be lower in the Mpumalanga area. The contribution of domestic fuel burning to SO_2 emissions was found to be higher in inland areas where coal is utilized (FRIDGE 2004).

The study found that a relatively small number of all respiratory cases occur within the Mpumalanga region when compared with other areas (8 700 respiratory cases in Mpumalanga compared with 119 000 respiratory cases in the total study area). Power generation was calculated to be responsible for 51% of all these cases and emissions from Sasol Secunda and Highveld Steel and Vanadium contributed 17% and 7% respectively of all cases, whereas 19% of cases were attributable to domestic fuel burning (FRIDGE 2004).

The identified electricity generation intervention that was expected to have the greatest impact on health outcomes was the desulfurisation of power station emissions (FRIDGE 2004). The study assumed a reduction of 94.7% of SO_2 emissions from electricity generation and Sasol Synfuels, which is significantly higher than required by the Minimum Emission Standards. The study found that technology solutions at power stations such as flue gas desulfurisation were not economically or financially feasible, with costs outweighing the benefits (FRIDGE 2004). The study focused on >90% reduction in SO_2 of sulfur compounds and recommended that a marginal cost/marginal benefit approach be used to assess whether partial implementation would not be more feasible. The study did identify the potential for job creation should flue gas desulfurisation be implemented, with a total of 8 458 jobs (FRIDGE 2004).

Only household interventions had positive NPVs due to the high cost of interventions at Eskom/Sasol (FRIDGE 2004). The study found that, although ~1% of all emissions are attributable to domestic fuel burning, these emissions result in 69% of the predicted health costs and therefore have a significantly higher health cost per unit of emissions (FRIDGE 2004). The impact on mortality for Sasol- and Eskom-related interventions was calculated as 1.85 cases per annum for Sasol desulfurisation and 6.37 cases per annum for Eskom desulfurization. It was further calculated that 10.97 cases per annum could be avoided with PM control at power stations (FRIDGE 2004).

The study also conducted a stakeholder analysis and details how impacts on stakeholders (government, firms and households) were calculated per intervention analysed (FRIDGE 2004).

2.4.3 Eskom Study

The purpose of this study was to assess various emission scenarios for Eskom and included baseline as well as growth scenarios (Scorgie & Thomas 2006). The impact on ambient air quality for each scenario was determined using dispersion modelling. The study area was the Mpumalanga Highveld. Eskom power station stack emissions, including current and future emissions of existing power stations, and estimated emissions from return to service (RTS) and proposed future power stations were used in the study (Scorgie & Thomas 2006). Other sources, specifically fugitive dust emissions from Eskom ash dumps and open-cast mining operations, household fuel-burning releases, industrial emissions and vehicle tailpipe emissions were also included. The study noted that the extent and spatial location of atmospheric emissions from certain source types contribute significantly to air pollution concentrations in certain parts, e.g. biomass burning and spontaneous combustion at collieries could not be accurately quantified and were therefore omitted from the simulations (Scorgie & Thomas 2006).

The exposed population was calculated using 2001 census data. The modelled pollutant concentrations were then superimposed over the population density data to determine exposure (Scorgie & Thomas 2006). Baseline mortality data for Gauteng and Mpumalanga was used. Baseline morbidity was calculated using hospitalisation data for Kwa-Zulu Natal, which was adjusted for all provinces. Concentration response functions, recommended by a toxicological study, were used to determine increased mortality and morbidity incidence due to predicted pollution levels (Scorgie & Thomas 2006). The use of vulnerability factors was considered in the study, but was not seen as feasible at the time of the study (Scorgie & Thomas 2006). The study found that hospital admission data was not available for South Africa at the time of the study. A baseline concentration threshold was used in the calculations (in the case of SO_2 - 25ug/m3) and health risks were only calculated for concentrations above this threshold level (Scorgie & Thomas 2006).

The study analysed base-case ambient air quality monitoring data which indicated that although ambient SO_2 concentrations were below the annual average for protection of human health, the annual average for the protection of ecosystems (EC directive 80/779/EEC) were exceeded at most of Eskom's monitoring stations (Scorgie & Thomas 2006). The study predicted that emissions (all sources assessed, including new and RTS stations) accounted for 550 deaths per year and 117 200 respiratory hospital admissions over the study area. Of these, SO_2 emissions were calculated to account for 28% of total non-accidental mortality and 5% of respiratory hospital admissions (Scorgie & Thomas 2006).

Eskom baseline emissions at the time of study (2006) were responsible for 17 non-accidental mortalities and 661 respiratory hospital admissions (3% and 0.6% of totals, respectively). When comparing mortality impacts from SO_2, NOx and PM, SO_2 emissions accounted for 100% of the mortality impacts. The study further found that future power stations and expansions significantly increased mortality risk (617 mortality cases compared to 17). The higher impact is due to larger emission load due to the size of the proposed facilities (Scorgie & Thomas 2006).

2.4.4 Kusile Study

A study by Blignaut (2012) attempted to calculate the damage costs associated with a new coal-fired power station on the Mpumalanga Highveld. A bottom-up approach, or impact pathway analysis, was used in the study. The study included an evaluation of the impacts of PM_{10}, SO_2, NO_x and CO_2. Where local data was not available, a benefit transfer cost method was applied. The study area comprised a zone of maximum ground level concentration, in this case a 25 km radius around the proposed power plant (Blignaut 2012). Dispersion modelling was conducted to obtain the impact of the proposed power station. The exposed population in the vicinity of the plant was estimated using the population density reported in the EIA for the proposed power plant.

Concentration response functions were obtained from a study by Sakulniyomporn *et al.* (2011) which considered data from developing and developed world epidemiological studies. The ERFs used in the study were assumed to be linear with no threshold value (Blignaut 2012) and relative risk (RR) was used to calculate the ERFs, as was done by Sakulniyomporn *et al*. (2011). Due to the absence of local ERFs, an adjustment was made to obtain low, central and high estimates of the risk factors (Blignaut 2012).

The study evaluated the impact of the proposed power plant on water use, due to the increased water demand of a new plant (Blignaut 2012). The study notes that the water tariff is not an appropriate measure of the value of water and the water cost was rather calculated using the net marginal revenue of water (Blignaut 2012). This method entails calculating the additional revenue that could be generated if the water was allocated to a different use. A production function approach was used, with the baseline with Flue Gas Desulfurisation (FGD) compared to baseline with no FGD, conventional water-cooled generation, solar, wind and biomass generation (Blignaut 2012). The difference between baseline and the alternative presents the opportunity cost.

The impact of mining the coal required for feed to the power plant was evaluated using published external cost estimates; adjusted and transferred using the Purchase Power Parity (PPP) and exchange rate. The impact of mining on climate change, water use road accident mortality, water pollution, impact on health and loss of ecosystem goods and services was evaluated and costed using various previous studies (Blignaut 2012). The study notes that the international studies reviewed covered three aspects related to coal mining: climate change impacts related to mining and transportation, human health impacts due to air pollution, and fatalities due to road transportation (Blignaut 2012). The loss of ecosystems was additionally valued at the foregone benefits of agricultural production (Blignaut 2012).

The study calculated the damage cost of the new power stations to be significant. It found that water costs dominated the externality costs (70% of all externality costs evaluated), with mining externality costs having the second highest impact (Blignaut 2012).

2.4.5 Discussion

Many of the larger studies reviewed utilized a bottomed-up approach (Van Horen 1996; FRIDGE 2004; Blignaut 2012; Scorgie & Thomas 2006). All the studies included in the review noted that local concentration-response functions and valuations for health and morbidity outcomes were not available at the time of study and benefit transfer was used in these cases (Van Horen 1996; FRIDGE 2004; Blignaut 2012; Scorgie & Thomas 2006). The Van Horen (1994) study found that morbidity outcomes had a small contribution to overall costs. Van Horen (1994) also adjusted concentration-response functions to reflect uncertainties in the South African context. Blignaut (2012) utilized the same concentration-response functions used by Sakulniyomporn *et al.* (2011) in the Thai study. While some studies used a threshold pollutant concentration below which no adverse effects were assumed (Scorgie & Thomas 2006), most did not (Van Horen 1996; Blignaut 2012). In order to calculate the cost of morbidity outcomes, a variety of methods were used. The FRIDGE study used medical aid data in its calculations. Van Horen (1994) used the sum of time away from work multiplied by wage rate, medical costs (from a variety of sources) and COIDA compensation. Van Horen (1994) found that, due to the bottom-heavy age distribution of the South African population, mortality outcomes were higher in the <65 years age group.

Most studies indicated that the water tariff was not an appropriate measure in calculating the cost of water (Van Horen 1996; Blignaut 2012). Van Horen (1994) used the cost of the Lesotho Highlands Phase 1 projects (LHP1) to value water and Blignaut (2012) used the marginal revenue method whereby the cost of water was calculated as diverted from other uses.

Vegetation and material damage costs were not included in the Van Horen (1994) study due to a lack of information to evaluate the economic impact. These impacts were also not included in the FRIDGE study as concentration-response functions were either not available or were not applicable to local conditions (FRIDGE 2004).

The studies that were evaluated all used a bottoms-up approach to account for location-specific factors such as meteorology, population location and size and differences in source characteristics (FRIDGE 2004; Scorgie & Thomas 2006; Blignaut 2012).

3 PART B

3.1 METHODOLOGY FOR CASE STUDY

A simple financial cost-benefit analysis will be conducted to assess the desirability of SO_2 reduction on the Highveld of South Africa. A bottoms-up approach will be used as this approach takes local conditions into consideration. This is particularly important on the Highveld due to meteorological conditions (refer to Section 3.2.3) and current ambient air quality (refer to Section 3.2.2). The bottoms-up methodology is also the preferred methodology in the local and international studies discussed (refer to sections 2.3.9 and 2.4.5).

The impacts of SO_2 reduction will then be identified (an approach that is similar to Van Horen (1996)) and the Empire State Study (Rowe *et al*. 1995) will be used whereby impacts will be classified into categories according to availability of information and expected impact.

The impact of SO_2 reduction on ambient air will be evaluated using dispersion modelling to determine the changes in concentrations at defined receptor points, including communities, sensitive populations and sensitive ecosystems. Concentration-response functions will be used to determine health impacts on communities.

Once the impacts are identified and the extent of the impacts evaluated, the impacts will be valued. The most applicable method to value the impact will be used (refer to Section 2.2.3). Where local data of sufficient quality is available, this data will have preference. Benefit transfer will be used for data that is not available locally and it will be adjusted using the purchase power parity, GDP, income or particular sensitivity, as deemed appropriate. The valued impacts will be used to calculate a Net Present Value (NPV).

Sensitivity analysis will be conducted on variables that are seen to have a large impact on the outcome or have significant uncertainty attached. The most probable outcome will be used as the central estimate, with high and low values as alternatives for the purpose of the sensitivity analysis.

The process flow methodology is shown in Figure 7.

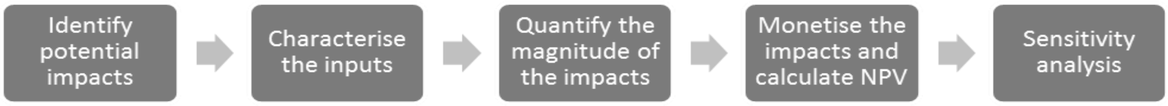

Figure 7: Study methodology

3.2 SITUATION ANALYSIS

3.2.1 South African Legal Framework

a) Air Quality Act (AQA)

The Air Quality Act (National Environmental Management: Air Quality Act, 2004) governs air quality management in South Africa. The AQA aims to govern air quality in a holistic manner and moved the focus of regulation towards the improvement of ambient air quality, rather than source control as under the previous Atmospheric Pollution Prevention Act. To achieve this objective, the Air Quality Act provides for several instruments to improve air quality such as minimum emission standards (MES) providing for the licensing of facilities, and the declaration of priority areas in non-attainment areas.

Section 7 of the Act provides for the establishment of a national framework for air quality management to achieve the objectives of the AQA. The national framework is binding on all organs of state and any organ of state must give effect to the national framework when performing a duty.

Section 9 provides for the establishment of National Ambient Air Quality Standards (NAAQS) that are applicable at a national level. Sections 10 and 11 make provision for stricter standards at a provincial and local level.

Section 18 makes provision for the establishment of priority areas where air quality exceeds the national ambient air quality standards or air quality is negatively impacted. On declaration of a priority area, a priority area air quality management plan must be developed and implemented.

Section 21 provides for the listing of activities and associated minimum emission standards for significant emitters. These standards automatically apply to the Section 21-listed activities. Stricter standards may be applied by the licensing authority where local conditions are so required.

The Act further makes provision for the listing of controlled emitters, controlled fuels as well as several other measures to improve ambient air quality.

b) The National Framework for Air Quality Management in the Republic of South Africa

The purpose of the Framework is to give effect to the objectives of the Air Quality Act (AQA) by providing a medium- to long-term plan for its implementation. The Framework sets out mechanisms and procedures for air quality management, national norms and standards related to air quality, as well as any other matter that is necessary for achieving the objectives of the AQA. The roles and responsibilities of the three tiers of government and obligation on industry are contained in the Framework.

The Framework notes that, when listing an activity, the risk of economic impacts should be minimized by targeting industries where the benefit is expected to outweigh the cost, and that the listing of activities must be informed by analysis such as a cost-benefit analysis. The Framework further notes that the listing of activities and associated emission standards should not be done in a manner that leads to mass non-compliance or unjustified economic impacts. The Framework specifies that the Best Practicable Environmental Option (BPEO) should guide the setting of emission standards and that practicability should be informed by a cost-benefit analysis.

c) National Ambient Air Quality Standards (DEA Government Notice 1210 of 2009)

The National Ambient Air Quality Standards (NAAQS) were published in 2009 (National Environmental Management: Air Quality Act, National Ambient Air Quality Standards, 2009). The NAAQS were initially published for seven pollutants, with limit values guided by the WHO guidelines at the time of promulgation, as shown in Table 1.

Table 1: South African National Ambient Air Quality Standards

Pollutant	**Averaging period**	**Concentration ($\mu g/m^3$)**	**Permitted frequency of exceedance**	**Compliance date**
SO_2	10 minutes	500	526	Immediate
	1 hour	350	88	Immediate
	24 hours	125	4	Immediate
	1 year	50	0	Immediate
NO_2	1 hour	200	88	Immediate
	1 year	40	0	Immediate
PM_{10}	24 hours	120	4	Immediate – 31 December 2014
	24 hours	75	4	1 January 2015
	1 year	50	0	Immediate – 31 December 2014
	1 year	40	0	1 January 2015
O_3	8 hours	120	11	Immediate
Benzene	1 year	10	0	Immediate – 31 December 2014
		5	0	1 January 2015
Pb	1 year	0.5	0	Immediate
CO	1 hour	30	88	Immediate
	8 hour	10	11	Immediate

Subsequent to the initial promulgation, a NAAQS was promulgated for $PM_{2.5}$ (National Environmental Management: Air Quality Act, National Ambient Air Quality standards for

Particulate Matter with Aerodynamic Diameter Less than 2.5 micron meters) as indicated in Table 2.

Table 2: National Ambient Air Quality Standards for $PM_{2.5}$

Averaging period	Concentration (ug/m^3)	Frequency of exceedances per annum	Compliance date
24 hours	65	4	Until 31 December 2015
24 hours	40	4	1 January 2016 – 31 December
24 hours	25	4	1 January 2030
1 year	25	0	Until 31 December 2015
1 year	20	0	1 January 2016 – 31 December
1 year	15	0	1 January 2030

d) Priority areas

Priority areas are areas where ambient air quality is considered to be problematic or where the NAAQS are exceeded. Subsequent to the promulgation of the AQA, three priority areas have been declared in South Africa by the time of this report:

a) The Vaal Triangle Air-shed Priority Area (VTAPA), which comprised the central and southern parts of the Gauteng Province and northern regions of the Free State Province (declared in April 2006);

b) The Highveld Priority Area (HPA), which comprised the western parts of the Mpumalanga Highveld and eastern parts of the Gauteng Province (declared in November 2007); and

c) The Waterberg/Bojanalo Priority Area (WPA), which is located in the North West and Limpopo Province (declared in June2012).

The study area falls within the HPA. Source apportionment studies and air quality management plans (AQMP) were completed for the VTAPA as well as the HPA. The AQMPs are in the process of implementation for both areas.

e) Minimum Emission Standards

Minimum emission standards (MES) were first promulgated in South Africa in 2010. A subsequent revision was promulgated in 2013 (National Environmental Management: Air Quality Act: List of Activities which result in atmospheric emissions which have or may have a significant detrimental effect on the environment, including health, social conditions, economic conditions, ecological conditions or cultural heritage, 2013). The minimum emission standards list 10 categories of industries, divided into several subcategories, and set out minimum emission standards for each subcategory. The standards limit emissions from point sources and also contain special arrangements for certain categories (for example requirements for tanks containing volatile material). The point sources have emission limits on a concentration basis and each source is regulated separately. An international review was conducted to inform the standards, after which a public participation process as required by the Act was conducted to finalise the standards. No publicly available regulatory impact assessment or cost-benefit analysis was conducted.

The MES requires new plants to operate under the emission limits for new plants from the date of the first promulgation. The MES further requires existing facilities to comply with existing plant standards from 1 April 2015, and with the stricter new plant standards from 1 April 2020. The standards do not make provision for grandfathering, although postponement of the compliance timeframes is possible. Various industries have applied for postponement of the compliance timeframes subsequent to the 2013 promulgation of the standards.

Category 1, Subcategory 1.1 of the MES, contains the standards that are applicable to combustion installations. Standards for solid fuel combustion installations (used primarily for steam raising or electricity generation) with design capacity that is equal or greater than 50 MW heat input per unit, based on lower calorific value of the fuel used, is shown in Table 3.

Table 3: Standards that are applicable to combustion installations

Pollutant	Existing plant standard (mg/Nm^3)	New plant standard (mg/Nm^3)
Particulate matter	100	50
Sulfur dioxide	3500	500
Oxides of nitrogen	1100	750

The MES does distinguish between existing and new facilities, as required by the Framework, but affords existing facilities a relatively limited timeframe of five years to comply with the same standard as a new facility. The new plant standard of 500 mg/Nm^3 will be used to calculate the impact of the regulation.

3.2.2 Current ambient air quality situation on the Highveld

The Highveld Priority (HPA) area was declared in 2007, comprising the Mpumalanga Highveld area and parts of the Gauteng Province. An air quality management plan was drawn up for the priority area and gazetted in 2011 (DEA 2011). The HPA study notes that although most of the HPA has relatively good air quality, hot spots where the ambient air quality standards are exceeded were found in nine regions according to the dispersion modelling conducted for the HPA (DEA 2011). In terms of measured SO_2, exceedances of the daily and hourly values were monitored across the HPA. The permitted number of these excursions (four occurrences per annum) was exceeded at the Witbank and Ermelo stations for daily SO_2 (DEA 2011).

Historical data from the Eskom monitoring stations (during the period 2001 to 2005) indicate exceedances of the hourly SO_2 ambient standard at all of the monitoring stations analysed and exceedances of the daily standard at seven of the nine stations analysed (Scorgie & Thomas 2006). The annual ambient air quality standard was not exceeded, however at six of the stations analysed, the EC annual limit for the protection of ecosystems (20 $\mu g/m^3$) was exceeded. Exceedances were seen to be most significant close to the Kendal power station (Scorgie & Thomas 2006). The 2014 Department of Environmental Affairs State of the Air Report (DEA 2014) reports that the annual standard was not exceeded in 2012 or 2013 at the five monitoring stations (Secunda, Hendrina, Witbank, Ermelo and Middelburg), but was exceeded at the Witbank station in 2011 and the Hendrina station in 2009 (DEA 2014). An analysis of the Sasol and Eskom monitored data for the period 2010 to 2012 indicates that exceedances of the hourly and daily data can be seen, but that the annual average SO_2 values were well within the NAAQS, except for Witbank. An analysis of the ambient data indicates a strong peak of SO_2 measured at the monitoring stations at midday, indicating an industrial signature (Sasol 2014; Eskom 2014).

Data from the Highveld Priority Area monitoring network is published on a monthly basis by the South African Weather Service (SAWS 2015). The report for June 2015 indicates exceedances of the hourly standard at the Ermelo, Hendrina and Secunda stations – seven exceedances in total (from January to June 2015) and significant exceedances at Witbank (43 exceedances for the same period). Daily exceedances have only been reported at the Witbank station, exceeding the number of annual allowable exceedances in June already.
The ambient data indicates that, across the study area, short-term ambient air quality exceedances (daily and hourly average) are more widespread than exceedances of the longer-term standard (annual average) for SO_2.

3.2.3 Meteorology of the Highveld

A full description of the meteorology and reactivity of the Mpumalanga Highveld atmosphere is beyond the scope of this study. A few important factors influencing the choice of methodology and models are however discussed in this section. The Highveld area is well known to exhibit conditions that are unfavourable for atmospheric dispersion (Tyson, Kruger & Louw 1988; Tyson, Garstang & Swap 1996). The area has surface inversions nearly every night and elevated inversion regularly during winter, which inhibits the dispersion of pollutants (Tyson *et al.* 1988). Due to a high-pressure system prevailing over the region, high atmospheric stability, clear skies and low wind speeds are often experienced, particularly during winter (Tyson *et al.* 1988). A study by Tyson *et al.* (1996) found that stable layers are formed in the troposphere, and that they inhibit vertical mixing. These stable layers are persistent and can have residence times of days (Tyson *et al.* 1996). An analysis of wind fields indicates a slowly circulating, anticyclonic recirculation vortex over Southern Africa, particularly in winter. These conditions are associated with highly stable atmospheric conditions, high incoming solar radiation, little cloud cover and low precipitation that inhibits dispersion of pollutants over the study area (Tyson *et al.* 1996). Moist and unstable conditions are more frequently seen in the summer (Tyson *et al.* 1988). Pollution concentration may therefore be elevated at higher elevations, which could lead to risks of acidic deposition (Tyson *et al.* 1988).

An analysis of ambient SO_2 concentrations over the Highveld shows a distinct diurnal pattern, with higher SO_2 concentrations at midday and lower concentrations at night-time (Tyson *et al.* 1998; Sasol 2014). This is indicative of an industrial signature where pollutants released via tall stacks reach ground level during daytime turbulent conditions. At night-time, inversion prevents these pollutants from reaching ground level. The Highveld atmosphere is fairly reactive in converting SO_2 to sulfate aerosols, particularly during summer months (Tyson *et al.* 1988). Analysis of sulfate aerosol concentrations indicated no significant seasonal variation in concentrations, most likely due to increased chemical reactivity during summer, even with increased dispersion (Tyson *et al.* 1988).

A study by Held, Snyman & Scheifinger (1993) analysed atmospheric particulates over a network of sites across the Highveld. The network included ground-level sites as well as a number of elevated sites (for example on a stack or at higher altitude). The study found definite seasonal variations in sulfate concentrations, particularly pronounced at elevated sites, with another study indicating concentrations peaks of up to 70 $\mu g/m^3$ at one elevated site (Held *et al.* 1993; Scheifinger & Held 1997). A further study by the same authors indicated that the diurnal pattern of sulfates was less pronounced than nitrates, but that a maximum can be seen in summer (Scheifinger & Held 1997). The study found that stagnating conditions can persist for extended periods, over several weeks, during which sulfate concentrations above 20 $\mu g/m^3$ were measured (Scheifinger & Held 1997). Die study further highlighted that atmospheric processes such as downmixing and washout impacted significantly on sulfate concentrations and highlighted the importance of transport processes in sulfate chemistry (Scheifinger & Held 1997).

A study by Igbafe (2007) measured long-term in-situ ambient concentrations of sulfate species over the Mpumalanga Highveld at the Eskom Elandsfontein ambient air quality monitoring station. The study found sulfate species to be present at significant levels throughout the year (Igbafe 2007). The chemical transformation of SO_2 to sulfates and the ambient impact of sulfates will therefore also be included in the study.

The stable atmospheric conditions, frequent inversions and the impact of SO_2 on ambient sulfate concentrations will be considered when choosing an appropriate dispersion model.

3.2.4 Identification of sources

The results of the dispersion modelling conducted for the Highveld priority area study (DEA 2011) indicated that more than 97% of the ambient SO_2 in all the hotspots located on the Mpumalanga Highveld could be attributed to the industrial sector, of which the power generation and petrochemical sectors contributed the bulk of the emissions (DEA 2011). The study showed that industrial sources were the largest contributors to emissions in terms of tonnages, as shown in Table 4.

Table 4: Source contributions to the HPA emissions inventory

Source category	SO_2 emissions (t/a)	SO_2 contribution (%)
Power generation	1337521	82
Petrochemical	190172	12
Other industrial	84481	5
Motor vehicles	10059	1

According to the HPA study, power generation and petrochemical operations contribute the majority of SO_2 emission, approximately 94%. This study will therefore focus on these two source categories.

The FRIDGE study also identified the petrochemical industry (comprising the Sasol Secunda facility on the Highveld), the power generation industry (comprising various Eskom power stations) and Highveld Steel and Vanadium as the main industrial contributors of SO_2 (FRIDGE 2004).

Therefore, the Eskom power stations in the study area as well as the Sasol Secunda facility were included in the study. Only facilities that were fully operational during the modelling period were included in the study. The new Kusile station (under construction and FGD ready) as well as the Komati station (in process of being returned to service fully) was therefore not included. The location and emissions from these sources are detailed in Table 5. The source information required for modelling purposes was extracted from the atmospheric impact reports compiled in support of the postponement applications (Sasol 2014; Eskom 2014).

Table 5: Sources included in study

Point Source Name	SO_2 emission load (t/a)	SO_2 emission concentration (mg/Nm^3)[1]
Sasol Stack East	91429	854
Sasol Stack West	81323	854
Arnot Stack 1	38637	695
Arnot Stack 2	38637	695
Duvha Stack 1	68618	686
Duvha Stack 2	68618	686
Hendrina North Stack	56871	973
Hendrina South Stack	56871	973
Kriel Stack 1	56167	655
Kriel Stack 2	56167	655
Majuba Stack 1	87582	739
Majuba Stack 2	87582	739
Matla Stack 1	89082	909
Matla Stack 2	89082	909
Camden Stack 1	21325	853
Camden Stack 2	21325	853
Camden Stack 3	21325	853
Camden Stack 4	21325	853
Grootvlei Stack 1	23929	657
Grootvlei Stack 2	23929	657
Kendal Stack 1	109019	973
Kendal Stack 2	109019	973
Tutuka Stack 1	89216	1043
Tutuka Stack 2	89216	1043

1. estimated using reported stack conditions. Based on annual average emissions, not daily basis as required by MES.

3.3 TECHNOLOGY EVALUATION

3.3.1 Options to reduce SO_2 emissions

Sulfur dioxide emissions can either be reduced from existing facilities by reducing the sulfur in the feed to the boiler or by end-of-pipe abatement measures. A brief discussion on each technology and its applicability is presented below.

a) Fuel choice

The choice of fuel, in particular the sulfur content, will influence SO_2 emissions. SO_2 emissions can be reduced by using a fuel with lower sulfur content, such as low-sulfur oil, low-sulfur coal or natural gas (World Bank 1999). Fuel switching is naturally only possible if a suitable alternative is available at reasonable cost. Both Eskom and Sasol utilise low quality coal that has a relatively low sulfur content (approximately 1%) (Eskom 2014; Sasol 2014). Higher-quality coal is reserved for the export market and is considered to be too expensive as a fuel source (Eskom 2014). Eskom also notes that their boilers were not designed to operate on high-grade coal, as they were specifically designed to utilise low-quality, cheaper coal as feed (Eskom 2014). Low-sulfur oil will have to be imported, as it is not available locally. Eskom is in the process of constructing two additional coal-fired units, indicating that coal remains the feed of choice for the utility. The availability of sufficient quantities of natural gas to replace the use of coal at Sasol and Eskom as a fuel source is also highly doubtful in the short term (PWC 2012).

b) Fuel cleaning

Sulfur compounds can be removed from the feedstock prior to combustion, which would result in a reduction of SO_2 emissions. The World Bank indicates that up to 70% of the sulfur in high-sulfur coal is associated with the pyritic fraction. It is further noted that approximately 50% of the pyritic sulfur can be removed from the coal, resulting in a 20 to 30% reduction in sulfur (World Bank 1999). Organic sulfur, however, cannot be removed using gravity-based beneficiation techniques. Beneficiation will therefore not be able to reduce the sulfur content sufficiently to allow for compliance with the required standards. Sasol indicates that their emissions would only be reduced to an average of 1200 mg/Nm^3, which still exceeds the MES of 500 mg/Nm^3 (Sasol 2014).

It should be noted that beneficiation has an impact on water use and treatment, waste generation and increased coal consumption. Additional water will be required for the beneficiation step, which will then require treatment prior to re-use (World Bank 1999). A coal-containing, high-sulfur waste stream will result from the beneficiation step, which will require safe disposal. Sasol indicates that, for their operation, this will result in an additional waste stream of 440 000 tons per year (Sasol, 2014). In order to compensate for the coal lost in the discard stream, additional coal will have to be mined, which will reduce the life of the operating mines supplying Sasol and Eskom (Sasol 2014; Eskom 2014). It is possible to use circulating fluidized bed technology (CFB) to utilise the discard coal, see point e below (Utt & Giglio 2011).

Eskom does indicate in their postponement documentation that fuel cleaning is currently in place at their Lethabo power station due to the poor quality of the coal (Eskom 2014).

c) Sorbent injection

Sorbent injection involves the injection of an alkali absorbent into the flue gas (World Bank 1999). The injection point can be in the flue gas duct between the pre-heater and the Electrostatic Precipitator (ESP)/Fabric Filter Plant (FFP) or directly into the furnace above the flame (World Bank 1999). Limestone or hydrated lime can be used as sorbent and the optimal range for reaction is between 750 and 1250^{0}C for furnace injection and 150^{0}C for duct injection (EPA 2005). The reaction with hydrated lime occurs at lower temperatures (300°C to 650^{o}C). The sorbent reacts with SO_2 and O_2 to form $CaSO_4$, which is removed along with the fly ash. These techniques have a reduction efficiency of between 30% and 60% (Srivastava 2000), with the World Bank quoting an upper limit of 70% (World Bank 1999). This reduction would not be sufficient to reach the required standard. Eskom indicated that due to the fact that PF boilers are utilised, the temperatures are too high for sorbent injection to be effective (Eskom 2014). Sasol further indicated in their application that their current ESPs are not able to accommodate the additional load and would have to be upgraded due to the increase in PM emissions (Sasol 2014).

d) Flue gas desulfurisation

Flue gas desulfurization (FGD) processes can be either dry or wet. In both cases FGD can be once-through or regenerative.

Regenerative processes produce a concentrated SO_2 stream that can be converted into sulfuric acid or elemental sulfur, depending on the sorbent. Several sorbents can be utilised, such as sodium sulfate, magnesium oxide, sodium carbonate amine or activated carbon (Srivastava 2000). The feasibility of these techniques would depend on the availability and cost of a suitable sorbent as well as a market for the product produced. An analysis by the US-EPA in 2000 indicated that regenerative processes make up a small portion of worldwide FGD installations and that there has been no significant increase in regenerative FGD installations since the 1980s, likely due to high O&M costs and marketability of the product (Srivastava 2000). The study by Orfanoudakis, Vakalis, Krallis, Hatziapostlou & Vlachakis (2005) noted that, based on current trends in the installations of new FGDs, it is unlikely that regenerative FGD would find increased application.

Wet FGD is the most widely installed FGD technique worldwide (Srivastava 2000). Wet FGD utilises a scrubber system in which SO_2 is removed by sorption and reaction in a slurry. Limestone, lime or seawater can be used as the sorbent (EPA, 2000). Sodium and ammonium sorbents can also be used (Orfanoudakis *et al.* 2005). Orfanoudakis *et al.* (2005) indicate that the preferred sorbent is limestone, followed by lime. Most installations are simple, counter-flow spray towers (Srivastava 2000). Scrubbers are normally operated at a pH of between 5 and 6 with a stochiometric ratio of 1.01 to 1.1 moles $CaCO_3$ per mole SO_2 (EPA 2000). Spent sorbent is dewatered and disposed of in a slurry pond (EPA 2000). The FGD spent sorbent stream can be utilised as a saleable gypsum product. The EPA notes that the quantity of gypsum produced as a by-product may necessitate a dedicated plant to make use of the tonnages produced. Alternatively, the feed must be split between several facilities (Srivastava 2000). Should all the power stations as well as Sasol produce a gypsum product, the impact thereof on the local market must be evaluated. Furthermore, the transport costs will have to be taken into consideration (Srivastava 2000). In a review by the EPA it was noted that although efficiencies up to 98% have been achieved, most units are designed for 90% efficiency (Srivastava 2000). The World Bank quotes the efficiency of these units as >90% (World Bank 1999). Energy consumption for wet limestone FGD is typically approximately 2% of the net generating capacity of the unit prior to the addition of abatement equipment (Srivastava 2000).

A seawater process utilises naturally alkaline seawater in the scrubbers (Srivastava 2000). The absorbed SO_2 is oxidised to sulfate, which is a natural ingredient in sea water (Srivastava 2000). The EPA notes that the increase in sulfates is usually within the natural variability of seawater (Srivastava 2000).

Dry FGD is used for low- to medium-sulfur content coal, less often for high-sulfur coal (EPA 2000). Lime is used as the sorbent in this process, but not limestone (Eskom 2014). Lime slurry with nominal solids content is sprayed into the dryers at a lime stochiometric ratio of 0.9 for low-sulfur coals and between 1.3 and 1.5 for high-sulfur coals (Srivastava 2000). The water content of the slurry is controlled to avoid saturation of the flue gas (Srivastava 2000). Water treatment is not required, as the water is completely evaporated in the scrubber. Several modules are required for larger units (typically above 250 MW_e). The efficiency of these units is quoted as between 50% and 70% (Orfanoudakis *et al.* 2005) and between 70% and 90% (World Bank 1999).

In addition to the reduction of SO_2 emissions, FGD processes have the additional benefit of reduced-sulfate secondary pollutants as well as a reduction in mercury emissions (Srivastava 2000).

e) Replacement of existing fleet with compliant boilers

Compliance with the emission standards could also be achieved by replacing the existing pulverized fuel boiler fleet with newer technology such as circulating fluidized bed (CFB) boilers. SO_2 reduction in CFB boilers is achieved by the addition of a sorbent such as limestone to the fluidized bed. The sorbent is injected directly into the fluidized bed where it reacts with the SO_2 (Aziz & Dittus 2011). Depending on the sulfur content of the fuel, additional FGD may not be required (Aziz & Dittus 2011; Utt & Giglio 2011). These boilers have additional benefits such as increased efficiency, lower thermal NOx emissions due to lower operating temperatures and a more even temperature profile, as well as the ability to handle lower-quality fuels and variations in fuel quality (Utt & Giglio 2011). CFB boilers would therefore be technically suitable as replacements for current PF boilers, as well as for the utilization of coal discards created by beneficiation.

3.3.2 Technology most likely to be implemented and associated requirements

Since wet FGD is a widely installed and well-developed technology (Srivastava 2000), it is likely that this will be the technology of choice for power generation facilities. Eskom notes in their postponement application that wet FGD is their technology of choice for SO_2 reduction (Eskom 2014) and impacts related to wet FGD will therefore be used in the study. Eskom further indicates that, due to water availability challenges, only two stations, Medupi and Kusile, will be fitted with FGD technology (Eskom 2014). Eskom is currently constructing two additional power plants (Medupi and Kusile) that are utilizing PF technology; it is therefore not likely that the replacement of the fleet with newer technology will be considered viable in the short term.

Even though the boiler technology at Sasol is similar to Eskom, an important difference is the fact that the boiler plant is situated within an industrial complex (Sasol 2014). The boilers supply steam to the running Sasol facility, unlike Eskom which primarily has energy generation as the purpose. This unique Sasol situation places constraints on boiler downtime, as boilers have to supply the plant on a continuous basis (Sasol 2014). A further constraint is related to space availability around the boilers, which will necessitate the demolition of existing equipment to accommodate new FGD equipment (Sasol 2014). Due to the co-dispersion of other pollutants via the same stack as the boiler off-gas, an additional power usage cost of 100 MW will be required to reheat the flue gasses after FGD (Sasol 2014). Sasol indicates in their postponement application that, due to space constraints and water requirements, semi-dry FGD will be their technology of choice (Sasol 2014). Impacts related to semi-dry FGD will therefore be utilised for the purposes of this study.

3.4 IDENTIFICATION OF EXTERNALITIES

The reduction of SO_2 will require the installation and operation of flue gas desulfurization (FGD) equipment. The inputs associated with FGD include:

- capital cost;
- lime or limestone;
- water; and

- additional electricity usage.

The outputs associated with FGD include:

- treatment of water;
- disposal of waste; and
- greenhouse gas emissions.

The expected benefits associated with the reduction of SO_2 are:

- health benefits (mortality and morbidity);
- impact on ecosystems and water resources due to acid deposition;
- impact on vegetation through respiration;
- reduced corrosion impacts;
- visibility impacts due to reduction in secondary aerosols;
- impact on buildings and monuments due to wet deposition; and
- increased employment due to construction activities and associated skilling of labour.

The externalities described above are further discussed in the sections below.

Externalities related to coal mining and electricity generation are not included in this study, as the study aims to only quantify the impact of reducing SO_2 emissions from these facilities in accordance with the requirements of the MES.

Externalities will be classified as follows:

Class 1 externalities: Sufficient information exists to quantify and assign a cost to the externality.

Class 2 externalities: Insufficient data exists to quantify the externalities, but the externality can potentially be significant.

Class 3 externalities: Sufficient information exists to quantify the externality, but insufficient information exists to assign a cost.

Class 4 externality: The externality is expected to have an insignificant impact on the outcome of the study or the cost is already internalized.

3.5 DISCUSSION OF EXTERNALITIES – COSTS

3.5.1 Direct Capital Cost

The installation of FGD will require direct capital cost. FGD has been widely installed internationally and capital cost benefits based on actual costs are available in literature (World Bank, 1999; Orfanoudakis *et al.* 2005; Cleetus 2012). Eskom provided reviewed cost estimates for the installation of FGD in their postponement report (Eskom 2014), however Sasol did not provide costs. For the purposes of this study the Sasol capital cost will be assumed to be the same per unit of output as Eskom and no provision will be made for costing the increased complexity associated with the retrofitting of this facility, since that additional cost is assumed to be a small fraction of the total cost for Eskom and Sasol combined.

Eskom estimates for the capital cost are shown in the table below. For comparative purposes and to obtain high and low estimates for sensitivity analysis purposes, other sources quoting average capital costs per unit of output have been consulted (World Bank, 1999; Orfanoudakis *et al.* 2005; Cleetus, 2012) as shown in Table 5.**Error! Reference source not found.** **Table 4:** Summary of available capital cost estimated for the installation of FGD (Exchange rate of USD0.086 per ZAR; escalated to 2020 costs)

The impact of the installation of FGD on operational costs will be evaluated per item (lime, water, waste disposal and water treatment) in subsequent sections. From the costs shown above, a central most likely estimate as well as high and low estimates for sensitivity analysis will be determined for use in the study.

Direct capital cost can be calculated with reasonable certainty and will be included in the study as a Class 1 impact.

3.5.2 Water

Eskom receives the bulk of its water from the Vaal River System, with most power stations in the study area reliant on the Vaal River Eastern Subsystem (VRESS) and Grootvlei being supplied from the Vaal Dam (Eskom, 2014). In its report Eskom notes that the Vaal River Eastern Subsystem (VRESS) and the Vaal Dam systems are interlinked, as water can be transferred from one system to another. Eskom estimates that the installation of FGD on all of its units will require an additional 70 million m^3 per annum or an increase of Eskom's total water consumption by an estimated 20% (Eskom 2014).

Sasol receives 80% of its total global water demand from the Vaal River System (Sasol 2014a). According to Sasol's SD report for 2013, this would equate to approximately 120 million m^3 per annum, or 4% of the Vaal River System's supply capacity. Sasol's operations in Secunda estimate that 4 million m^3 of water will be required for FGD operation per annum (Sasol 2014).

Sasol and Eskom are strategic water users, which in water deficit situation would mean that other users will have to reduce consumption to ensure supply to these entities.

The additional water requirements for Sasol and Eskom are summarized in Table **5**.

Table 5: Additional water requirements for the operation of flue gas desulfurization

Source	**Current water use (Mm^3)**	**Additional water use (Wet FGD)**	**Additional water use (semi-dry)**
Majuba	29	8	5
Kendal	4	8	5
Duvha	39	7	5
Matla	45	7	5
Tutuka	39	7	5
Kriel	41	6	4
Arnot	27	9	3
Hendrina	25	9	3
Camden	20	8	2
Grootvlei	9	6	2

Komati	10	11	2
Kusile	4	8	-
Sasol	120	4	-
Total	412	98	41

It is noted in the Eskom report that the infrastructure and costs associated with the transfer of water from the Vaal system to the power stations have not been quantified. Sasol does not mention these infrastructure costs in its report. Due to the lack of information, only costs related to supply to the Vaal River System will be considered in this study.

Since the water for FGD is to be supplied from the Vaal River system, the water supply potential and associated costs of this system will be investigated. Only water quantity and not quality will be assessed.

South Africa is a water-deficit country, with evaporation rates exceeding precipitation rates in most parts (DWAF 2004). An Internal Strategic Perspective (ISP) study was conducted in 2004 to inform the Department’s view (Department of Water Affairs, South Africa) of water resource management (DWAF 2004). Resource modelling indicated that by 2025, interventions would be required to supplement a shortfall of 44 million m^3 of water per annum (DWAF 2004). As a result of this study’s recommendations, the Large Bulk Water Reconciliation Strategy Study for the Vaal River System was undertaken to identify and recommend interventions that will ensure a positive water balance (DWAF 2006). In the short term, resource modelling that was conducted indicated that unless unlawful water use can be eliminated and supply side savings measures are not at least partially effective, a water-deficit situation will ensue, and therefore additional supply infrastructure is likely to be required (DWAF 2006). This was confirmed by a study by Blignaut and Van Heerden (2009) that investigated the potential impact of various macro-economic policy decisions on water use and water availability. The study found that it was not possible to stimulate water-intensive industries without reallocating water resources away from other users (Blignaut, 2009).

A study by the Department of Water Affairs (Basson, 2010) found that acid mine drainage (AMD) processing should be prioritised, as it is the option that would be able to yield water the quickest. The study confirmed that the next project to be implemented after AMD treatment would be the Lesotho Highlands Water Project Phase 2 (LHWP 2) followed by a phased implementation of the Thukela Project (Basson, Combrinck, Scroder & Rossouw 2010). The LHFP Phase 2 project entails the building of water supply infrastructure, including the Polihali dam and a hydropower facility. Funding for the water supply infrastructure will be supplied by South Africa (R7.8 billion) and funding for the hydropower facility will be supplied by Lesotho (R 7.6 billion). The yield of the Polihali dam is estimated to be 458 million m^3 per annum (TCTA 2013). The Thukela Project comprises two dams with transfer infrastructure to augment the Vaal River supply with a transferable yield of 454 million m^3 per annum (TCTA 2013).

The study calculated the royalties payable to the Lesotho government for the LHWP Phase 2 and these royalties were included in the calculation of the marginal cost of water from this project (Basson *et al.* 2010). The study found the cost of water from LHWP Phase 2, including royalties, at 6.14 R/m^3 of raw water (Basson *et al.* 2010). A study to compare the costs between LHWP Phase 2 and Thukela Project calculated the royalties payable as R1.37 R/m^3 of raw water (Basson *et al.* 2010) with a total cost of 4.4 R/m^3 of raw water including royalties (Basson *et al.* 2010) (2010 costs).

Sufficient information therefore exists to calculate the cost of water supply and this externality will be included in the study as a Class 1 impact.

3.5.3 Lime and impacts of limestone mining

The quantity of lime required for the operation of FGD was quoted by Sasol and Eskom in their respective postponement applications. Sasol requires approximately 180 000 tons of lime per annum, while Eskom will require an estimated 5 000 000 tonnes per annum of lime (Sasol 2014; Eskom 2014). Limestone consumption for the production of lime varies with limestone properties, end-product specification, limestone purity, etc, but generally two tons of limestone are required for each ton of lime produced (DME 2005). Additionally, two parts of CO_2 are produced for every part lime produced.

Limestone and dolomite deposits in South Africa that are of economic significance are hosted in five sedimentary units. The largest resource is located in the Northern Cape in a relatively narrow 150 km long belt (DME 2005). Limestone deposits in South Africa are of high quality, but are remotely located at a considerable distance from end users (DME 2005; Douglas 1969). The study by Douglas (1969) indicated that transport costs constitute a major percentage of the cost to the customer. A study by the DME (2005) found that energy and transport cost can make up to 70% of the final product price. The price can vary, depending on the consumer requirements, which dictates the source of the lime or limestone (DME 2005). The Department of Mineral Resources (2010) lists limestone and dolomite producers, with detail on products supplied and specification. Due to the amount of lime required, only suppliers that produce lime or limestone in bulk at a high enough grade can be considered as potential suppliers (>85% $CaCO_3$). Although a detailed market analysis is beyond the scope of this project, initial market screening indicates that mines that should be able to supply the required quantities and qualities are situated far away from the Highveld and that transport impacts can therefore be potentially significant. Impacts related to the supply cost and transpiration cost are seen as a Class 1 externality, as sufficient information exists to quantify these impacts.

In South Africa, surface mining is used to mine limestone (Douglas 1969). Many of the negative environmental impacts from mining are localised with mitigation measures that can be put in place. Where such mitigations are available, this study will assume that the cost of these mitigation measures has been internalized. Blasting noise and blasting dust are the most common and most visible impacts of surface operations (Langer, 2001). However, rock-blasting technology is highly developed and can be properly managed to minimize environmental impacts (Langer 2001), and this was therefore assumed to be internalized costs (Class 4). The impact of surface mining on habitats, biota and watercourses are potentially more significant. Deposits that have caves and caverns can host ecosystems with unique biota that have adapted to the conditions of the habitat. When rocks are moved, cave passages are destroyed or exposed to light, which destroys the ecosystems. Aquatic communities can be disturbed when quarrying intersects active groundwater conduits (Langer 2001). The degree to which groundwater can be impacted depends on whether quarrying is done in the saturated or unsaturated zone. If quarrying is done in the unsaturated zone, impacts will tend to be more localized (Langer 2001). The environmental impacts on groundwater quality and ecosystems appear to be site specific and should ideally be investigated at each site along with mitigation measures that can be put in place. Due to the site-specific nature of these impacts, mining ecosystem impacts and water resource impacts will be classified as Class 2 impacts.

3.5.4 Water treatment

The installation of FGD will require additional water treatment to allow for reuse or release of the water due to the build-up due to recirculation of pollutants in the water. Additional water treatment capacity will therefore be required if the current system does not have spare capacity. Furthermore, the composition of this waste water may be different to the current water streams, for example increased chloride concentration, and may require different treatment. The quantification of the volume of water to be treated as well as a cost estimate for treatment can be calculated, and this is therefore seen as a Class 1 impact.

3.5.5 Waste disposal

Although a saleable gypsum product can theoretically be produced, it is unlikely that a market will exist for all the products from the power stations as well as Sasol. Furthermore, it should be noted that the specifications for saleable gypsum are quite stringent and it is not certain that these specifications will be met (Sasol 2014). A waste disposal system will therefore be required for the FGD by-product.

The additional waste stream was estimated to be 350 000 tons per annum for Sasol (Sasol 2014) and 9 500 000 tons per annum for Eskom (Eskom 2014). Disposal costs for waste streams can be estimated and this impact is therefore classified as a Class 1 impact.

3.5.6 Greenhouse gas emissions

The efficiency of generation units will be reduced with the installation of FGD due to the increased power consumption of the FGD units and auxiliary equipment. This decrease in efficiency will have an impact on the greenhouse gas emission intensity of the unit, meaning that more greenhouse gases will be emitted per unit output. Energy consumption for wet limestone FGD is typically approximately 2% of the net generating capacity of the unit prior to the addition of abatement equipment (Srivastava 2000). CO_2 emissions will also increase due to the chemical reaction between the limestone and the SO_2 ($SO_2 + CaCO_3 \rightarrow CaSO_4 + CO_2$). Sasol indicated an increase of 535 000 tons of CO_2 emissions per annum in their postponement application (Sasol 2014) and Eskom estimated an increase of 4 million tons of CO_2 per annum from the direct reaction, as well as an increased electricity requirement of 2 255 GWh/year for the auxiliary equipment (Eskom 2014).

The amount of additional CO_2 generated can therefore be quantified; however due to the relatively small percentage of the total greenhouse gas emissions and significant uncertainties associated with the monetization of the impacts, this impact is considered a Class 3 impact for the purposes of this study.

3.6 DISCUSSION OF EXTERNALITIES – BENEFITS

3.6.1 Impact on ecosystems and water resources due to acid deposition

Pollutants are deposited on the earth's surface by wet and dry deposition. Wet deposition refers to the removal of pollutants from the atmosphere by water such as rain, snow or impaction of the particles with water droplets, for example in clouds. Dry deposition is the removal of pollutants from the atmosphere by processes such as gravitational sedimentation or impaction diffusion onto surfaces. Acidification can occur due to the deposition of S- and N-based acids, while the deposition of N species leads to eutrophication. Deposition can affect water quality by direct deposition or precipitation onto surface waters as well as through water drainage through affected soils (Tyson *et al.* 1988). Dry deposition can either be measured directly or derived from ambient concentrations using inferential methods (Scorgie & Kornelius 2009).

Acid neutralisation capacity (ANC) is an important indicator for determining the capability of soils to neutralise acid deposition. Hydrogen ions displace other cations from the soils, releasing these cations to surface water (NAPAP 2005). Base cations are also released from mineral weathering. These base cations neutralise acidity, thereby increasing the pH of the soils. The balance of these two processes determines whether more cations are leached from the soil due to acidification than can be replaced by weathering (NAPAP 2005). The weathering rate of the parent material therefore plays an important role in acidification and is therefore a critical input (Olbrich, Skoroszewski & Van Tienhoven 1996). Clay soils are negatively charged in general and have a higher capacity to exchange positively charged cations, whereas sandier soils have a lower capacity to exchange cations, making them more sensitive to acid deposition (Bird 2011).

Critical loads are used to assess the sensitivity of ecosystems to deposition. The EEA report (EEA 2014a) defined critical loads as: "the upper limit of one or more pollutants, deposited to the earth's surface that an ecosystem such as a lake or a forest can tolerate without being damaged in its function or its structure". The critical load is determined by linking a sensitive element, such as growth rate, to a chemical criterion, such as ratio of base cations to aluminium that should not be exceeded. The deposition load that results in the exceedance of this limit is then calculated (EEA 2014a).

The National Acid Precipitation Assessment Programme (NAPAP) in the United States highlights that the effect of deposition occurs along a continuum and that it is therefore not possible to identify a single or set of concentrations that can represent a threshold for negative impacts (NAPAP 2005). Site-specific studies are therefore required. Various studies conducted for the study area indicated potential exceedances of critic loads. In a study conducted to assess the risk from acid deposition on commercial forestry in Mpumalanga, Olbrich *et al.* (1996) found that, based on modelled values for weathering rates of soils, there is a portion of forestry soils that are at risk of deposition ranging from 0.5% of the total area under forest to 50%, depending on the species. Scorgie & Kornelius (2009) modelled acid deposition over the Highveld using the Calpuff modelling system. The study found that large power generation and industrial sources contributed significantly to regional acid deposition (Scorgie & Kornelius 2009). The study found that the central Highveld had the highest deposition rates, exceeding 35 kg/ha/year close to large point sources. The study further showed that wet deposition was found to contribute a higher percentage to total deposition in the vicinity of these tall stacks (Scorgie & Kornelius 2009). Over the Highveld, the model indicated a near linear relationship between trends in SO_2 emissions and trends in sulfur deposition (Scorgie & Kornelius 2009). A study by Josipovic, Annergarn, Kneen, Pienaar & Piketh (2011) calculated total deposition rates for sulfur and nitrogen species. Sensitivity of soils was determined using the cation exchange capacity of the soils and the base saturation. Exceedances were determined by overlaying the deposition calculated over the sensitivity of the soil. The study suggests further fieldwork to test soil acidity and evaluate the effects of acidification in areas where exceedances were modelled, and noted the difficulty in relating critical load exceedances with biological damage resulting from changes due to deposition (Josipovic *et al*. 2011).

Other studies have indicated that it is unlikely that critical loads have been exceeded. In a study by Van Tienhoven, Olbrich, Skoroszewski, Taljaard & Zunckel (1995), the weathering rate of soils was used to determine the critical load of soils, as the weathering rate was assumed to represent the buffering ability of the system. The study found that lower surface water critical loads were calculated for upland areas of the Mpumalanga area, and areas with sandy soils with low levels of base cations (Van Tienhoven *et al.* 1995). The study concluded that there was no immediate concern over acid deposition on the Highveld, but noted that dry deposition values were not included, which could result in exceedances of critical loads if included (Van Tienhoven *et al.* 1995). A study by Bird (2011) reassessed the soil properties at 18 sites and surface water quality at five river sites, based on the results of a previous assessment conducted in 1993 (Bird 2011). The study showed significant acidification over the area with 92% of sites showing decreased pH(H_2O) values. The study found increased acidic as well as basic ion concentrations, which could be due to the co-deposition of basic substances along with acidic deposition (Bird, 2011). The soils therefore have buffering capacity due to basic deposition as well as weathering (Bird 2011; Van Tienhoven *et al.* 1995) with increased cations potentially introduced by fly-ash and fly-ash dump emissions or due to veld fires which occur frequently on the Highveld (Bird 2011). The study concluded that it was unlikely that critical loads have been exceeded (Bird 2011), but that the most sensitive soils (sandy solid with less than 4% clay content) located in higher rainfall areas may be at risk of approaching critical loads (Bird 2011). In terms of surface water quality, assessment of the surface water quality at the five river sites did not indicate any impacts from S and N deposition, and spatial differences in the ionic make-up of the water indicate that land-use and water use prior to discharge influence water quality more greatly (Bird 2011).

Studies in Europe and the United States have shown potentially significant impacts associated with deposition (EEA 2014a; NAPAP 2005). A recent report by the European Environment Agency (EEA 2014a) assessed European ecosystems to analyse the impact of the deposition of N- and S-containing gaseous pollutants on ecosystems. The study found a decline in acidification and eutrophication of European ecosystems from the late 1980s that can be attributed to abatement measures to reduce SO_2 and NO_x emissions, closing of older facilities, as well as a reduction in heavy industries. A resultant increase in grassland biodiversity can be seen as a result (EEA 2014a). A potentially similar impact can be expected in South Africa, as pH of rainwater over the Highveld has been proven to be lower than pH of rain in areas that are relatively free from man-made pollution, with pH levels similar to that of North America and Europe (Tyson *et al.* 1988; Turner, 1993). The National Acid Precipitation Assessment Programme (NAPAP) in the United States as well as the studies in Europe have noted that ecosystem recovery is a long-term process that can take several decades and that biological process can take even longer to recover (EEA 2014a; NAPAP 2005).

Although acid deposition can be quantified, the impact thereof on ecosystems has not been quantitatively proven and a greater level of confidence is required in order to determine whether or not measures to abate SO_2 and NO_x are justified on ecological grounds (Josipovic *et al.* 2011). It is likely that a reduction in deposition would result in increased biodiversity, as has been the case in the United States and Europe (NAPAP 2005; EEA 2014a). It should be noted that, due to the long-term impact of deposition and the high deposition rates in South Africa, this impact is potentially significant and should be included in similar analyses as soon as sufficient information is available. At present, there is not sufficient information available to quantify and assign a monetary value to the impact and this impact is therefore classified as a Class 2 impact.

3.6.2 Impact on vegetation through respiration

A detailed discussion and analysis on the uptake and impact of gaseous pollutants on plants is beyond the scope of this study. Gaseous SO_2 uptake in plants is primarily through the stomata. Mosses and lichens are more susceptible to the negative impacts on SO_2, which is also reflected in the lower critical levels when compared with other plant species. These types of plants do not have cuticles or stomata that can close, and are continuously exposed to pollutants (WHO 2000). Lichens and mosses are metabolically active in moist conditions and as SO_2 is very soluble in water, these plants are often the most sensitive species. Due to their sensitivity, critical loads are often determined based on their sensitivity (WHO 2000). Studies in Europe have shown that lichen species diversity and abundance are correlated with ambient SO_2, with increasing diversity when SO_2 emissions were reduced in the 1980s and 1990s (Colls & Tiwary 2009).

Pollutants have both long-term and short-term impacts on plants, which can be economically significant (Colls & Tiwary 2009). Short exposures to high concentrations of SO_2 can cause visible impacts such as foliar necrosis in plants. However, the WHO notes that various studies (Colls & Tiwary 2009) have indicated that these acute effects are of less importance than long-term effects, which may occur at much lower concentrations. Plants may show symptoms of reduced growth and yield and may only show limited visible symptoms such as chlorosis (WHO 2000). It should be noted that, with most studies, isolating the effect of SO_2 can be complicated by the presence of other pollutants, most notably NO_x and ozone. Most of the studies analysing the effect of SO_2 on pathogens and pests have been conducted under much higher SO_2 concentrations and very few of these studies have linked the impacts to the host plants (WHO 2000).

The WHO notes that concentration-response relationships for various agricultural crops are available, but that these relationships have not been established for many native species. The situation in South Africa is similar, with very little information available for native plant species, as indicated by Josipovic *at al.* (2009). However, the UNECE-CLRTAP values have been derived for broad categories of plants and can be seen as reasonably accurate (WHO 2000).

A study by Josipovic *et al.* (2009) assessed the potential impact of pollutants by establishing a passive sampling network of 37 sites. Data was collected monthly for two years and compared with critical levels for vegetation. The study area included the Mpumalanga Highveld. The study notes that although the prevailing wind direction of the study area is from the northwest, recirculation and stagnation seen on the Highveld necessitate the use of a grid covering a large area. A reference site, Elandsfontein, was identified for the assessment as it is located in the area with highest pollution levels, central to the industrial activities on the Highveld.

South African ambient standards applicable at the time of the study, WHO guidelines for the protection of human health and the UNECE-CLRTAP levels were used for comparative purposes, as summarized in Table 6 (EC Directive 80/779/EEC).

Table 6: Limits used to assess SO_2 exceedances

Reference	Limit (ug/m^3)	Averaging period	Applicability
SA NEMA: AQA and SANS 1929	50	annual mean	Human and animal health
WHO AQ Guidelines	50	annual mean	Human and animal health
UNECE-CLRTAP (EC Directive 80/779/EEC)	30	annual mean and half-year (winter) mean for (semi)-natural and agricultural vegetation	Agricultural Crops
	20		(Semi)-natural vegetation
	20		Forest ecosystems
	10		Lichen

The study found that some levels are exceeded in the central Highveld where major sources are located for SO_2. The sites downwind and located close to the power stations and industrial sources showed elevated SO_2 impact. The pollutant levels were found to be below all the critical levels used in the study in remote areas and areas upwind from industrial activity on the Highveld. The UNECE-CLRTAP Lichen standard of 10 ug/m3 was exceeded at the Elandsfontein, Standerton and Amersfoort sites. The UNECE-CLRTAP standard for forest and (semi)-natural vegetation was exceeded at a single site, Elandsfontein, which is located in the centre of the industrial activities of the Highveld. The Scorgie & Thomas study (2006) modelled exceedances of the EC standard for forest and (semi)-natural vegetation across large portions of the Highveld due to SO_2 emissions from power generation facilities.

Although the concentrations of pollutants can be compared with various standards, the actual impact on ecosystems cannot at this stage be accurately quantified and this impact is therefore classified as a Class 2 impact.

3.6.3 Impact on buildings and monuments and corrosion

The National Acid Precipitation Assessment Program (NAPAP) in the United States aimed at coordinating acid rain research notes that, by knowing the mechanisms involved in the degradation of materials and structures, one can predict the impacts of acid deposition (NAPAP 2005). The NAPAP study notes that acid deposition more often has a negative impact on cultural assets due to changes in their appearance, than a negative impact on operational structures such as bridges and buildings (NAPAP 2005). At the time of the NAPAP study, a monitoring or measuring network did not exist that could assess the impact of acidification on materials and structures (NAPAP 2005).

Atmospheric pollutants are known to negatively impact on metals, with a direct correlation between the protection provided by zinc coatings and the level of atmospheric pollution (Tyson *et al.* 1988). Useful life of these materials has been nearly halved in certain areas and fencing and roofing made of unpainted galvanised steel are at risk of similarly being affected by ambient pollution levels (Tyson *et al.* 1988). The Van Horen study notes that apart from anecdotal evidence suggesting increased corrosion rates, measured rates were low, and quantitative information was not available at the time of the study (Van Horen 1996).

Although the potential negative impact of SO_2 ambient concentrations on structures, monuments and corrosion is recognised, there is insufficient information available to quantify this impact and it is therefore classified as a Class 2 impact.

3.6.4 Visibility impacts due to reduction in secondary aerosols

Secondary aerosols such as sulfates can take up water in humid conditions, leading to particle growth. These particles are in the size range that scatters light and can cause visibility impairment. Tyson et al (1998) found that visible degradation on the Highveld could be caused by sulfate aerosols based on subjective experience, but that data was not available to support this. Nephelometer measurements at the Elandsfontein site indicated good visibility (Tyson *et al.* 1988). The Van Horen study (1996) noted that visibility may be impaired on the Highveld, but that attributing the reduced visibility to a specific cause was not easy. Visibility may be impaired by natural sources such as water vapour and natural dust as well as by anthropogenic emissions (Van Horen 1996). The high degree of uncertainty as well as the non-transferability of the valuations done elsewhere means that visibility impacts are considered to be Class 3 impacts for the purposes of this study.

3.6.5 Impacts on employment due to construction activities and associated skilling of labour

It is recognized that the installation of FGD would require skilled labour, leading to increased employment. Due to the number of facilities that have to be retrofitted during a short period of time, it can be expected that skilling of unskilled labour will also be required. The value of increased employment could be reflected by the wages earned. Wage rates can therefore be seen as both a cost (added to the direct capital cost) or a benefit (benefit due to increased employment). The net impact would therefore be zero and it would not be appropriate to add labour cost to either side of the equation. The impact of upskilling could be reflected by the increased earning potential of the individual. However, since it is not certain which number of individuals would be impacted, a monetary value cannot be assigned to this impact and the impact is classified as a Class three impact.

3.6.6 Health

Epidemiological studies are often used to evaluate the increased risk due to air pollution, as the studies evaluate the impact on humans in real-world conditions (WHO 2000b). Mortality impacts of exposure can be estimated by using time-series studies or cohort studies. Time-series studies measure the proportional increase in daily death rate that is attributable to recent exposure, while cohort studies follow large populations for years to determine the mortality that is attributable to exposure (WHO 2000a). The results from cohort studies are generally preferred due to their more comprehensive assessment of the effects of exposure (WHO 2000a).

The choice of health impacts to be evaluated will be determined by the availability of good quality concentration response functions and the ability to monetise the benefits (Jalaudin *et al.* 2009). Furthermore, where health impacts are clearly overlapping, only one outcome should be included (Jalaudin *et al.* 2009).

3.6.6.1 Assessment procedure

The WHO recommends a procedure to be followed to calculate health impacts that can be attributed to the air pollution in the Environmental Burden of Disease Series (WHO 2004). The procedure uses the ambient PM concentrations as a proxy for health impacts; however a similar procedure can be followed for other pollutants. The procedure differentiates between the impacts of long-term and short-term exposure and cautions that short-term exposure impacts should not be added to long-term impacts to avoid double counting. The mortality impacts due to long-term exposure will already include the impacts due to short-term exposure and the author notes that the estimated effect due to short-term exposure is generally much lower than estimates using long-term exposure. Long-term exposure calculations are preferred to short-term estimates; however short-term estimates should be calculated to provide additional information or serve as comparison (Ostro 2004). Long-term estimates are preferred as they can be related to years of life lost (YLL) due to premature death, whereas short-term estimates are not suitable for this type of calculation Ostro 2004). Calculations for children can be added to the YLL calculation, as significant years are lost, but should not be added to short-term calculations, as these impacts are already included in the total mortality impact (Ostro 2004). The two largest long-term studies (the ACS and Six Cities studies, described later) examined participants that were young to middle-aged adults (<30 years); therefore child mortality should be added (Ostro 2004).

The WHO notes that sufficient evidence exists that relates exposure to cause specific mortality, but does not recommend this approach when insufficient data on baseline cause-specific mortality exists (Ostro 2004). However, in developing countries the mortality due to the impact of air pollution may be overestimated due to the relatively smaller fraction of respiratory death compared with other causes of mortality (Ostro 2004).

To calculate the number of cases that can be attributed to air pollution (E), equation 7 is used:

$$E = AF \times B \times P \qquad \text{Equation 7}$$

with:

E = the expected number of deaths per year due to exposure

AF = attributable fraction of death due to exposure

B = population incidence of death (for example in deaths per person)

P = size of exposed population

AF is given by equation 8:

$$AF = \frac{p(RR-1)}{1+p(RR-1)} \qquad \text{Equation 8}$$

with p the proportion of the population exposed and RR the relative risk for the health outcome due to exposure (WHO 2000b). If the whole population is exposed, p=1. For more than one exposure category, for example age groups, the fraction for each category can be summed. For p = 1, equation 9 applies:

$$AF = \frac{RR-1}{RR} \qquad \text{Equation 9}$$

With:

RR = the relative risk of death due to exposure given by equation 10:

$$RR = e^{\Delta deaths \Delta modelled\ change\ in\ concentration} \qquad \text{Equation 10}$$

For short-term impacts, a linear exposure function can be used, as shown in equation 11:

$$RR = \frac{e^{\alpha+\beta x}}{e^{\alpha+\beta x_0}} \qquad \text{Equation 11}$$

which simplifies to equation 12:

$$RR = e^{\beta(x-x_0)} \qquad \text{Equation 12}$$

For long-term exposure, a log-linear equation is recommended, as shown by equation 13:

$$RR = \frac{e^{\alpha+\beta ln(x+1)}}{e^{\alpha+\beta ln(x_0+1)}} \qquad \text{Equation 13}$$

which simplifies to equation 14:

$$RR = \left[\frac{x+1}{x_0+1}\right]^{\beta} \qquad \text{Equation 14}$$

Due to the uncertainties associated with these estimates, a range of impacts is normally provided. A probability distribution of impacts is estimated in more complex analyses (WHO 2000b). Upper and lower confidence levels can be used to calculate intervals for the relative risk (Ostro 2004).

Once the number of cases has been estimated, a monetary cost can be assigned to the impact (WHO 2000b).

3.6.6.2 Cohort studies

To assess the impact of air pollution on mortality, two long term, multi-city cohort studies were initiated in the United States. The Six Cities study by Dockery, Pope, Xu, Spengler, Ware, Fay, Ferris & Speizer (1993) followed 81 111 participants from six US cities. The American Cancer Society (ACS) study by Pope, Thun, Namboodiri, Diockery, Evans, Speizer & Heath (1995) was a larger study with 552 138 adults using data from 151 monitoring stations. Both studies indicated increased mortality associated with air pollution. The data from both these studies were reanalyzed by the Health Effects Institute using stricter convergence criteria to ensure the validity of the findings (Krewski, Burnett, Goldberg, Hoover, Siemiatycki, Abrahamowski & White 2000). The HEI re-analysis study analysed the results of the study by Pope *et al.* (1995) and Dockerty *et al*. (1993) and concluded that the quality of the data used in the two studies was generally good (Krewski *et al*. 2000). Of particular interest to this study, the HEI study conducted spatial analysis of SO_2 concentrations and found an association between annual average SO_2 concentrations and increased mortality risk, even when correcting for sulfates and fine particulates (Krewski *et al.* 2000). The HEI study noted that a reanalysis of the data from the study by Pope *et al*. (1995) indicated evidence of non-linear concentration-response curves for sulfates and fine particulate, with shallow exposure curves below 10 to 15 ug/m^3, and steeper curves for higher exposure (Krewski *et al.* 2000).

Subsequent to the reanalysis by the HEI, Pope, Burnett, Thun, Calle, Krewski, Ito & Thurston (2002) expanded the original ACS study to assess the relationship between particulate and sulfur pollution and mortality, hereafter referred to as the Extended ACS Study. The 2002 study doubled the follow-up time from the original study and expanded on the original dataset (Pope *et al.* 2002). The study indicated that sulfur oxide pollution, measured as SO_2 and/or sulfates, was significantly associated with all-cause mortality (Pope *et al.* 2002). The extended ACS study was again subject to reanalysis by the Health Effects Institute (Krewski *et al.* 2009).

The studies used Cox proportional hazard regression modelling to determine risk ratios, as summarized in Table 7.

Table 7: Concentration response functions derived from cohort studies

Study	Pollutant	Outcome	RR (95% CI)	Study concentrations
Six Cities (Dockery *et al.* 1993)	fine PM	all-cause mortality	1.26 (1.08 - 1.47)	change in 8.8 ug/m^3
ACS (Pope *et al.* 1995)	fine PM	all-cause mortality	1.17 (1.09 - 1.26)	change in 24.5 ug/m^3
		Lung cancer	1.03 (0.80 - 1.33)	
		Cardiopulmonary	1.31 (0.92 - 1.24)	
HEI (Krewski *et al.* 2000)	fine PM	all-cause mortality	1.18 (1.1 - 1.23)	change in 24.5 ug/m^3
		Lung cancer	1.02 (0.8 - 1.30)	
		Cardiopulmonary	1.32 (1.19 - 1.46)	
Extended ACS study (Pope *et al.* 2002)	fine PM	all-cause mortality	1.06 (1.02 - 1.11)	change in 10 ug/m^3
Extended ACS study re-analysis (Krewski *et al.* 2009)	sulfates	all-cause mortality	1.07 (1.05 - 1.09)	change in 5 ug/m^3
		Cardiopulmonary mortality	1.06 (1.03–1.09)	
		Lung cancer	1.04 (0.97–1.11)	
		IHD	1.14 (1.10–1.19)	
	SO_2	all-cause mortality	1.02 (1.02 - 1.03)	change in 14 ug/m^3

3.6.6.3 Time-series studies

Various studies have investigated the association between ambient pollutant levels and mortality and morbidity outcomes. Detailed analyses of these studies have been conducted as part of European and US health reviews. Due to the extent of the data available, this study focused on the meta-analyses conducted in these reviews, with added studies for specific pollutants and health outcomes as required. The US-EPA conducted a thorough analysis of available epidemiological evidence and concentration-response relationships for PM (US EPA 2004). The purpose of the review was, in part, to support decision making regarding appropriate PM NAAQS. For a comprehensive review, the reader is referred to the USEPA review. The report contains information on various single-city studies in addition to the multi-city studies (USEPA 2004). In order to limit city-specific effects, multi-city studies are preferred for benefit transfer (Ostro 2004).

Two large time-series studies aimed at investigating the short-term impact of pollution are the NMMAPS (National Morbidity, Mortality and Air Pollution Study) in the United States and the APHEA 1 and APHEA 2 (Air Pollution and Health: a European Approach) studies.

The NMMAPS study evaluated the impact of air pollution on mortality in the 90 largest cities as well as morbidity and mortality of elderly residents in the 14 cities with daily PM measurements. The NMMAPS data was reanalyzed by the Health Effects Institute as the data was to be included in the review of the PM NAAQS (Dominici, McDermot, Daniels, Zeger & Samet 2003; Schwartz, Zanobetti & Bateson 2003; USEPA 2004). The reanalysis adjusted the increased risk for PM mortality to be significantly lower than in the original analysis. The analysis for pollutants other than PM was then also adjusted (Dominici *et al.* 2003). The analysis of this dataset did not indicate an association between SO_2 and mortality (Dominici *et al.* 2003).

The APHEA 1 study analysed data from 15 European cities (10 Western European cities and 5 Central European cities) to assess the association between mortality from respiratory and cardiovascular outcomes and daily air pollution (Zmirou, Schwartz, Saez, Zanobetti, Wojtiniak, Touloumi, Ponce de Leon, Moullec, Bacharova, Schouten, Ponka & Katsouyanni 1996). The study noted that, although the time studies in the US have not indicated an association between mortality and SO_2, this was not the case for the European studies (Zmirou *et al.* 1996). It was found that SO_2 showed a weak association with cardiovascular and respiratory mortality for the central European cities; however the Western European cities showed a more significant association between mortality and SO_2 (Zmirou *et al.* 1996). The RRs for cause-specific mortality of 1.02 (CI 1.01 - 1.03) were relatively close to all cause estimate of 1.03 (CI 1.02 - 1.04) for an increase of 50 ug/m^3 of SO_2 (Zmirou *et al.* 1996). The APHEA 1 study also found associations between hospital admissions for asthma in children, but not with other respiratory admissions. The data was reanalyzed by Samoli, Schwartz, Analitis, Petsakis, Wojtyniak, Touloumi, Spic, Balducci, Medina, Rossi, Sunyer, Anderson & Katsouyanni (2003) and the results were as follows: the pooled estimate for daily mortality was 1.031, for Western European studies the RR was 1.051 and for Eastern European cities the RR was found to be 1.041.

To confirm the findings and expand the information, the APHEA 2 study expanded on the APHEA 1 study and, as part of APHEA 2, the association between daily SO_2 levels and respiratory hospital admissions was investigated (Sunyer *et al.* 2003). The APHEA 2 study included analyses for 34 cities. A wide range of daily SO_2 values was reported, ranging from 6.8 ug/m^3 to 32.5 ug/m^3. Only daily admissions for asthma in children aged 0 to14 were found to be associated with SO_2, but the association was sensitive to the inclusion of other pollutants (Sunyer *et al.* 2003). SO_2 was not associated with other respiratory admissions in other age groups. The study does note that the results cannot distinguish between SO_2 and other pollutants, but nevertheless suggests that asthma admission in children can be reduced by reducing SO_2 (Sunyer *et al.* 2003). As part of the APHEA 2 study, the association between SO_2 levels and hospital admissions for cardiovascular disease was investigated (Sunyer *et al.* 2003). Two pollutant models were used to determine whether the associations were due to SO_2 exposure or other pollutants such as PM. The association of SO_2 with all cardiovascular admissions became non-significant after adjustment for other pollutants, however IHD admissions for the >5 year age group remained significant, with an adjusted RR of 1.007 (CI 1.001 - 1.013) (Sunyer *et al.* 2003). The results therefore suggest that SO_2 may play an independent role in triggering IHD events (Sunyer *et al.* 2003).

The impacts of air pollutants on health were extensively reviewed by the Quantification of the Effects of Air Pollution on Health in the United Kingdom (COMEAP) and a series of concentration response coefficients was produced (Stedmann, Linehan & King 1999). For SO_2 the concentration response functions recommended by COMEAP are shown in Table 8.

Table 8: Concentration response functions recommended by COMEAP for SO_2

Study	Pollutant	Outcome	RE as % (95% CI)	Concentrations
COMEAP (as cited by Stedmann *et al.* 1999)	SO_2	all-cause mortality	0.6	change in 10 ug/m^3
		Respiratory hospital admissions	0.5	change in 10 ug/m^3

In a subsequent study, COMEAP investigated updated information on the mortality impacts of PM and the impacts of SO_2 and NOx were not included (Ayres 2009). The report noted that the reanalysed work by Pope *et al*. (2002) was the most reliable concentration response functions for PM and should be applied in the UK (Ayres 2009). COMEAP conducted a further study to determine whether sulfates should be analysed separately to $PM_{2.5}$ or whether $PM_{2.5}$ should be adjusted to sulfate. The available data on sulfate epidemiology was reviewed and the study concluded that sulfates are independently associated with adverse health impacts (Ayres 2009).

Hoek, Brunekreef & Verhoef (2000) investigated the association between air pollution and daily mortality over a nine-year period in the Netherlands. Concentrations of SO_2 ranged from 1 to 247 ug/m^3 (P50 = 10 and P95 = 29) and SO_4 concentrations from 0.7 to 36.1 (P50 = 3.8 and P95 = 9.6). The study also found that the RRs for rural areas were not substantially different to RRs of cities. The data for this study was reanalysed in 2003 (Hoek 2003). The RR associated with total mortality was found to be 1.033 (CI 1.007 -

1.063) for a 25ug/m^3 change in sulfates and 1.026 (CI 1.016 - 1.036) for a 50 ug/m^3 change in SO_2 (Hoek 2003). Cause-specific mortality is also included in the analysis.

Fung, Luginaah, Gorey& Webster (2005) investigated the impact of air quality on cardiac hospital admissions in a single city in Canada. The study found that SO_2 concentrations were significant for cardiac admissions of the elderly (>65 years), but not for any other age group (Fung, 2005). For current day (two-day and three-day mean SO_2 concentrations) the increased percentage of cardiac admissions under the elderly was found to be 1.026 (CI 1.00 - 1.053), 1.04 (CI 1.006 - 1.076) and 1.056 (CI 0.0156 - 0,099) respectively associated with an increase of 51 ug/m^3 of SO_2 (Fung *et al.* 2005).

A study involving Polish and Czech schoolchildren aged seven to 10 found that SO_2 was associated with increased risk of respiratory symptoms in areas with outdoor-air SO_2 concentrations between 44 and 97 ug/m^3 (Pikhart, Bobak, Gorynski, Wojtyniak, Danova, Celko, Briggs & Elliot 2005). The following ORs were determined for a 50 ug/m^3 increase in SO_2: wheezing 1.32 (CI 1.10 - 1.57); lifetime prevalence of wheezing 1.13 (CI 0.99 - 1.3); and lifetime prevalence of asthma as diagnosed by a doctor 1.39 (CI 1.01 - 1.92) (Pikhart *et al.* 2005).

Another review by Levy, Diez, Dou, Barr & Dominici (2012) examined available epidemiological studies on the toxicity of fine particulate matter constituents, and 42 studies were evaluated. The data from countries were analyzed separately and country-level results were aggregated across geographical regions (Levy *et al.* 2012). Pooled data for all-cause mortality yielded a risk estimate of 2.8% (CI 0.9 - 4.6) per 10 ug/m^3 increase in sulfates (Levy *et al.* 2012).

A study by Kim, Peel, Hannigan, Dutton, Sheppard, Clark & Vedal (2012) investigated the lag structure for hospital admissions using daily $PM_{2.5}$ constituent concentrations (the study found that, in general, the $PM_{2.5}$ constituents showed shorter lags for cardiovascular admissions and longer lags for respiratory admissions, with secondary aerosols such as sulfates showing more delayed lags) (Kim *et al.* 2012). The pollutant concentrations were adjusted for the preceding two to 14 days and the adjusted RRs recalculated as shown in Table 9.

Table 9: RRs calculated for sulfates

Study	Pollutant	Outcome	RE (95% CI)	Concentrations
Kim *et al.* (2012)	sulfates (with lag)	Cardiovascular hospital admissions	1.004 (0.995 - 1.013)	change in 1ug/m^3
Kim *et al.* (2012)	sulfates (with lag)	Respiratory hospital admissions	1.007 (0.985 - 1.029)	change in 1ug/m^3

As part of the REVIHAAP (review of evidence on health aspects of air pollution) project (WHO 2013), the latest available studies on the health impacts of air pollution were reviewed in order to answer questions related to air-quality policies in Europe. The review found various studies indicating increases in respiratory and cardiovascular hospital admissions, with some studies indicating a link between sulfates and cardiovascular mortality (WHO 2013; Atkinson, Mills, Walton & Anderson 2014; Ito, Mathes, Ross, Nadas, Thurston & Matte 2011). The review noted that sulfates are associated with other harmful components resulting from the burning of fossil fuels and that sulfates could be acting as an indicator or could increase the bioavailability of other components such as metals (WHO 2013).

A study by Atkinson *et al.* (2014) reviewed epidemiological time-series studies investigating the association between mortality and hospital admissions and fine particle components. Of the 63 studies identified, 35 studies investigated the impact of secondary inorganic aerosols. A subgroup analysis was conducted for sulfates and the associations were found to be larger in North American studies than in European studies (Atkinson *et al.* 2014). The results of the analysis are resented in Table 10.

Table 11: Health impacts associated with sulfates

Study	Pollutant	Outcome	RE as % (95% CI)	Concentrations
Atkinson *et al.* (2014)	sulfates	all-cause mortality	0.15 (0.06 - 0.25)	change in 1ug/m^3
		cardiovascular mortality	0.21 (-0.01 - 0.44)	change in 1ug/m^3
		respiratory mortality	0.23 (-0.07 - 0.52)	change in 1ug/m^3
		Cardiovascular hospital admissions	0.12 (0.04 - 0.29)	change in 1ug/m^3
		Respiratory hospital admissions	0.14 (-0.07 - 0.35)	change in 1ug/m^3

Sulfates were therefore associated with all cause, cardiovascular and respiratory mortality as well as with cardiovascular and respiratory hospital admissions (Atkinson *et al.* 2014). The study by Atkinson *et al.* (2014) drew mostly on studies conducted in North America and Europe, but notes a recent growth in studies conducted in Asia.

In order to better understand the health impacts of pollution in Asia, the Health Effects Institute initiated the Public Health and Air Pollution (PAPA) programme in 2002 (Anderson, Atkinson, Balbus, Brauer, Chapman, Chowdury, Cohen, Demerjian, Ebi, Favarato, Greenbaum, Mehta, North, O'Keefe, Pandy, Pope, Smith, Speizer, Walsh & Zhang 2010). The study focused on four cites in Asia: Bangkok, Hong Kong, Wuhan and Shanghai. Additional data from studies in India and Vietnam were included later (Anderson *et al.* 2010). No cohort studies have been conducted in Asia and therefore health estimates are often based on the data from the ACS study (Pope *et al.* 2002). There are, however, differences between the US populations in that study and the Asian population. The ambient levels of air pollution in Asia is much higher than in the US (annual average PM>80 ug/m^3, SO_2 of 20 – 40 ug/m^3; NO_x 40 - 50 ug/m^3) and the proportional concentrations of these pollutants are also different (Anderson *et al.* 2010). Furthermore, baseline health status and energy use differ between the Asian and US populations

(Anderson *et al.* 2010). The impact of socio-economic status of the populations could also affect vulnerability of populations to air pollution (Anderson *et al.* 2010). Similar to the South African population, a large portion of the Asian population is exposed to high levels of indoor pollution, which may affect the vulnerability to outdoor air pollution (Anderson *et al.* 2010). The time series studies and results from PAPA can be used to assess the impact of short-term exposure, but cannot be used to determine impacts on life expectance, and cohort studies should be used for this purpose (Anderson *et al.* 2010; Pope *et al.* 2002). The PAPA study further noted limitations regarding environmental and public health infrastructure, with gaps in knowledge on information such as cause of death (Anderson *et al.* 2010).

Table 10. Health impacts associated with SO_2 in Asia (PAPA study)

Study	Pollutant	Outcome	RE as % (95% CI)	Concentrations
Anderson *et al.* (2010)	SO_2	all-cause mortality	0.75 (0.48 - 1.02)	change in 10 ug/m^3
		cardiovascular mortality	1.16 (0.53 – 1.80)	change in 10 ug/m^3
		respiratory mortality	1.12 (0.76 – 1.47)	change in 10 ug/m^3
		Respiratory hospital admissions	0.51 (-0.17 – 1.19)	change in $10ug/m^3$

SO_2 is highly soluble and readily absorbed in the upper respiratory tract (Lipmann 2006). The effects of acute SO_2 exposure were found to be rapid, but generally short-lived after exposure (Lipmann 2006). Asthmatics were found to be particularly sensitive, but a wide range of responses were recorded for non-asthmatics (Lipmann 2006). Lipmann (2006) cautions against the use of low concentrations of SO_2 as an indicator for mortality as some studies have shown an increase in the relative risk associated with SO_2 under declining concentrations of SO_2, while other studies showed a reduction in death rate with lower SO_2 exposure.

3.6.6.4 Infant mortality

Results from mortality studies on infants contain varying results (Curie & Neidell 2004; Lin, Pereira, Nishioka, Concecao, Braga & Saldiva 2004; Woodruff, Grilla & Schoendorf1997; Hajat, Armstrong, Wilkinson, Busby & Dolk 2007). A cohort study by Woodruff *et al.* (1997) in the United States indicated increased mortality risk for infants exposed to PM_{10}, but did not examine the impact of SO_2 separately. A study conducted in the United Kingdom found an increased risk associated with SO_2 with an RR of 1.02 (95% CI 1.01 to 1.04) for a 10 µg/m^3 increase, but no further associations between air-pollutant levels and infant mortality. A study by Curie and Neidell (2004) indicated an association between infant mortality and CO and PM_{10} concentrations, but due to low ambient concentrations in the study area, SO_2 was not included in the study, although the report refers to several other studies and outcomes related to SO_2. A study by Lin *et al.* (2004) examined the association of infant mortality with PM_{10} and SO_2 and found that both pollutants had increased associated risks (6% (95% CI 4-8) increased risk for an increase in 9.2 µg/m^3 of ambient SO_2)). Due to the correlated nature of the pollutants, a single index representing both PM_{10} and SO_2 was proposed to evaluate increased mortality risks in infants (Lin *et al.* 2004).

3.6.6.5 South African applications and studies

Epidemiological studies are limited in South Africa, although some studies have investigated the link between PM exposure and respiratory health (Terblanche, Opperman, Nel, Reinach, Tosen & Cadman 1992). A study by Terblanche *et al.* (1992) investigated the health effects of pollution in the Vaal Triangle area, using data obtained from health questionnaires, personal monitoring and ambient monitoring. The pollution monitoring was however only done for PM. The study found a higher incidence of respiratory symptoms than similar studies abroad (Terblanche *et al.* 1992). PM values were found to exceed the US limit applicable at the time of the study (260 ug/$m^{3)}$ (Terblanche *et al.* 1992). The study also indicated that bio-aerosols were a cause for concern as there was only a limited time during a year where the atmosphere was free from bio-aerosols and both pollen and fungal counts were above the limits for allergy sufferers (Terblanche *et al.* 1992). The study by Terblanche *et al.* (1992) indicated a reporting bias in their results whereby individuals who perceived pollution to be a serious concern reported higher incidence of symptoms.

A study by Zwi, Davies, Becklake, Goldman, Reinach & Kallenbach (1990) compared the respiratory health of children of school-going age resident in relatively polluted areas in the Highveld with that of children resident in less-polluted areas. The study found that although more respiratory symptoms were reported by the group resident in the polluted area, their lung function, as assessed by spirometry, was not negatively affected (Zwi *et al.* 1990). While both the Terblanche and the Zwi studies contain interesting results, neither derived concentration-response relationships.

A study by Wichmann (1996) studied available epidemiological studies conducted in South Africa for associations between air pollution and adverse health impacts. The study found that although there was evidence of an association between air pollution and adverse health outcomes, none of the studies derived concentration-response curves for the criteria pollutants (Wichmann 1996). For further information on the available epidemiological studies and the results thereof, the reader is referred to Wichman (1996). The Wichman study concluded that there was a strong case for acknowledging the health risks associated with indoor and outdoor air pollution and a quantitative intervention study was recommended.

In the study by Norman, Cairncross, Witi & Bradshaw (2007), the WHO comparative risk assessment methodology was used to determine the burden of disease that can be attributed to indoor and outdoor pollution. The study by Norman *et al.* (2007) calculated the increased mortality attributed to PM_{10} and $PM_{2.5}$ exposures. The study did not consider gaseous pollutants as these pollutants are strongly correlated to PM and including those effects may overestimate the impact (Norman *et al.* 2007). The study only considered the impact of pollutants on mortality, and morbidity outcomes were not included due to a lack of information on incidence rates (Norman *et al.* 2007). The study calculated population-attributable fractions (PAF) using the relative risk of mortality and the prevalence of exposure. The size of the population was then used to calculate national PAFs and these national PAFs were applied to number of deaths and years of life lost (YLL) (Norman *et al.* 2007). Norman *et al.* (2007) note that there is increased uncertainty in applying morbidity data to developing countries, since both a concentration-response relationship and a baseline incidence rate are required.

In a study by Cairncross, John and Zunckel (2007), current international air quality indices were evaluated in order to propose an air-quality index for South Africa. The UK Air Pollution Index (API) uses a scale from one to 10 with the index value of four assigned to the National Ambient Air Quality standard (Cairncross *et al.*, 2007). The US-EPA AQI defines the scales as 100 reflecting the ambient air-quality standard and 500 corresponding to a 'significant harm level' (Cairncross *et al.* 2007). The UK system uses a 15-minute average SO_2 concentration and the US system a daily value. The API proposed by the Cairncross study calculates the total risk as the sum of the risks attributable to each pollutant. The relative risk factors used were taken from a WHO study (WHO, 2000). The incremental risk values were assumed to be constant, with RR=1 for zero exposure. The relative risk attributed to SO_2 exposure was 1.004 per 10 ug/m^3 increase in concentration (incidence of 100 000) for a daily averaging period (Cairncross *et al.* 2007). A concentration was assigned to each risk level, so that the incremental risk of that pollutant was the same (Cairncross *et al.* 2007).

In a thorough review of available concentration-response functions for use in the Eskom study (Scorgie & Thomas 2006), Infotox Pty (Ltd) (Fourie 2006) recommended appropriate risk factors. A review of international data was undertaken and concentration-response functions recommended for use in the study. The report highlights various sources of uncertainty when evaluating health risks using concentration-response functions. Uncertainty related to the representativity of the population of a study as well as the use of statistical models and the reliability of these models can be introduced into a study (Fourie 2006). The Fourie (2006) study recommended concentration-response function for long-term (annual average) and short-term (daily average) health outcomes.

It was noted that South African populations differ from the study populations from which the concentration-response functions were derived (Fourie 2006). South African populations are largely poorer than populations from developed countries and may as a result have poorer access to medical care and adequate nutrition. Higher prevalence rates of HIV/AIDS and TB may contribute to populations being more susceptible to air pollution than the populations studied. Health impacts may be underestimated as a result of these differences (Fourie 2006). The study recommended the use of vulnerability factors to address potential underestimations, but none were available at the time of the study. The study further noted a potential difference in the demographic distribution of South African populations when compared to the populations in developed countries due to lower life expectancies and higher reproductive rates (Fourie 2006).

The Eskom study (Scorgie & Thomas 2006) focused on total non-accidental mortality as a measure of the long-term health effect, and respiratory hospital admissions as the short-term health impact. Both morbidity and mortality impacts were calculated using short-term exposure concentration-response functions. Mortality was based on modelled annual mean concentrations and hospital admissions on modelled peak daily concentrations. A baseline concentration for SO_2 of 20 ug/m^3 was used. The concentration-response relationships selected are shown in Table 12.

Table 12: Concentration-response functions used in the Eskom study (Scorgie & Thomas 2006)

Health effect	% increased risk per 50 ug/m^3 increased concentration in air	Source
Total non-accidental mortality (SO_2)	10 (based on annual average SO_2 concentrations)	Pope *et al.* (2002)
Respiratory hospital admissions (SO_2)	2.5 (based on daily average SO_2 concentrations)	COMEAP

The FRIDGE study used concentration-response functions from a number of previous studies. The concentration-response functions related to SO_2 exposure are shown in Table 13 below. The function shown below was multiplied by the exposure (population multiplied by pollutant concentration).

Table 13: Concentration response functions used in the FRIDGE study

Health effect	Function	Source
Daily mortality >=65 years	1.01×10^{-8}	Holland & Watkiss (2002)
Daily mortality <65 years	1.38×10^{-9}	Holland & Watkiss (2002)
Respiratory hospital admissions – daily exposure	2.01×10^{-6}	Maddison 1997 as referenced in World Bank Group (1998)

The only South African study found that derived concentration-response functions for air pollutants is a study by Wichmann and Voyi (2012), investigating the association between ambient daily concentrations of PM_{10}, SO_2 and NO_x and respiratory, cardiovascular and cerebrovascular mortality in the city of Cape Town. Data for the period 1 January 2001 to 31 December 2006 was used for the evaluation. Data from three ambient air-quality monitoring stations in the city of Cape Town was used. The study found associations between SO_2 and respiratory, cardiovascular and cerebrovascular mortality of -0.7% (95% CI -0.45%; 3.3%), 3.3% (95% CI 0.6%; 3.5%) and 5.3% (95% CI 0.0%; 3.5%) respectively for a 10 µg/m^3 change in SO_2 concentration (Wichmann & Voyi 2012). The study concluded that there was a significant association between SO_2 and cardiovascular mortality.

3.6.6.6 Benefit transfer of concentration response functions

The WHO notes that transferring mortality risks, particularly from developed to developing countries, can be problematic due to the differences in the causes of mortality of the populations (WHO 2000a). When data for a specific location is not available, it is preferable to use data from multi-site studies or meta-analytic studies, provided the populations are not different regarding their response to exposure (WHO 2000a). The use of studies conducted in a different locale can be justified when there are no studies available for the study area and if the estimates are theoretically justifiable (WHO 2000a). Effect modifiers should ideally be used to adjust data between different populations, however such data is not currently available (WHO 2000a). A better understanding of the mechanism of chronic effects could further justify the transferability of data (WHO 2000a).

When data is used in a study, the following should be taken into consideration:

- Risk factors should be applied within the concentrations range they were determined.
- Sources and pollutant mix should be similar.
- The differences between population exposures, such as time outdoors, etc, should be considered (WHO 2000a).

It is further noted that a health impact assessment should consider the differences in risk for different age groups to account for vulnerable groups such as the elderly (WHO 2000a) and individuals living with AIDS (Fourie 2006).

The WHO cautions against adding estimates of effects of individual pollutants, unless the species are not obviously correlated (WHO 2000a). Summing the impacts of all pollutants may overestimate the real impact of the pollutants. One method that can be used is to limit the study to a single pollutant, which would provide an indication of the ‘minimum’ health impact. Including all pollutants will provide an induction of the ‘worst case’ health impacts (Fourie 2006).

3.6.6.7 Limitations of estimates

Epidemiological studies generally correlate health effects with increased ambient concentrations of at least one pollutant. In many cases, the pollutant under study is correlated with other pollutants and differentiating the health effects that can be attributed to each pollutant separately becomes difficult. The study further notes that gaseous pollutants and PM are confounding variables in epidemiological studies. The results of the study will be dependent on the assumptions made by the researcher and can over- or underestimate the impact of a specific pollutant. A study by Krewski *et al.*(2000) found that statistical significance alone is not an indicator of causality as the pollutant under study may act as a marker for another pollutant. A plausible toxicological mechanism for the health effect must be present.

The reanalysis of the original Six Cities and ACS studies concluded that the results of the reanalysis were not sufficient to identify causal relationships between mortality and exposure to pollutants, but that increased concentrations of pollutants were associated with increased mortality, based on the evidence from the two epidemiological studies analyzed (Krewski *et al.* 2000). However, a causal relationship is more likely when a concentration-response relationship exists, when no other plausible explanation for the effect can be identified or when the same effect has been observed in different settings and using different analytical methods (Krewski *et al.* 2000). Due to co-linearity between pollutants, it is often not possible to determine with precision which pollutant contributes which fraction of an outcome (Krewski *et al.* 2000).

The WHO indicates that the direct expected health benefit may not occur immediately on removal of the hazard and that the number of cases prevented may be less than the number of cases predicted (WHO 2000b). The actual benefit will depend on the specific health outcome, the timeframe and effect of the exposure. The benefit realized will depend on the effect of competing risks, once the one contributing case is removed or reduced (WHO 2000b). The number of cases estimated may therefore not necessarily be the number of cases that are actually prevented (WHO 2000b).

3.6.6.8 Data on baseline mortality and morbidity

The calculation of health benefits using concentration response functions requires the use of baseline incidence data. The quality of the baseline data is an important consideration in deciding the health endpoints to be included in a study (Ostro 2004). Baseline all-cause as well as cause-specific mortality data for South Africa is available through Statistics South Africa (StatsSA 2014) and all-cause mortality for South Africa is also documented by the World Health Organisation (WHO 2014). Baseline morbidity data is contained in the South African Demographic and Health Survey (SADHS 2003) and in a review by the Medical Research Council (Ehrlich & Lithoo 2006). A review of private hospital admissions for the Hospital Association of South Africa provides further information on baseline cause-specific hospital admission rates (Da Costa 2009).

3.6.6.9 Discussion on the evaluation of health impacts

In order to reduce city-specific influences, it is preferable to use multi-city, cohort studies for mortality estimates as cohort studies capture specific information of the study population to limit bias. This is also the approach recommended by the Jalaludin *et al.* (2009) review. These studies are suitable for both mortality calculations and reduced life expectancy (Ostro 2004). The largest of these studies is the ACS study (Pope *et al.* 2002) which included the most participants and ambient stations in the analysis. The study data was reanalysed by the Health Effects Institute (HEI) and found to be of high quality (Krewski *et al.* 2009). The reanalysed data will be used for adult mortality calculation for SO_2 and sulfates. As these studies were conducted in a developed country (United States), a sensitivity analysis will be conducted using concentration response functions calculated by Wichmann and Voyi (2012) for South Africa and data from PAPA (Anderson *et al.* 2010) calculated for Asian countries. To evaluate child mortality, the data from the ACS study is not appropriate, as the study subjects were all adults (Ostro, 2004). Infant mortality will be calculated as a sensitivity as there is insufficient literature linking ambient pollution levels to infant mortality (Jalaludin *et al.* 2009). The concentration-response function derived in the Lin *et al.* (2004) study will be used, as the study was conducted in an area with relatively higher SO_2 ambient concentrations.

In order to determine the morbidity impacts, used COMEAP data for SO_2 respiratory hospital admissions, as data from APHEA 2 only indicated that SO_2 may trigger IHD, but did not show an association with cardiovascular hospital admissions when analysed with PM (Sunyer *et al.* 2003). For morbidity impacts of sulfates, the information contained in Atkinson *et al.* (2014) is a recent and comprehensive review of the available information on fine particulate components and health and will therefore be used. The study by Pikhart *et al.* (2005) will be used to quantify asthma risk.

Due to the way in which the concentration response functions are derived, it would be impractical to include an evaluation of health impacts for averaging periods shorter than one day, even though such impacts can occur. The impacts of the shorter exposures (hourly and sub hourly) will be reported along with the daily impacts in epidemiological studies, but cannot be distinguished from one another as hospital admissions or mortality is reported on a daily basis, not hourly. The health impact study will therefore focus on long term (annual averaging) and short term (daily averaging) impacts.

In order to avoid double counting of air pollution impacts, only the impact of SO_2 and sulfates will be considered. In the review by Jalaludin *et al.* (2009) it was noted that the long-term impact on mortality as well as impacts on respiratory and cardiovascular hospital admissions were key health impacts that were assessed by all the reviewed cost-benefit reports and a similar approach will be followed in this study.

Although the uncertainties and limitations of existing data are acknowledged, the health impacts of SO_2 and sulfates can be adequately quantified using transferred existing concentration-response functions and baseline mortality and morbidity data from South Africa. Costs can be assigned to these impacts. The impact of SO_2 reduction on health is therefore classified as a Class 1 impact.

3.7 SUMMARY OF COSTS AND BENEFITS

Impact	Class 1	Class 2	Class 3	Class 4
Costs				
Direct capital cost	x			
Water	x			
Lime and limestone cost and transport	x			
Limestone mining		x		
Water treatment	x			
Waste disposal	x			
Greenhouse gas emissions			x	
Benefits				
Health benefits (mortality and morbidity)	x			
Impact on ecosystems and water resources due to acid deposition		x		
Impact on vegetation through respiration		x		
Impact on buildings, monuments and corrosion		x		
Visibility impacts due to reduction ins secondary aerosols			x	
Increased employment due to construction activities and associated skilling of labour				x

3.8 CALCULATION OF NPV

The methodology for the monetisation of impacts is described in the section below.

In the evaluation of costs and benefits, 2020 was used as the base year for all calculations. The base year was chosen as the year in which the requirements for existing plants to reach new plant standards comes into effect. In the absence of any indication otherwise, it was assumed that the new plant standard of 500 mg/Nm^3 does not change during the lifetime of the facilities.

The study further assumes that all retrofits could be completed by 2020. This assumption ensures that the maximum benefit intended resulting from the regulations is accounted for.

The timeframe of the study was taken as 30 years, ending 2050, when all the currently operating (2015) existing facilities are planned to reach end of life. It was assumed that the plants would be decommissioned according to schedule and that their lifetime will not be extended. The decommissioning dates were taken from Eskom and Sasol's postponement applications (Eskom 2014; Sasol 2014). Costs and benefits were only calculated for the remaining life of each facility.

The analysis further assumes that water and lime transport and storage infrastructure would be in place by 2020 as this assumption calculates the maximum benefit from the regulations. This assumes that all the required water and limestone inputs are available and can be transported to the facilities.

As most of the power stations are located on the Highveld, the study area is taken as Highveld Priority Area boundary, as described in Section 3.8.2.3 below.

A discount rate of 3%, as applied in various international studies was used (Rowe *et al.* 1995; Muller *et al.* 2009). A discount rate of 3% can be considered appropriate when the impacts can be considered within one generation. The period under study is 30 years, which can be considered within one generation. A sensitivity analysis was conducted using an 8% discount rate (Mullins *et al.* 2007) and a zero discount rate to determine the impact of the discount rate (refer to Section 2.2.4 for further information on the selection of the discount rates). Costs and benefits were escalated to adjust for inflation.

3.8.1 Evaluation of costs

3.8.1.1 Capital cost

The direct capital cost estimates per kW were adjusted to the base year (2020) and compared for consistency. Various sources were consulted to determine the variability of estimated capital costs for FGD installation per kW (World Bank 1999; Orfanoudakis *et al.* 2005; Cleetus *et al.* 2012; Eskom 2014). The mean estimate was calculated using the Eskom estimation of costs of R5950/kW, which is above the mean World Bank estimate of R4664/kW and lower than the EPA estimate of R7393/kW. The Eskom estimate was used for the mean estimate as it is based on local conditions and is comparable to the international literature. The total capital cost was calculated using the kW output of each facility and assumes capital spend at the base year.

The calculated direct capital cost was R187 billion (2020 costs) for all the Eskom stations currently in operation within the study area as well as the Sasol facility.

3.8.1.2 Water cost

The cost of water was calculated based on the expected remaining life years of the facilities. The cost of water supplied by LHWP Phase 2, as calculated by the Department of Water Affairs in 2010, of R6.14/m^3 of raw water (refer to Section 3.5.2) was used to calculate the costs of additional water supply. This data was used as it takes the capital cost and operating costs of providing the additional water into consideration.

The estimated mean cost of providing water at that cost was calculated as R 32 billion (present value 2020) over the lifetime of the facilities (up to 2050).

In terms of treating the waste water from the process, the Kusile Waste Management Licence application indicates that a three-step process with the following steps will be used to treat the waste water: 1) Pre-treatment, 2) Evaporation/Concentration, and 3) Crystallisation. The application notes that the treated water will be suitable for re-use and can therefore act as a substitute for raw water. Since raw water costs have already been included in the calculation, adding the cost of water treatment would double count the cost of water. Data on the amount of raw water that could be substituted by treated water was not available, therefore the higher cost of the re-used water was not included in the study. The calculation of the water cost can be refined by including water treatment costs.

3.8.1.3 Lime costs

The lime price was calculated using the aggregate production and sales values obtained from the Department of Mineral Resources (SAMI 2005). The cost calculated was R176/ton (2020 prices) and excludes any transport and handling costs. The transport cost to the facilities was taken as 70% of the supplied price (DME 2005), increasing the cost of lime to R300/ton.

The total lime requirement quoted by Eskom (Eskom 2014) and Sasol (Sasol 2014) was apportioned per kW to account for the decommissioning of power plants. Lime was calculated according to the remaining life of the facilities. The costs were discounted to 2020 costs.

The cost for lime was calculated as R63 billion over the study period. A sensitivity analysis was not conducted for the lime costs, as the costs calculated are based on actual costs and can be considered conservative, as handling infrastructure and costs on site was not included. The lime cost estimate can be refined by using negotiated supply price of lime and using the true cost of lime supply, including transport and handling infrastructure costs.

3.8.1.4 Waste costs

It was assumed that the waste generated would be disposed of in a similar manner as fly ash currently. It was therefore assumed that the ash-handling system was in place and significant upgrades to the existing systems were not required, due to the relatively small volume of additional waste compared to current fly ash volumes.

To calculate the average cost of disposal on a compliant waste disposal facility, the cost and volume of the proposed Kusile disposal facility (Dhemba 2014) was used to calculate a disposal cost per ton based on the cost of the facility and the capacity. This value was used as it is indicative of the capital costs of providing an on-site facility for the disposal of the waste. No costs for rehabilitation of the waste management facilities or operational costs of the facilities were taken into consideration.

The cost calculated for disposal was R11 billion over the study period. A sensitivity analysis was not conducted for the waste disposal costs, as the costs calculated are based on actual costs of a waste disposal facility and can be considered conservative, as the additional marginal rehabilitation costs due to increased waste volumes were not included.

3.8.2 Evaluation of benefits

The only Class 1 benefit identified was a health impact benefit. In order to evaluate the benefit, dispersion modelling was conducted. The change in predicted ambient SO_2 and sulfate concentrations was used, together with concentration-response functions and baseline incidence data to calculate the expected health benefits as detailed in the sections below.

3.8.2.1 Scenarios

In order to assess the change in ambient air quality resulting from compliance with the minimum emission standards, all power generation facilities located in the Highveld Priority Area as well as the Sasol Secunda facility, were included in the study.

Two scenarios were evaluated for each source:

1. Baseline scenario – the current annual average emissions for each source were used to establish a baseline contribution of each source to ambient SO_2 and sulfate concentrations.

2. Compliance scenario - the estimated contribution of each source under theoretical full compliance with the new plant standard of 500 mg/Nm^3 SO_2 emissions was used as the compliance scenario.

The difference between the ambient impacts of these two scenarios was used to calculate the expected benefits of regulation. Compliance was assumed from 2020 and benefits were calculated for the remaining expected lifetime of each facility.

3.8.2.2 Source inventory

The location and emissions from the sources included in the study are detailed in Section 3.2.4. The source information required for modelling purposes was extracted from the Atmospheric Impact Reports compiled in support of the postponement applications (Eskom, 2014; Sasol, 2014). A complete source inventory is provided in Appendix A.

3.8.2.3 Dispersion modelling methodology

The study area was taken as the Highveld priority area, as this is where the majority of coal-fired power stations are located. The study area location of sources and wards used in the study are shown in Figure 8.

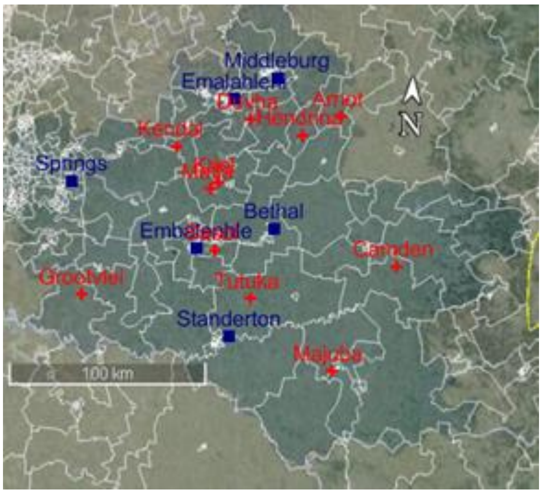

Figure 8: Study area

The South African dispersion modelling regulations (DEA 2014, Government Notice 533) were consulted to guide model selection for the study. Calpuff was selected for the dispersion modelling as the model can handle calm conditions often experienced on the Highveld. The model is further recommended for long-range transport (>50 km) and for multiple sources (National Environmental Management: Air Quality Act: Regulations regarding air dispersion modelling 2014). Furthermore, the model includes chemical transformation. This is important for the study as SO_2 reacts in the atmosphere to form sulfates, which have to be accounted for in the study (refer to Section 3.2.3). A full photochemical model was not used due to data limitations.

The model was run for the period 2010 to 2012 to provide a three-year view of the expected changes in ambient concentrations. A meteorological data file compiled using Calmet was used in the study. The meteorological data was obtained using MM5 (Fifth-

Generation NCAR/Penn State Mesoscale Model) data tiles supplemented by surface station data from three ambient monitoring stations (Bosjesspruit, Sasol Club and Langverwacht stations). The meteorological file extended over the Highveld study area with a grid resolution of 1 km and 10 vertical layers. In the absence of measured upper air data, MM5 data provides a modelled approximation. As the benefit calculation takes the difference in concentrations modelled under baseline conditions and compliance conditions, the impacts of model uncertainty is reduced.

For the Calpuff model a grid resolution of 1 km was used. Chemical transformations were modelled using the Mesopuff II chemical transformation model, included in the Calpuff model. The Mesopuff II scheme takes the impact of relative humidity into consideration, with higher conversion rates from SO_2 to sulfates at higher relative humidity, whereas in the RIVAD scheme, the transformation is linear (Scire, Strimaitis and Yamartino 2000). The RIVAD scheme is recommended for rural modelling as it assumes low background VOC concentrations (Scire et al 2000), which is not a valid assumption over the entire modelling domain (Sasol 2014). Therefore, to ensure that the impact of secondary particulates in the form of sulfates is adequately considered, the Mesopuff II scheme was selected for this model. Ambient ozone and ammonia data from three monitoring stations was used as an input to the chemical transformation model. Wet and dry deposition was included in the model. Due to the height of the stacks, building downwash effects were not taken into consideration.

Additional discreet receptors were included in the simulation model to obtain exposure data at residential areas and monitoring stations. Model output hourly, daily and annual data was extracted from the dispersion modelling results at each receptor.

The modelling results for the annual average SO_2 concentrations for the baseline and compliance scenarios are shown in Figure 9 and Figure 10.

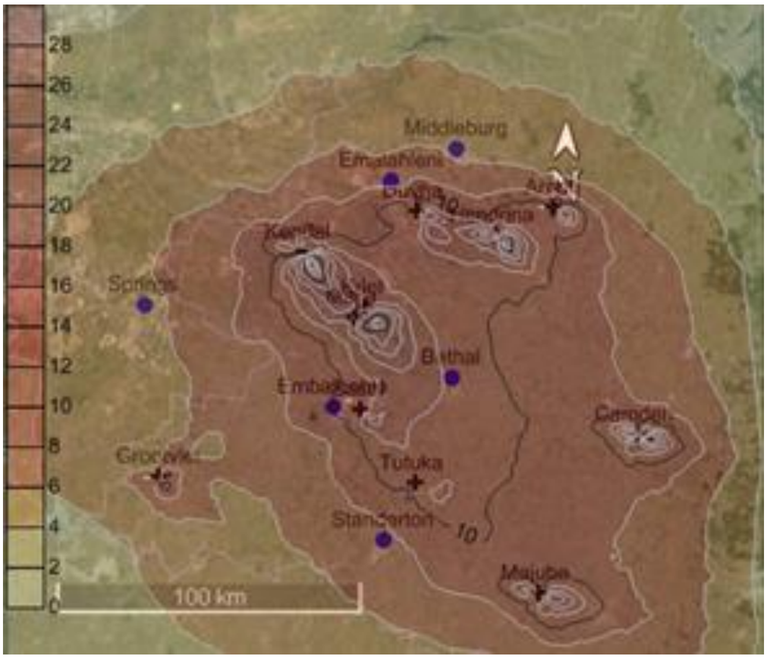

Figure 9: Modelled annual average modelled SO_2 concentration ($\mu g/m^3$) over the study area for baseline conditions

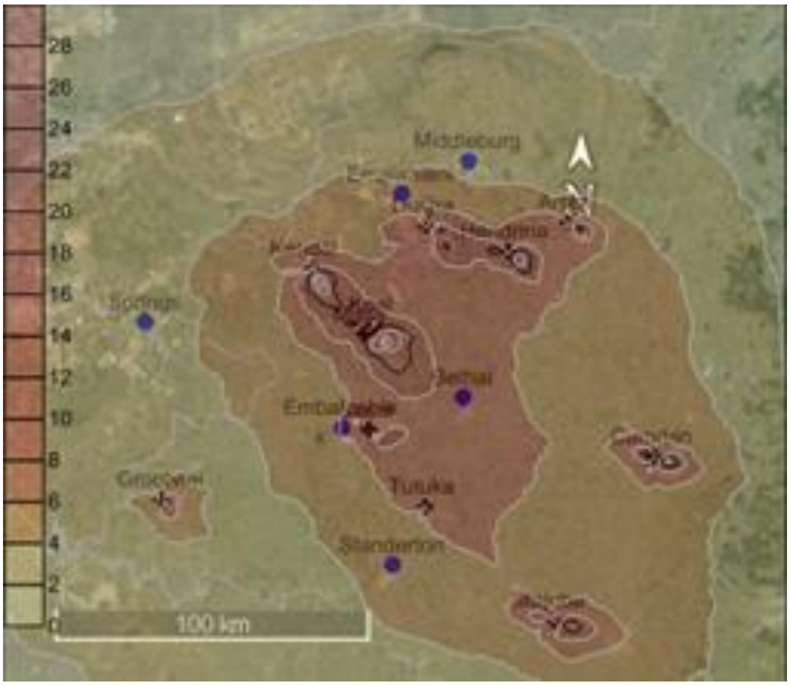

Figure 10. Modelled annual average modelled SO_2 concentration ($\mu g/m^3$) over the study area for compliance with the new plant standards

The dispersion modelling results indicate that the highest impacts of the sources considered occur in populated areas in close proximity to the sources.

3.8.2.4 Health impact assessment methodology

The health impact methodology used is shown in Figure 11.

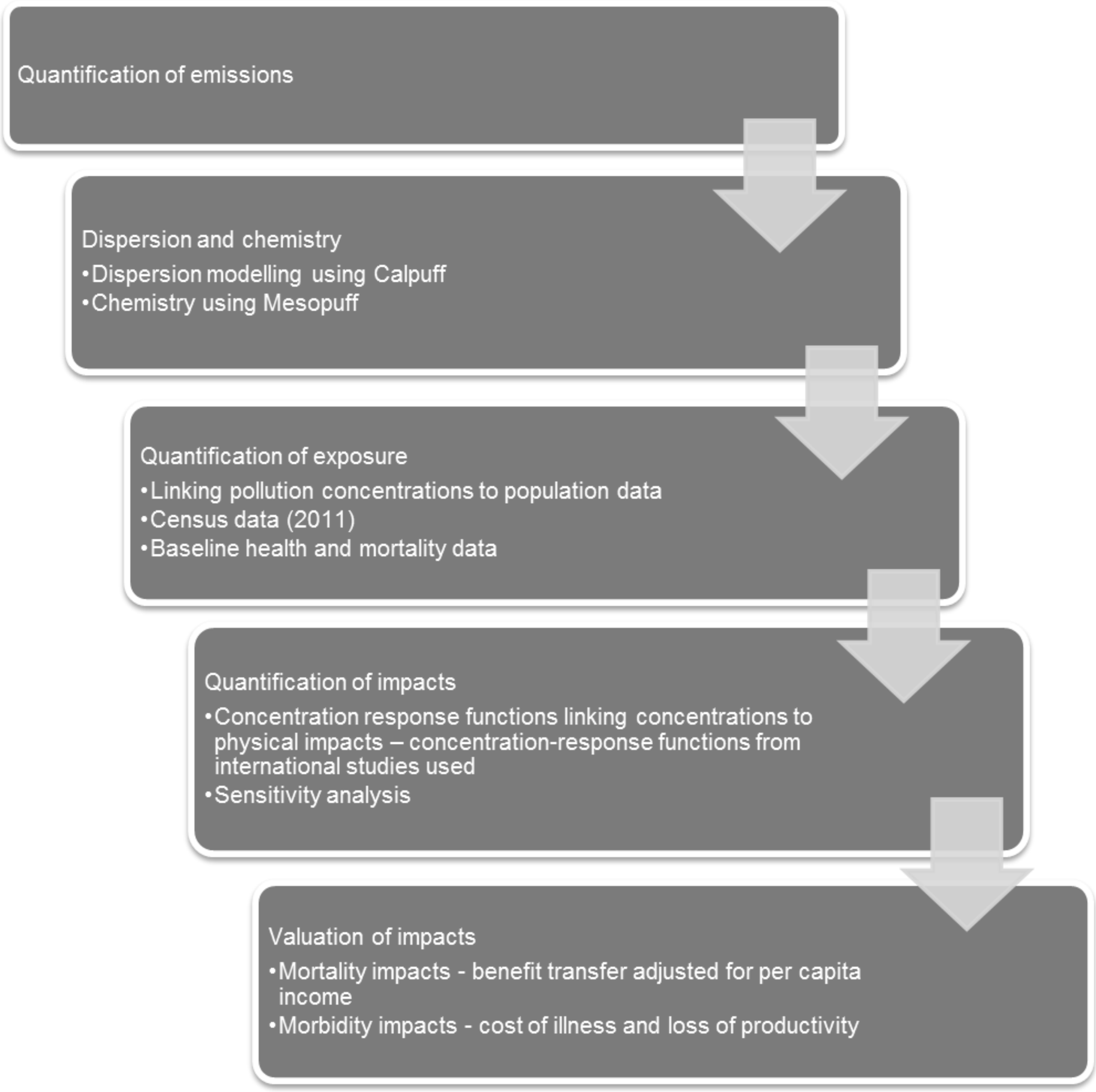

Figure 11: Health impact assessment methodology

The methodology for the health impact assessment was as follows:

- The changes in concentration of SO_2 and sulfates between the baseline and compliance scenarios were extracted from the dispersion modelling results for short-term (daily average) and long-term (annual average) impacts. No threshold values were used for safe levels of exposure. The impact of abatement on other pollutants such as PM and mercury was not included, as separate regulations exist to regulate these pollutants.

- Population data at each receptor point was obtained from the 2010 census data. Population data was aggregated according to age brackets (children under five, youth under 30, adults above 30 years of age and elderly over the age of 55). Census data is provided per ward and the geographical area of the ward is dependent on the population density, amongst others, with more densely populated wards smaller in area than rural wards. Ambient concentrations were extracted at discreet receptors where residential areas are located.

- Population data was overlaid on the dispersion modelling results and health impacts were calculated using concentration response functions. The averaging period used for the calculations was determined by the averaging period on which the concentration response function was calculated. Generally, concentration response functions from cohort studies were applied using the annual average concentrations and concentration response functions from time-series studies were applied using daily average values. For the long term impacts, this is presented graphically in Figure 12.

- Baseline mortality data for South Africa was obtained from publications by Statistics South Africa (StatsSA, 2013) as well as from the World Health Organization (WHO, 2014).

- Baseline morbidity data for South Africa was obtained from the South African Department of Health, Health and Demographic Survey (DoH, 2003) and the Chronic Disease of Lifestyle in South Africa Study (Ehrlich, 2006). Baseline hospital admissions were calculated using data from the HASA Study (Da Costa, 2009)

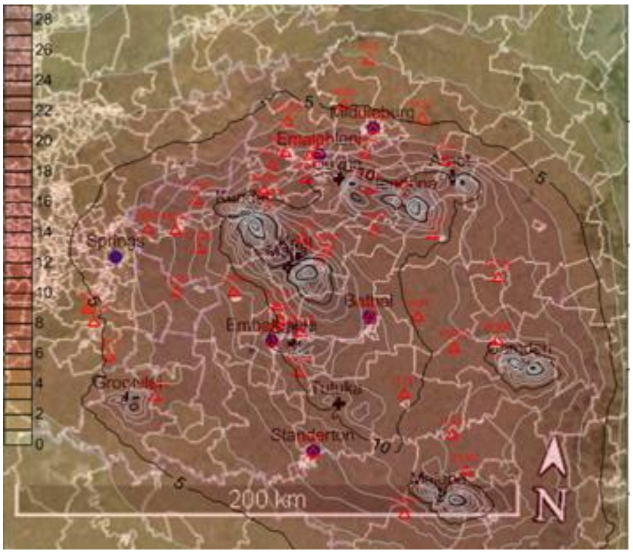

Figure 12. Calculation of long term health impacts: Population data overlaid with concentration data. Population data is shown by red triangles and wards by white borders.

The WHO-recommended methodology (Ostro 2004) was used to calculate the mortality and morbidity impacts as described in Section 3.6.6 of this report. Various studies have shown that the use of threshold values have a significant impact on the outcome (Rowe *et al.* 1995). Therefore, to conservatively estimate health impacts, no thresholds were used (As was the approach by Rowe *et al.* (1995), a linear approximation to non-linear concentration-response values was used.

The concentration response functions for the assessment of mortality impacts were obtained from the reanalysis of the ACS study (Pope *et al.* 2002; Krewski *et al.* 2009) as this is the largest, multi-city cohort study conducted to date. The functions used are shown in Table 14.

Table 14: Concentration response functions for mortality outcomes

Study	Pollutant	Outcome	RR (95% CI)	Study concentrations
Extended ACS study reanalysis (Krewski *et al.* 2009)	sulfates	all-cause mortality	1.07 (1.05 - 1.09)	change in 5 ug/m^3 (annual average)
	SO_2	all-cause mortality	1.02 (1.02 - 1.03)	change in 14 ug/m^3 (annual average)
Lin *et al.* (2004)	SO_2	Infant mortality < 5 years	1.06 (1.04 – 1.08)	change in 9.2 ug/m^3 (daily average)

The concentration response functions used for morbidity impacts are shown in Table 15. The concentration-response functions recommended for morbidity impacts for SO_2 by the Infotox study (Fourie 2006) were used. The results of the reanalysis of the Atkinson *et al.* (2014) Study were used to evaluate the sulfate-related health impacts as this study conducted a thorough review of studies assessing the impact of secondary aerosols. The study by Pikhart *et al.* (2001) was used to evaluate lifetime asthma prevalence.

Table 15: Concentration response functions for morbidity outcomes

Study	**Pollutant**	**Outcome**	**Percentage Change** RR (95% CI)
COMEAP (as cited by Stedman *et al.* 1999)	SO_2	Respiratory hospital admissions	0.05 per change in 1 ug/m^3 (daily average)
Atkinson *et al.* (2014)	sulfates	Respiratory hospitalisations	0.14 (-0.0070-0.35) per change in 1 ug/m^3 (daily average)
Atkinson *et al.* (2014)	sulfates	Cardiovascular hospitalisations	0.12 (0.04 – 0.29) per change in 1 ug/m^3 (daily average)
Pikhart *et al.* (2001)	SO_2	Lifetime asthma prevalence (children aged 7 – 10)	OR 1.39 (1.01 – 1.92) per 50 ug/m^3 change (daily average)

Mortality and morbidity incidence rates used in the calculation are shown in Table 16.

Table 16: Baseline mortality and morbidity rates

Health outcome	Age Group	Baseline	Source
Mortality rate	All	11.6 per 1000	WHO 2014
	all	11.1 per 1000	StatsSA 2014
Cardiovascular mortality	all	16.7% of deaths	StatsSA, 2014
Respiratory mortality	all	10.4% of deaths	StatsSA, 2014
Child mortality	<5 years	7.7% of total mortality. 5.7%<1 year and 2% 1-4 years)	StatsSA, 2014 (2013 data)
Respiratory hospital admissions	All	54.2 per 1000	Da Costa, 2009
Cardiac hospital admissions	All	15 per 1000	Da Costa, 2009
Asthma	All	8.1 (7.1; 9.0) %	Ehrlich, 2006
Asthma	Adults	4.4 % of women and 3.1% of men	SADHS, 2003

The Ehrlich and Jithoo (2006) study highlighted that the most prevalent chronic respiratory diseases in South Africa are chronic obstructive pulmonary diseases (COPD) and asthma. While the causes of asthma are not as well understood, it is widely understood that COPD is primarily caused by tobacco smoking (Ehrlich & Jithoo 2006). The SADHS (2003) study states that self-reported asthma prevalence could be overstated due to confusion with other respiratory health outcomes and that researchers may not be able to distinguish between asthma and COPD in South Africa. Due to the prevalence and likely diagnosis of a respiratory condition like asthma, asthma was used as a morbidity outcome, in addition to respiratory and cardiac hospital admissions. The prevalence rate quoted by Ehrlich and Jithoo (2006) was used in the study.

A study conducted by the Hospital Association of South Africa (HASA) reported on a Private Hospital review in 2009 (Da Costa 2009). Amongst other outcomes, the study contained information on hospitalisation rates and percentages of admissions for various health outcomes. Using the data from this study, a baseline respiratory hospital admission rate of 54.2 admissions per 1 000 people was calculated. Furthermore, the data indicated a cardiovascular hospitalisation rate of 15 admissions per 1 000 people. The data is based on private medical care, but will be utilised as similar data of sufficient accuracy was not available for public medical care.

3.8.2.5 Monetization of benefits

The valuation of the monetary impact of the mortality outcomes was done using benefit transfer. The data used for the calculation is shown in Table 17. The Value of a Statistical Life (VSL) approach was preferred over valuations using Years of Lives Lost (YLL) as the VSL approach is widely used and values for VSL are more readily available than values specifically derived for VOLY. A discussion on the relative merits and applicability of each approach is beyond the scope of this report, but is noted as a further area for study in valuing air-quality impacts in the South African context.

Table 17: Data for VSL calculation

Health outcome	**Baseline**	**Source**
Mortality South Africa	11.6 per 1000	WHO 2014
Mortality US	8.4 per 1000	WHO 2014
GDP per Capita SA (USD)	11970	WHO 2014
GDP per Capita US (USD)	53960	WHO 2014
Per capita expenditure on health SA.	1121	WHO 2014
Per capita expenditure on health US	9146	WHO 2014
VSL (USD)	5.9 (2002 USD)	Viscusi-Aldy Meta-Analysis (Viscusi & Aldy
VSL (R mil)	53 (2020 Rand)	Viscusi-Aldy Meta-Analysis (Viscusi & Aldy

Adjusting the United States VSL as described in Section 2.2.4, a VSL of R53 million (2020) was calculated. The income elasticity was taken as 1 to obtain a high estimate.

The data used for the valuation of the monetary impacts of the morbidity outcomes are shown in Table 18.

Table 18: Data used for the valuation of the monetary impacts of the morbidity outcomes

	Cost (R 2020) costs)	**Source**
Hospital admission (average length of stay 4.37 days)	23127	FRiDGE (Medscheme database 2004)
Asthma treatment cost per annum	1900	FRiDGE (Medscheme database, 2004)
Hospital admission (average length of stay 4.37 days) Specialist care in high care facility	11445	Western Cape Government: Department of Health, 2014
Outpatient care by specialist medical practitioner	276	Western Cape Government: Department of Health, 2014
Income lost per hospital stay	1666	Based on 4.37 days lost income
Income lost per outpatient visit	381	Based on 1 day lost income

The data obtained from the Medscheme information (FRIDGE) was used as a high estimate of healthcare costs and the data obtained from the Western Cape Government: Department of Health (2014) was used as a low estimate.

3.8.2.6 Health impact results

The results indicate that 25 cases of premature mortality per annum could be avoided by the reduction of SO_2 (on all stations) and 32 cases by decreased concentrations of sulfates when all facilities are still operational. The number of cases decreases as facilities reach their expected end of life. Using the VSL value of R53 million, a total benefit of reduced premature mortality was calculated as R114 billion. Three cases per annum of avoided premature child and infant mortality (children <5 years) was calculated. The mortality impacts of children and infants can be valued using a multiplier for the VSL; however given the relatively small number of additional cases calculated, this is unlikely to impact the NPV significantly. Using a multiplier of two due to the larger number of reduced life years, the calculated benefit in reduced infant mortality was calculated as R10 billion.

Additional respiratory hospital admissions due to increased SO_2 concentrations were calculated as 40 additional cases per annum and 47 additional respiratory hospital admissions were calculated per annum due to increased sulfate concentrations. Additional cases of cardiovascular hospital admissions were calculated as 11 additional cases. Due to the high baseline incidence of asthma, 3 000 additional cases of asthma were calculated. Using the data contained in Table 15, a morbidity benefit was calculated as R0.25 billion.

3.8.3 Calculation of NPV – Central Estimates

The base case, most likely estimates for the costs and benefits calculated at 2020 prices with a discount rate of 0% are shown in Table 19.

Table 19: Estimates for costs and benefits without discounting

Impact	**Valuation mean (Rbil)**	
Costs		**% of Total cost**
Direct Capital Cost	187	67
Water	29	10
Lime and limestone mining	55	20
Waste disposal	10	3
Benefits		**% of Total benefit**
Health benefits (adult mortality)	115	91.5
Health benefits (child mortality)	10.44	8.3
Health benefits (morbidity)	0.25	0.2
NPV	**-166**	

The results indicate that the total costs exceed the expected benefit by R166 billion. This indicates that, from a financial point of view, the cost of implementation of new plant SO_2 standards on all emitters exceeds the expected benefit of implementing these standards.

On the cost side, the most significant cost was the capital cost at 67% of the total cost. Operating costs, including water, lime and waste disposal costs, were calculated as R94 billion.

The most significant benefit calculated was the reduction in premature adult mortality of R115 billion, which accounts for 91.5 % of the total benefit.

In order to determine the sensitivity of the outcome to the discount rate applied, a sensitivity analysis was conducted using discount rates of 3% and 8% (refer section 2.2.4). The results of the analysis are shown in Table 20. The use of a discount rate has a limited impact, as the operating costs (R94 billion) and health benefits (R115 billion) are of the same order of magnitude and occur over the same time period (2020 to 2050). The discount rate is expected to have a much more significant impact if costs and benefits occur at different times, for example a large capital expenditure to realise a long-term benefit. The calculation of the benefits are sensitive to the discount rate, with the NPV of the adult mortality benefit decreasing from R115 billion at 0% discount to R77.2 billion at 3 % discount and R46 billion at 8% discount.

Table 20: Impact of using different discount rates

Impact	**Valuation (Rbillion)**			
Discount Rate	**0%**	**3%**	**8%**	
Costs				**8%**
Direct Capital Cost	187	187	187	187
Water	29	20	12	12
Lime and limestone mining	55	38	23	23
Waste disposal	10	7	4	4
Benefits				**3%**
Health benefits (adult mortality)	115	77.2	46.0	77.2
Health benefits (child mortality)	10.44	7.08	4.35	7.1
Health benefits (morbidity)	0.25	0.2	0.1	0.2
NPV	**-166**	**-173**	**-180**	**-148**

The highest NPV is calculated when an 8% discount rate is applied to the costs and a lower discount rate of 3% is applied to the benefits, as indicated in the last column of Figure 14. Applying different discount rates to costs and benefits ensures that more weight is attached to benefits occurring in future and is aligned with the concept of sustainable development.

In order to determine whether partial implementation of FGD would be feasibly, the analysis was repeated on the single station with the highest health cost impact. The analysis indicates that the most significant impact is from the Kendal power station, with approximately 25% of the total mortality benefit attributable to the single station. The results of the analysis indicate that, when using the central estimate for capital cost and the high estimate for mortality benefits, the costs exceed the benefits for all discount rates for the Kendal station as indicated in Table 21. This is due to the high capital cost of FGD implementation. However in this case, the yearly mortality benefit exceeds the operating costs (water, lime and waste disposal costs) for FGD.

Table 21. NPV Calculation for Kendal Station

Impact	**Valuation (Rbillion)**			
Discount Rate	**0%**	**3%**	8%	
Costs				**8%**
Direct Capital Cost	24	24	24	24
Water	5	3	2	2
Lime and limestone mining	11	7	4	4
Waste disposal	2	1	1	1
Benefits				**3%**
Health benefits (adult mortality)	37	24.8	14.2	24.8
Health benefits (child mortality)	1.06	0.71	0.41	0.7
Health benefits (morbidity)	0.02	0.0	0.0	0.0
NPV	**-5**	**-11**	**-17**	**-6**

3.8.4 Sensitivity analysis

Sensitivity analyses were conducted on variables that had a significant impact on the NPV calculation or had significant uncertainties associated with the variables chosen. On the cost side, the capital cost of the abatement equipment was the most significant cost (67 % of total cost). A sensitivity analysis was therefore conducted on the capital cost of the regulation.

The high estimate was calculated using the data from Cleetus et al (2012) and the low estimate using Orfanoudakis et al (2005). The calculated direct capital cost was R187 billion (high R306 billion; low R80 billion) for all the Eskom stations that are currently in operation within the study area, as well as the Sasol facility. The capital cost estimate had a large impact on the NPV with low and high estimates for capital costs 57% lower and 63% higher respectively than the central estimate.

The largest benefit calculated was associated with the reduction of mortality risks (91.5 % of total benefit). Due to the large number of uncertainties associated with the evaluation of mortality benefits, sensitivity analyses were conducted on various inputs into the calculation. The parameters selected for sensitivity analysis were the choice of dose response functions used and the monetisation of the benefit (VSL estimates).

In order to use the data from the extended ACS study as applied to the central estimate, similar structures in mortality patterns and levels of healthcare are implied. Furthermore, mortality impacts may be overestimated due to differences in baseline mortality data in the original study area (United States) and South Africa. To address these uncertainties, a sensitivity analysis was performed using the concentration response data contained in the Anderson *et al.* (2010) review, as the study was conducted in a developing country with higher pollution levels and high levels of indoor air pollution. A further sensitivity analysis was conducted using the South African data from the Wichmann and Voyi (2012) that indicated a significant association between SO_2 and cardiovascular mortality. The concentration response functions used are shown in Table 22.

Table 22: Concentration response functions for sensitivity analysis

Study	Pollutant	Outcome	RR (95% CI)	Study concentrations
Wichmann & Voyi 2012	SO_2	Cardiovascular mortality	1.033 (1.06 - 1.035)	change in 8 ug/m^3 (daily average)
Anderson *et al.* 2010	SO_2	cardiovascular mortality	1.16% (0.53 – 1.80)	change in 10 ug/m^3 (daily average)
	SO_2	respiratory mortality	1.12% (0.76 – 1.47)	change in 10 ug/m^3 (daily average)

In terms of increased premature mortality cases attributable to increased SO_2 concentrations, the impact calculated using the base case functions calculated 25 additional cases, with the Anderson *et al.* (2010) functions calculating 16 additional cases and the Wichmann and Voyi (2012) data calculating 32 additional cases. The associated health benefit associated with SO_2 mortality impacts calculated using the base data was R50 billion, compared to the Asian estimate of R26 billion and the South African estimate of R53 billion for the SO_2 only impact. The choice of concentration-response function could potentially have a large impact on the results and sensitivity analysis is recommended.

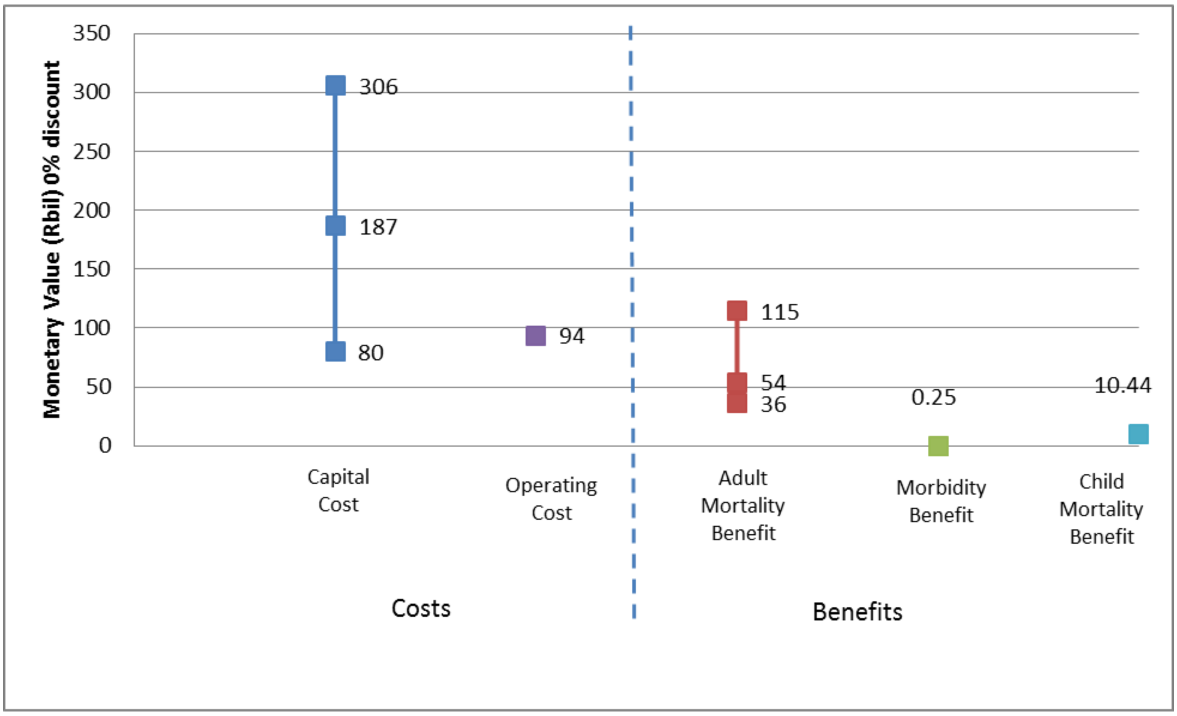

Figure 13. Sensitivity to concentration response functions used

Hammit and Robinson (2011) examined the elasticity that should be used when transferring VSL estimates from high- to low-income countries and found that using an elasticity larger than one may be more appropriate when extrapolating VSL studies to lower-income countries as it is unrealistic to assume that a lower-income individual would be willing to pay such a high proportion of their income for a small reduction in mortality risk. The impact of different VSL estimates using different elasticities is shown in Table 23. In order to contextualize the VSL estimates, the value that an individual is willing to pay (WTP) for a one in 10 000 annual mortality risk change calculated from the VSL value is expressed as a percentage of the median individual wage rate for South Africa. The table indicates that using elasticity below 1 yields unrealistically high VSL estimates when compared with the median wage rate of South Africa. The calculated VSL of R53 million that was used as the high estimate translates into a WTP for a 1 in 10 000 mortality risk reduction of 3.8% of South Africa's GDP per capita. A VSL value of R25 million, calculated using an income elasticity of 1.5, was used as the central estimate.

Table 23: VSL estimates using different income elasticities

Elasticity	**VSL (Rmil)**	**% of SA median wage rate**
1	53	8.7
0.6	97	16
1.5	25	4.1
2	12	1.9

In a study by Miller (1999), VSL estimates from various countries were compared and the study found that the values were typically about 120 times the GDP per capita. Applying this relationship to South Africa yields a VSL estimate of R16.7 million. This value was used as the low estimate for VSL and corresponds to a WTP of 2.7% of the median individual wage rate for South Africa for a one in 10 000 annual mortality risk change. The calculated mortality benefit associated with central, high and low VSL estimates is shown in Table 24.

Table 24: Sensitivity analysis using VSL values

VSL estimate	**R mil**	**Source**	**Mortality benefit (Rbil)**
VSL central estimate (R mil)	25	Viscusi-Aldy Meta-Analysis (Viscusi & Aldy 2002) adjusted with income elasticity of 1.5	54
VSL low estimate (R mil)	16.7	Miller (1999)	36
VSL high estimate (R mil)	53	Viscusi-Aldy Meta-Analysis (Viscusi & Aldy 2002) adjusted with income elasticity of 1	115

The results indicate that the VSL estimate has a significant impact on the NPV, due to the high percentage of total benefit attributable to a reduction in premature mortality impacts.

3.9 DISCUSSION

In conducting an economic analysis, only costs and benefits that can be quantified and monetised can be included in the analysis. To be quantified, sufficient data must exist to relate a change in ambient pollutant concentrations to a specific outcome. This information is often site specific, although in some cases benefit transfer can be used to supplement gaps in local information. Further information is then required to attach a monetary value to a quantified impact. The more information available on costs and impacts, the more reliable an economic analysis will be. Insufficient information could lead to an underestimation, particularly of benefits that are more difficult to quantify such as biodiversity impacts or impacts that are realised over long periods of time. Although such benefits cannot be included in the analysis, the likelihood of realising a benefit can be considered when deciding whether a project should be implemented or not.

Benefit transfer can be used when site specific data is not available, for example concentrations response functions and VSL values derived elsewhere. Understanding the underlying assumptions underpinning the source study and conditions can inform which data to use for benefit transfer, as benefit transfer assumes that conditions are comparable. Using data derived under significantly different conditions could potentially impact on the analysis by either over of underestimating impacts.

The economic analysis necessitates the use of many assumptions, particularly when all the information is not available. These assumptions can materially impact the result of the valuation and it would be beneficial to use sensitivity analysis to obtain a range of likely values. While this does not provide a definitive answer, it does indicate whether a project's benefits are expected to exceed the costs.

A summary of the costs and benefits calculated is shown in Figure 14. The results indicate that the costs (capital cost plus operating cost) exceed the benefits for all scenarios evaluated. The capital cost was calculated as the most significant input on the cost side and a sensitivity analysis was performed to determine the variability of the capital cost, as shown in Figure 14. The direct capital cost was calculated as R187 billion (high R306 billion; low R80 billion). The most significant benefit was calculated as the reduction in premature adult mortality (91.5 % of total benefit, calculated as R115 bil), with morbidity impacts and premature infant mortalities at a lower cost (R0.25 bil and R10.5 bil respectively). The operating cost of the abatement equipment was found to be comparable in magnitude to the expected benefits, even when only water, lime and waste disposal costs are considered. Excluding the operating cost and only utilising the capital expenditure as cost of regulation would have a significant impact on the NPV calculation.

Various factors and uncertainties could potentially impact the valuation, such as the changes in concentrations modelled, the concentration-response function used and health outcomes chosen for evaluation, accurate information on baseline incidence rates and the choice of VSL. The choice of concentration-response function used to calculate mortality outcomes is potentially significant. Concentration-response functions derived from a different population may not be accurate as baseline population health status, pollutant mix exposed to, occupational exposure and numerous other factors. Furthermore, the choice of an appropriate VSL has a significant impact (high R115 billion; low R36 billion) on the mortality valuation as indicated in Figure 14. This uncertainty can be reduced by conducting sensitivity analysis to calculate a range of plausible outcomes.

The sensitivity analysis using different discount rates (0%, 3% and 8%) had a significant impact on the health impact valuations. However, as operating costs and health benefits were similar in magnitude and occurred during the same time periods for this particular study, the overall NPV was not significantly affected by the choice of discount rate. The discount rate is expected to have a significant impact on the outcome of the analysis when capital spend is expected to realise benefits into the future and care should be taken in using an appropriate discount rate.

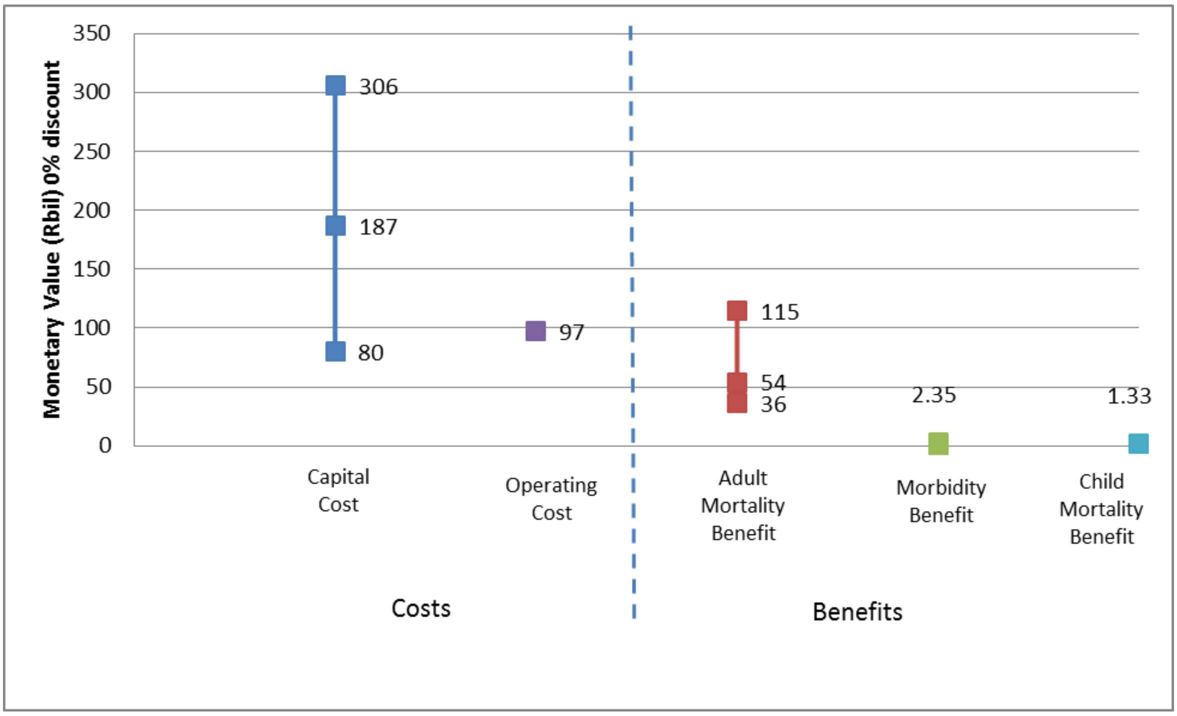

Figure 14: Summary of range of costs and benefits calculated – All Stations

For the installation of FGD at the station with the highest impact, Kendal, the mortality benefit exceeds the operating costs, due to the high impact of this station due to its proximity to residential areas as indicated in Figure 15. The NPV for the installation of FGD on this station is still negative (-R5 billion to –R17 billion), however the sum of the health benefits are higher in relation to the capital cost, comparable to the capital costs for the high estimate without discounting the health benefit.

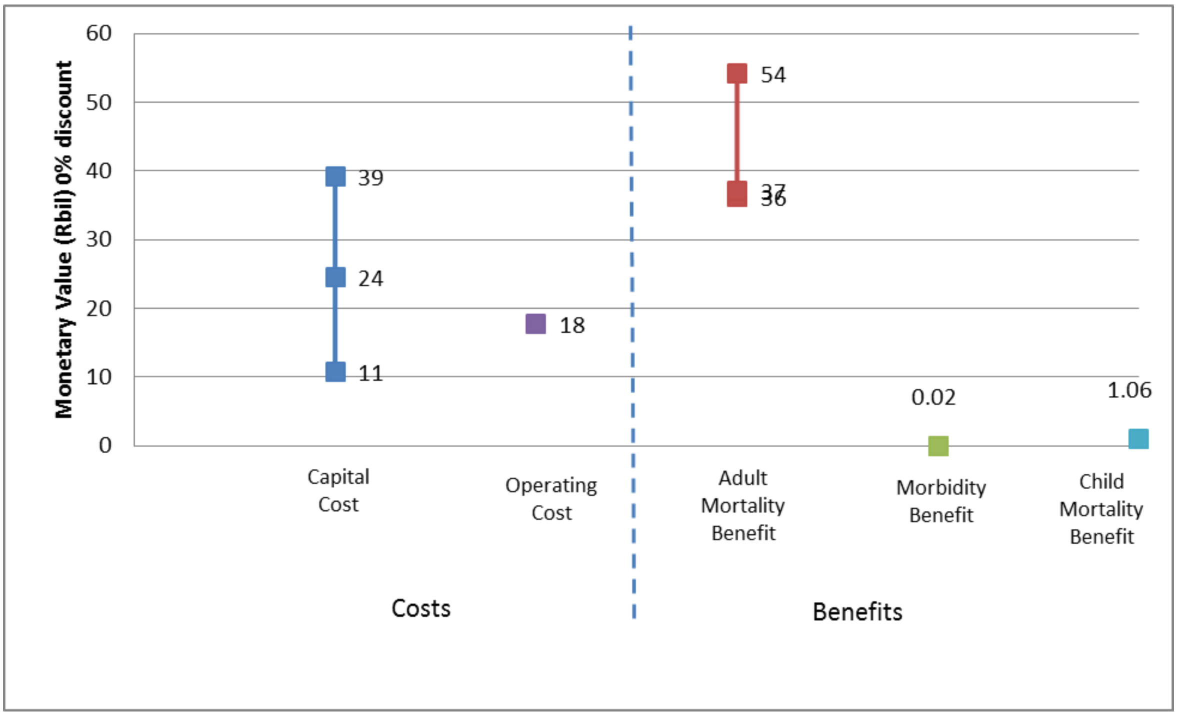

Figure 15. Summary of range of costs and benefits calculated – Kendal Station only

An intervention is generally considered economically viable if the benefits expected exceed the cost of regulation. In this particular study, the costs were found to exceed the benefits, due to the high capital cost of implementation and the significant operating costs associated with the operation of the abatement equipment.

4 PART C – CONCLUSION

Part C of the study discusses the conclusions of the case study as described in Part B. The study limitations and potential refinements for future studies are discussed.

4.1 STUDY CONCLUSIONS

The costs and benefits associated with the implementation of an SO_2 point source standard of 500 mg/Nm^3 for solid fuel combustion installations (Category 1.1 sources) were evaluated to determine the net present value of SO_2 regulation on the Mpumalanga Highveld of South Africa. All category 1.1 sources within the study expected to have a significant impact on ambient SO_2 concentrations were included in the study, as discussed in Section 3.2.4. An evaluation of the likely technology to be implemented to reach the new plant SO_2 emission standard of 500 mg/Nm^3 was conducted (refer to Section 0) and the installation of wet flue gas desulfurisation (FGD) was determined to be the technology of choice. The costs and benefits associated with the installation of FGD was identified and ranked into four categories, based on the expected impact and the availability of information. All costs and benefits that could be quantified and monetized (Category 1 impacts) were included in the evaluation.

The study used a bottoms-up or impact pathway approach to analyse the impact of emission reduction. While the methodology followed is widely used and well documented, uncertainty is introduced into the evaluation at each step of the analysis, requiring assumptions to be made which could significantly impact the result of the economic analysis. To a certain extent, uncertainties can be managed by well-considered and valid assumptions and further reduced by sensitivity analysis, yielding a range of potential values that could realise. A sensitivity analysis was conducted on the costs and benefits with the largest impact on NPV or the largest uncertainty associated with the calculation.

The impact of reduced ambient pollutant concentrations on human health was identified as the only quantifiable benefit. To evaluate the expected health benefit, the change in ambient pollutant concentrations was modelled using dispersion modelling. The choice of model and model parameters impact the expected ambient impact calculated which forms the basis of the health benefit calculation and are therefore an important consideration. The health impacts associated with reduced SO_2 and secondary sulfate concentrations were calculated using concentration response functions. The choice of appropriate concentration response functions and the applicability thereof in the South African context are important considerations, likely requiring further study. The impact on premature adult mortality was found to be the most significant benefit and dependent on the concentration response function selected and sensitive to the VSL estimate used (high R115 bil; low R36 bil). Potentially significant impacts such as the ecological impact of acid deposition was identified by the study, but was not included in the NPV calculation as they could not be adequately quantified.

The most significant costs associated with the implementation of FGD were the capital cost of installation and operating costs comprised of water and lime costs. The capital cost of FGD installation was found to be the most significant cost and was sensitive to the evaluation method (central R187 billion; high R306 billion; low R80 billion). In this case study, the operating costs of the abatement equipment was of the same order of magnitude as the health benefits. If only the capital cost of a regulation is included in an analysis, the cost implications can be significantly underestimated. Although relatively better defined than the evaluation of benefits, assumptions can also influence the calculation when evaluating the cost side of the NPV equation. The input costs such as water costs cannot be calculated on current cost of supply as the current water tariff is based on historical capital spend. Furthermore, availability of these resources and logistics associated with the provision has to be considered, which could significantly impact the NPV.

The discount rate applied was found to have a significant impact on the NPV and will be particularly important when costs and benefits flow over different time periods or when the benefit is intergenerational. A too high discount rate will render such benefits insignificant, while a too low discount rate does not adequately consider the time value of capital spend. It is recommended that capital flows and benefits be discounted at different rates, with health and ecological benefits discounted at a lower rate. This approach is in line with the concept of sustainable development.

A comparative analysis was conducted on the station with the highest health impact valuation to determine whether partial implementation of FGD could be considered viable. The results indicated that, due to the high capital cost of FGD installation, the NPV for the installation was negative under all scenarios, but that the expected mortality benefit exceeded the operating costs.

A breakeven VSL of R134 million was calculated as the VSL at which the costs and benefits of installing FGD on all the installations in the study area were equal. This implies that an individual is willing to pay (WTP) R13 402 for a one in 10 000 annual mortality risk change. This value equates to approximately 10% of South Africa's GDP per capita. The results indicate that, given the information currently available, it is unlikely that the benefit of reducing SO_2 emissions to the required standard outweighs the cost of implementation.

4.2 STUDY LIMITATIONS

The study made significant use of benefit transfer where local data was not available. Only one South African study deriving concentration response functions was found and no suitable VSL estimates for South Africa were identified. Significant uncertainty can be introduced into a study if site-specific information is not used. In order to reduce this uncertainty, sensitivity analysis was used to obtain a range of estimates.

The study reflects a basic financial CBA and can be refined to include the economic impact of the regulation, which could be significant. In this case, the basic financial CBA already indicates that it is unlikely that the benefits of regulation outweigh the costs and a full economic analysis is unlikely to change the answer, but will provide a more complete indication of the impact of the regulation.

In valuing the expected benefits of regulation, benefits such as ecological benefits due to reduced acid deposition, materials damage and impact on vegetation due to respiration could not be quantified due to insufficient information. These impacts could potentially be significant and should ideally be included in an analysis.

4.3 FURTHER WORK AND REFINEMENTS OF THE STUDY

The study evaluated mortality benefits in terms of VSL, as it is a widely used methodology. Mortality impacts could also be evaluated using the reduction in life years, or VOLY approach. It is recommended that a study be conducted to determine which approach is more appropriate in the South African context. No local VSL estimates were available for use in the study. The results indicate that the choice of VSL estimate has a significant impact on the mortality benefit, which accounts for the majority of the benefit associated with the implementation of the regulation. VSL estimates obtained from first-world studies may not be appropriate in the South African context and could overestimate the price an individual is willing to pay for a small reduction in premature mortality risk.

Due to the impact that health benefit calculations have on the evaluation of pollution control measures, further studies to calculate a South African-specific VSL/VOLY estimate for air quality improvements may be required. A standardised approach to VSL/VOLY would ensure that air quality-related CBAs have comparable results.

The ecological impact of acid deposition could potentially be significant on the Highveld, however insufficient information exists to quantify these benefits. Due to the long timeframes associated with the recovery of ecosystems and international experience with improved ecosystem functioning resulting from pollution reduction initiatives, further study is recommended.

The analysis was a simple financial analysis and the economic impact of the regulation has not been taken into consideration in this study. Many factors will influence the total economic impact, including the opportunity cost of the capital spend and the impact of increased electricity tariffs due to increased cost of production. The economic benefit of a reduced spend on healthcare will to a certain extent, offset the negative impacts and should also be considered. To an extent, the economic impact is already considered when utilizing a VSL approach, due to the value that the VSL represents. It would overestimate the benefit if that value is used as reduced healthcare spend. Eskom estimates a tariff increase of between 8 and 10% for full compliance with the PM, SO_2 and NOx standards. It is important to note that this increase will not have an associated increase in output, and it is more likely that electricity production will be negatively impacted due to increased downtime for installation of abatement and decreased overall efficiency due to additional energy demands, for example. Due to the large number of sectors that are impacted by an electricity tariff increase, the impact should be assessed using a general equilibrium model. In this case, the basic financial CBA already indicates that it is unlikely that the benefits of regulation outweigh the costs and a full economic analysis is unlikely to change the answer, but will provide a more complete indication of the impact of the regulation.

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6 APPENDIX A: STACK PARAMETERS

Point source number	Point source name	Lat	Long	Height of release (m)	Base height	Stack diameter (m)	Gas velocity (m/s)	Gas exit temp (K)	Gas volumetric flow rate (m3/h)	SO2 emission load (t/a)
1	Sasol Stack East	716.0252	7060.485	301	1612	14.4	23-27	458	11870000	91429
2	Sasol Stack West	714.1628	7060.72	250	1616	13.6	23-27	458	10558000	81323
3	Arnot Stack 1	779.6142	7127.657	195	1687	11.06	20.25	403	7003800	38637
4	Arnot Stack 2	779.6401	7127.456	195	1686	11.06	20.25	403	7003800	38637
5	Duvha Stack 1	734.2596	7126.405	300	1611	12.47	23.2	403	12592800	68618
6	Duvha Stack 2	734.3602	7126.632	300	1610	12.47	23.2	403	12592800	68618
7	Hendrina North Stack	760.383	7118.306	155.45	1651	11.14	15.4	408	7272000	56871
8	Hendrina South Stack	760.304	7118.047	155.45	1648	11.14	15.4	408	7272000	56871
9	Kriel Stack 1	717.541	7094.489	213	1633	14.3	17	403	10800000	56167

10	Kriel Stack 2	717.658	7094.286	213	1634	14.3	17	403	10800000	56167
11	Majuba Stack 1	774.816	6999.525	250	1723	12.3	22	398	15120000	87582
12	Majuba Stack 2	774.683	6999.307	250	1726	12.3	22	398	15120000	87582
13	Matla Stack 1	713.902	7091.326	213	1640	14.3	27	397	12528000	89082
14	Matla Stack 2	713.815	7091.525	275	1642	12.47	35	397	12528000	89082
15	Camden Stack 1	210.1305	7052.152	154.5	1682	8.74	13.8	423	3000000	21325
16	Camden Stack 2	210.2255	7052.176	154.5	1681	8.74	13.8	423	3000000	21325
17	Camden Stack 3	210.3001	7052.191	154.5	1679	8.74	13.8	423	3000000	21325
18	Camden Stack 4	210.3976	7052.216	154.5	1677	8.74	13.8	423	3000000	21325
19	Grootvlei Stack 1	648.9022	7038.171	152	1561	8.99	19.57	413	4474800	23929
20	Grootvlei Stack 2	648.94	7038.267	152	1563	8.99	19.57	413	4474800	23929
21	Kendal Stack 1	696.8005	7112.856	275	1643	13.51	27	399	14262000	109019
22	Kendal Stack 2	697.0509	7112.777	275	1636	13.51	27	399	14262000	109019
23	Tutuka Stack 1	733.787	7036.072	275	1642	12.3	74.71	408	10647720	89216
24	Tutuka Stack 2	734.0261	7036.091	275	1643	12.3	74.71	408	10647720	89216

7 APPENDIX B: SUMMARISED CENSUS DATA

	Receptor Code	Municipality	Receptor Name	Number of persons according to census				
				Age 0-5	age 5-18	age 18-35	age 35-55	age above 55
1	R_GM_Emba	Govan Mbeki	Embalenhle	14718	24691	41636	24030	6770
2	R_GM_Secu	Govan Mbeki	Secunda	3497	6169	12012	11530	4900
3	R_GM_Evan	Govan Mbeki	Evander	1058	1787	3836	3061	1081
4	R_GM_Lean	Govan Mbeki	Leandra	5121	1100	1710	921	603
5	R_GM_Kinr	Govan Mbeki	Kinross	2351	4098	6581	4618	1097
6	R_GM_Beth	Govan Mbeki	Bethal	7683	15182	20901	15601	7534
7	R_GM_Char	Govan Mbeki	Charl Cilliers	1393	2391	3793	2177	654
8	R_L_Stan	Lekwa	Standerton	11963	22983	30021	22195	10783
9	R_L_Morg	Lekwa	Morgenzon	2329	4707	5010	3834	1831
10	R_PS_Amer	Pixley ka Seme	Amersfoort	3295	7010	5450	3255	2237
11	R_PS_Perd	Pixley ka Seme	Perdekop	1332	2547	2431	1707	1052
12	R_M_Dave	Msukaligwa	Davel	1141	2440	2400	1577	926
13	R_M_Erme	Msukaligwa	Ermelo	9784	17254	26811	15950	6570
14	R_M_Ward8	Msukaligwa	Ward 8	1527	3116	3807	3576	1570
15	R_M_Brey	Msukaligwa	Breyten	2061	4025	4670	3123	1447
16	R_D_Aggr	Dipaliseng	Aggregate	5450	9664	13299	8996	4982

17	R_L_Heid	Lesedi	Heidelberg	7981	14015	22241	15767	7669
18	R_L_Ward10	Lesedi	Ward 10	724	1397	2031	2056	1021
19	R_L_Ward6	Lesedi	Ward 6	832	1300	1908	1451	720
20	R_L_Ward12	Lesedi	Ward 12	2254	3979	5935	4192	2027
21	R_VK_Delm	Victor Khanye	Delmas	6168	11149	15100	9765	3830
22	R_VK_Ward7	Victor Khanye	Ward 7	1265	2406	3442	2636	1576
23	R_VK_Ward9	Victor Khanye	Ward 9	1447	2492	3749	2896	1441
24	R_VK_Ward8	Victor Khanye	Ward 8	597	1075	1763	1743	907
25	R_ST_Ward7	Steve Tshwete	Ward 7	631	1092	1969	1506	628
26	R_ST_Hend	Steve Tshwete	Hendrina	3187	5812	7976	5156	2285
27	R_ST_Ward5	Steve Tshwete	Ward 5	311	577	1623	1033	407
28	R_ST_Ward6	Steve Tshwete	Ward 6	630	1306	2432	1942	490
29	R_ST_Ward8	Steve Tshwete	Ward 8	1433	2019	4395	2397	629
30	R_ST_Ward9	Steve Tshwete	Ward 9	833	1544	2011	1482	760
31	R_ST_Midd	Steve Tshwete	Middelburg	17122	30282	51156	38464	14850
32	R_ST_Ward16	Steve Tshwete	Ward 16	689	1408	2128	2226	1357
33	R_ST_Ward29	Steve Tshwete	Ward 29	990	1607	1892	1447	672
34	R_ST_Ward4	Steve Tshwete	Ward 4	411	705	2001	1349	543
35	R_E_Kwa	Emalahleni	Kwa Guqu	14481	25314	37340	24477	6696
36	R_E_Krie	Emalahleni	Kriel	1642	3228	6165	4779	2292
37	R_E_Ogie	Emalahleni	Ogies	1344	2024	3960	2554	971
38	R_E_Ward28	Emalahleni	Ward 28	3238	5850	9018	5985	2054
39	R_E_Ward32	Emalahleni	Ward 32	1246	1999	4301	3073	951
40	R_E_Ward25	Emalahleni	Ward 25	1993	3320	5303	3330	992
41	R_E_Ward19	Emalahleni	Ward 19	1256	1975	4852	3068	998

42	R_E_Ward29	Emalahleni	Ward 29	2778	4496	6639	4766	1097
43	R_E_Ward12	Emalahleni	Ward 12	1403	2241	6643	3805	1164
44	R_E_Ward34	Emalahleni	Ward 34	1507	2533	5256	4698	2458
45	R_E_Witb	Emalahleni	Witbank	15345	25263	51799	34183	15291

8 APPENDIX B: METEOROLOGICAL DATA

Wind rose data for the modelling period at three stations located on the Highveld are shown in Table and Table . A map indicating the position of the three stations relative to the sources is shown in Figure 16.

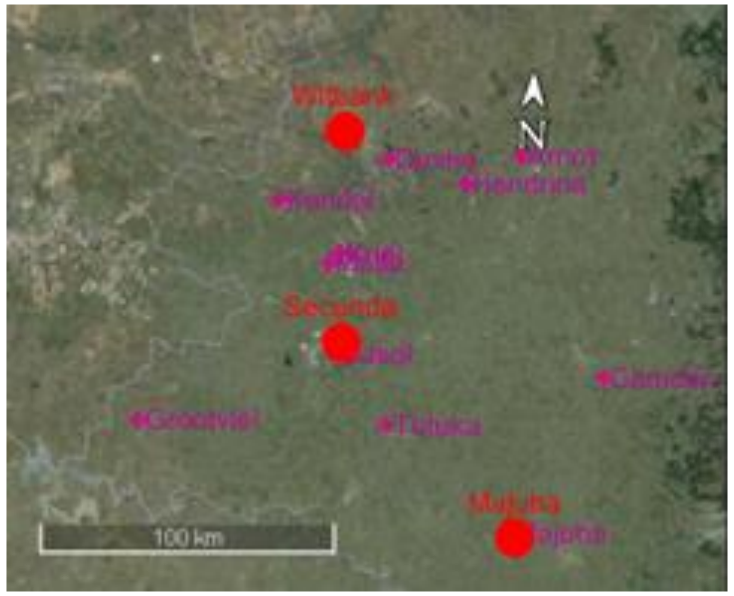
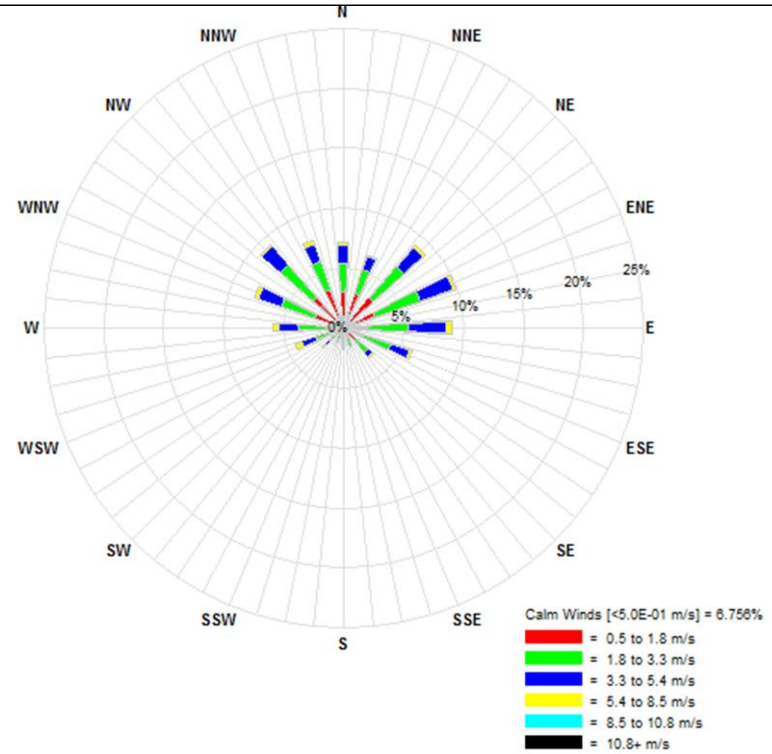
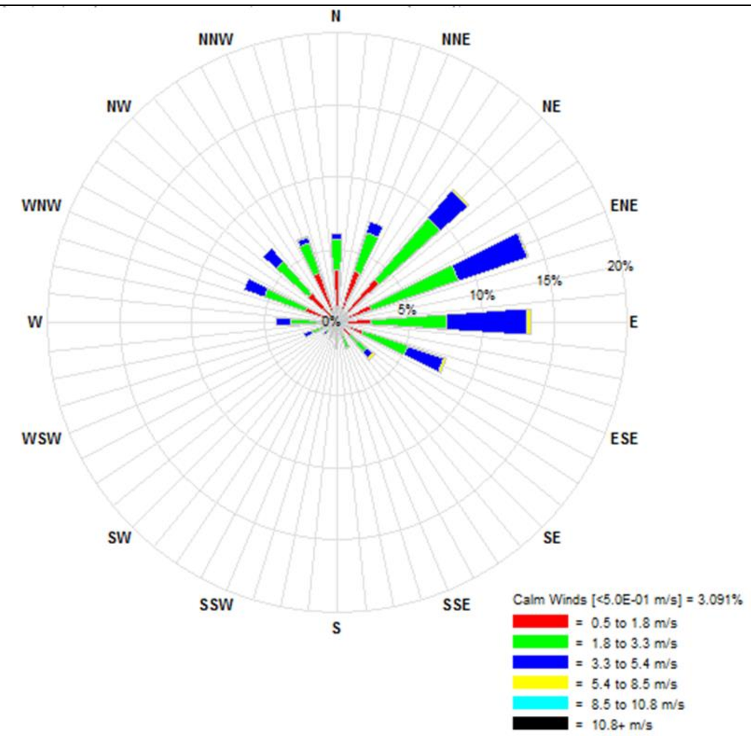
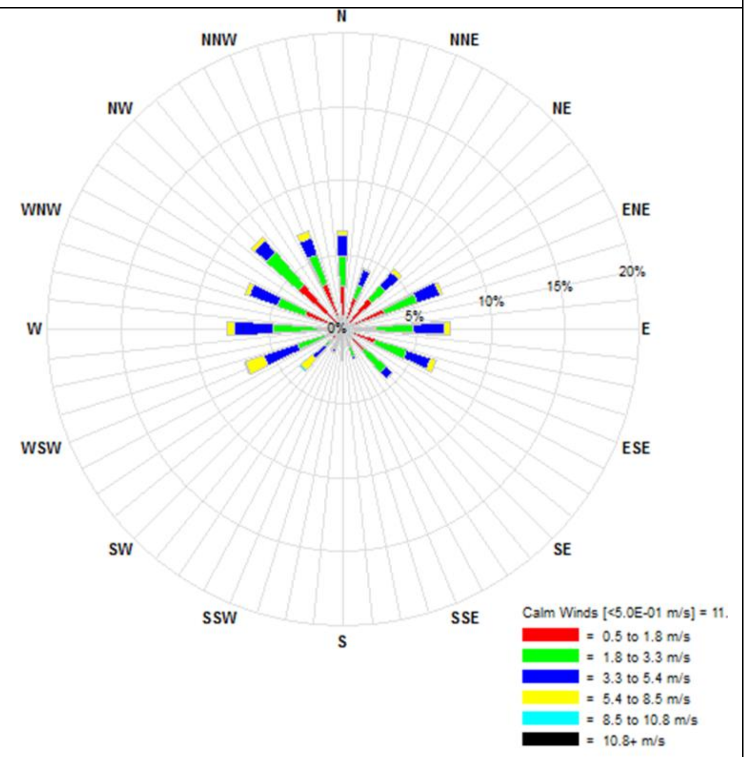
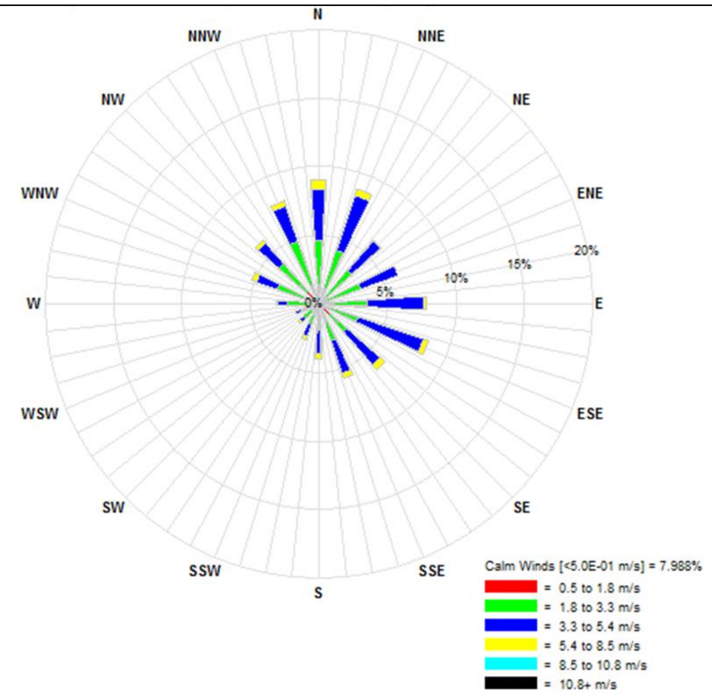
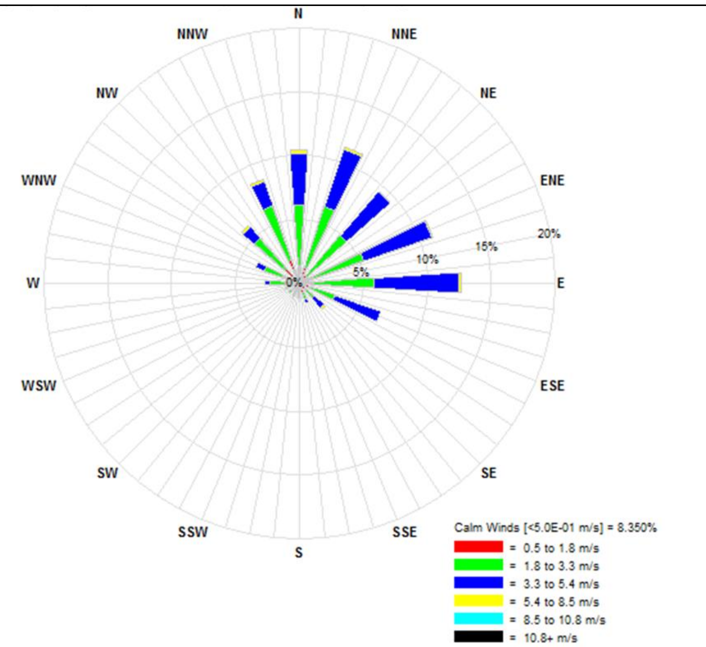
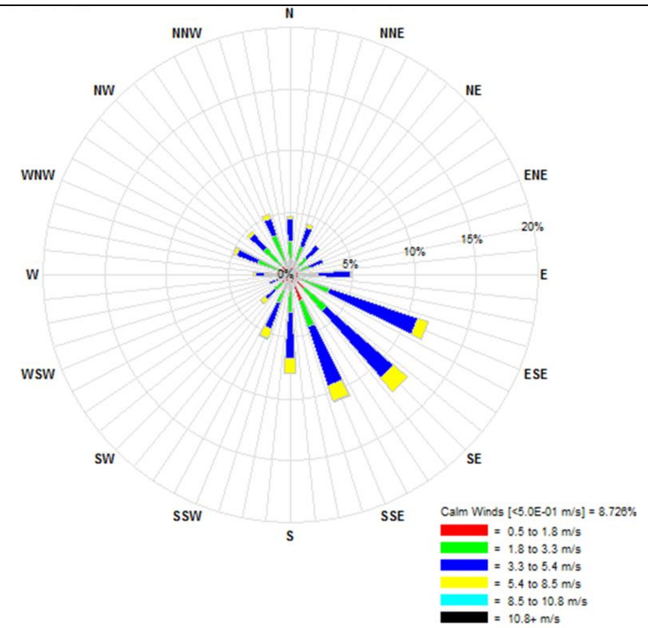


Figure 16. Location of stations for wind rose data

Table 24. Wind rose data for the modelling period, summer period and winter period

Location	Modelling period average	Summer	Winter
Secunda			
Witbank			

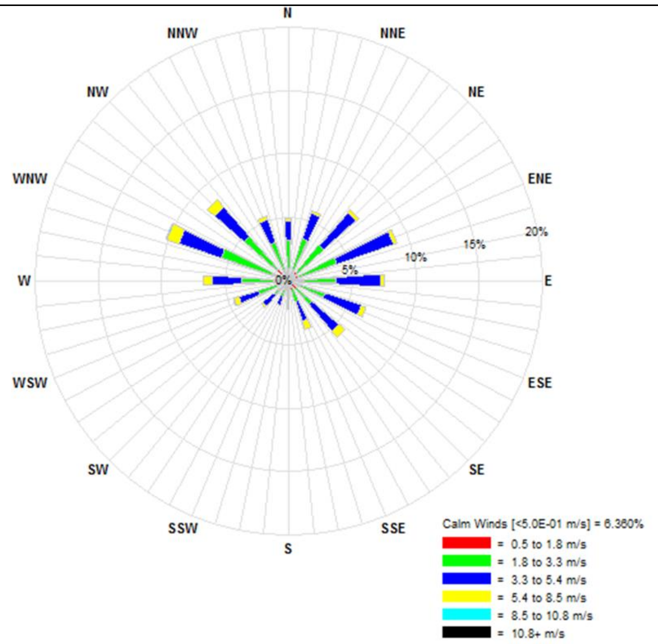
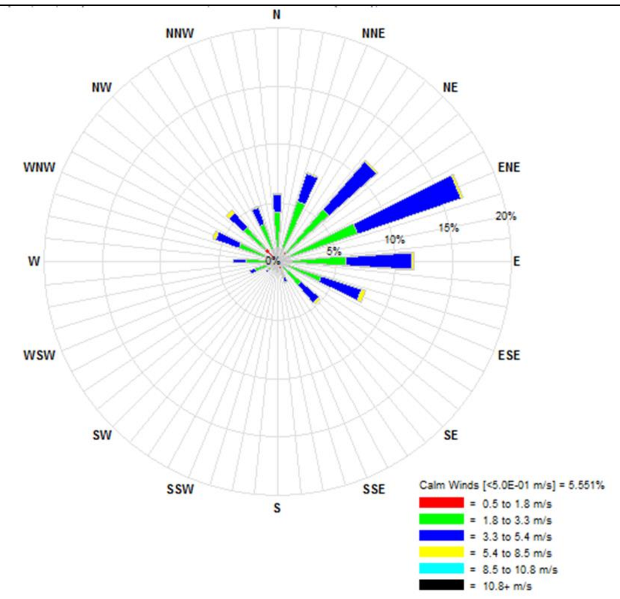
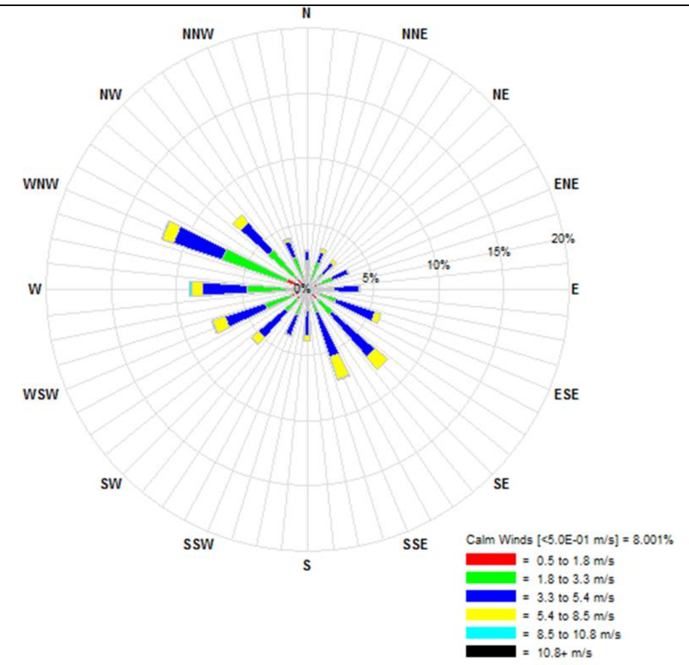
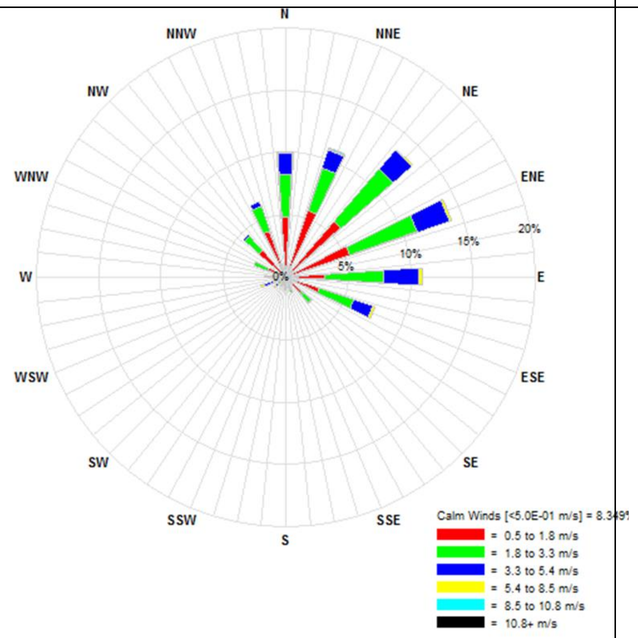
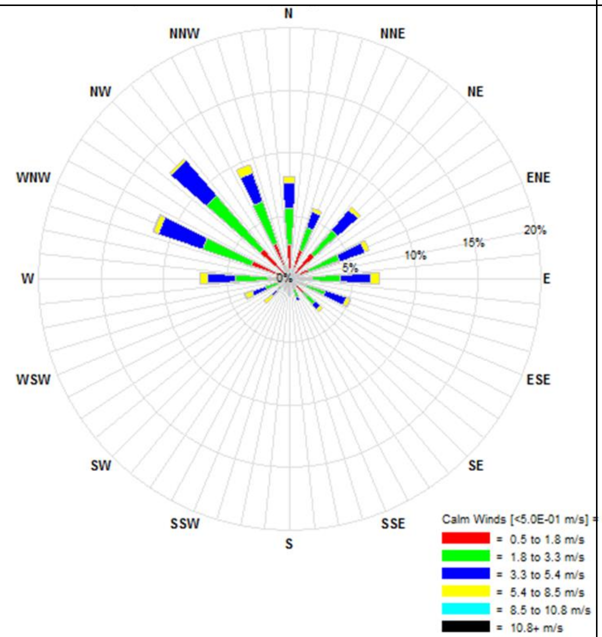
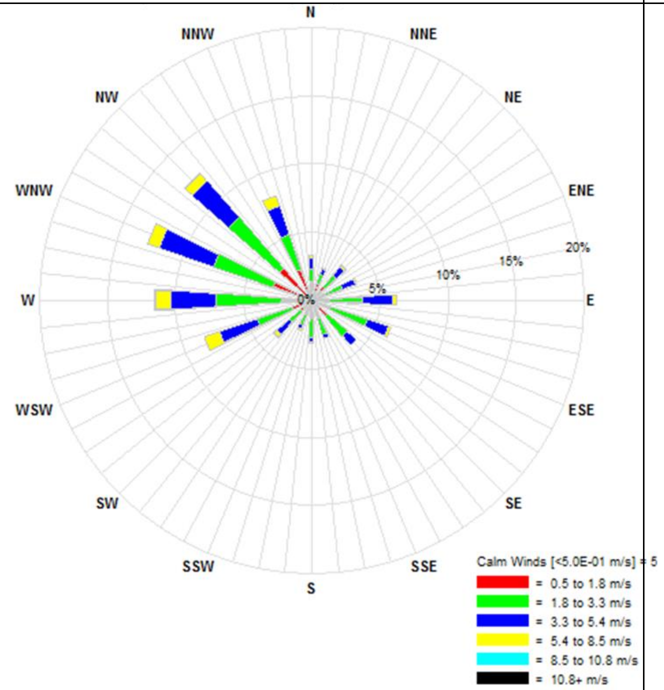
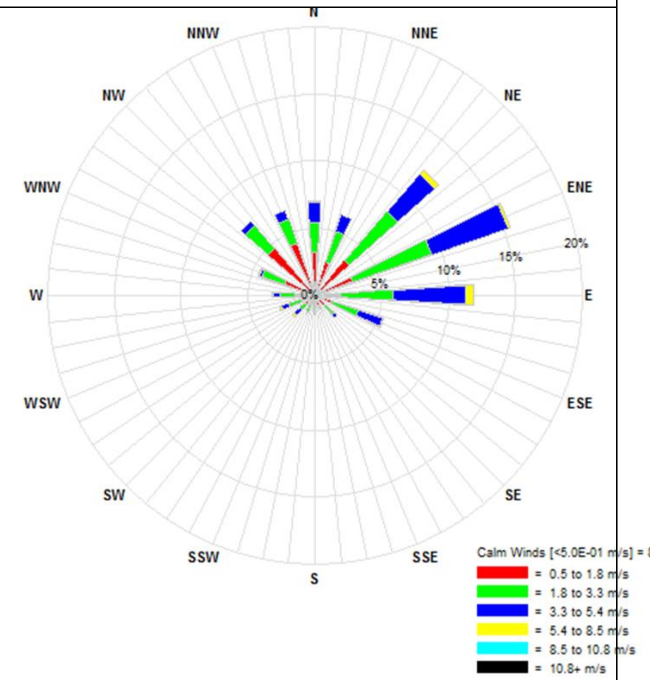
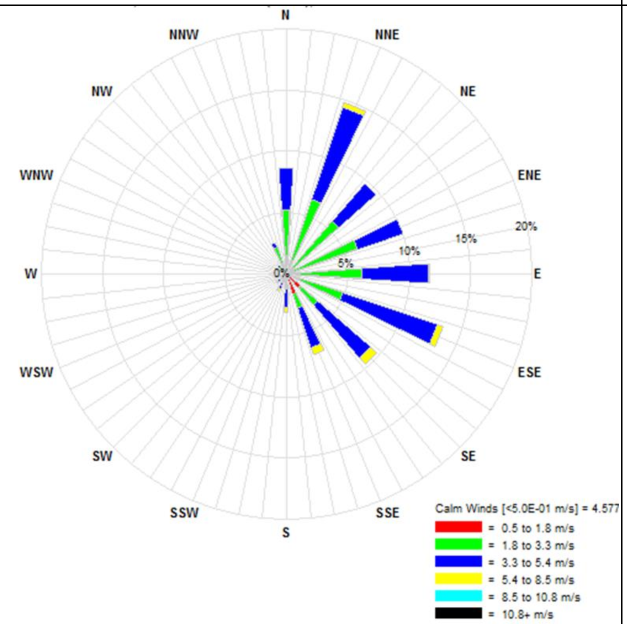
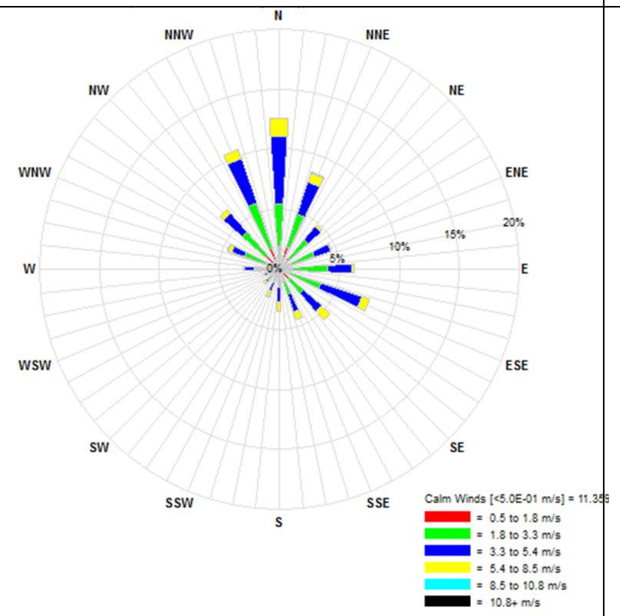
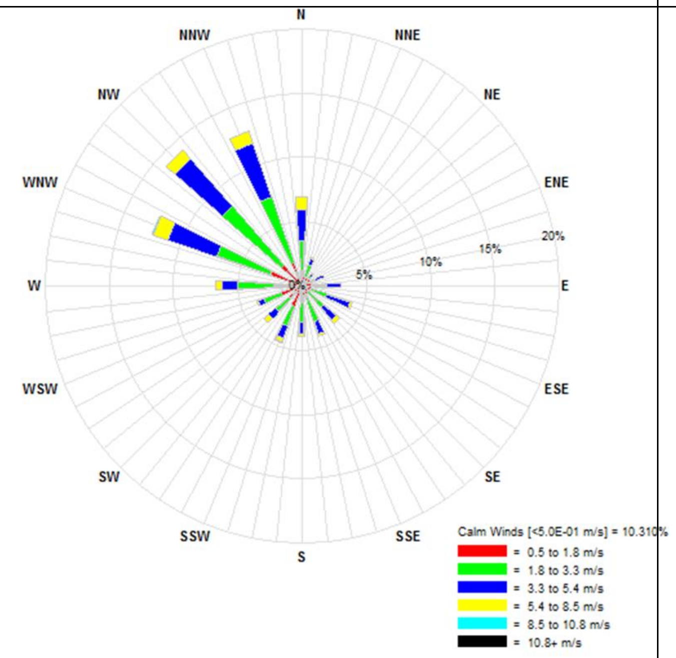
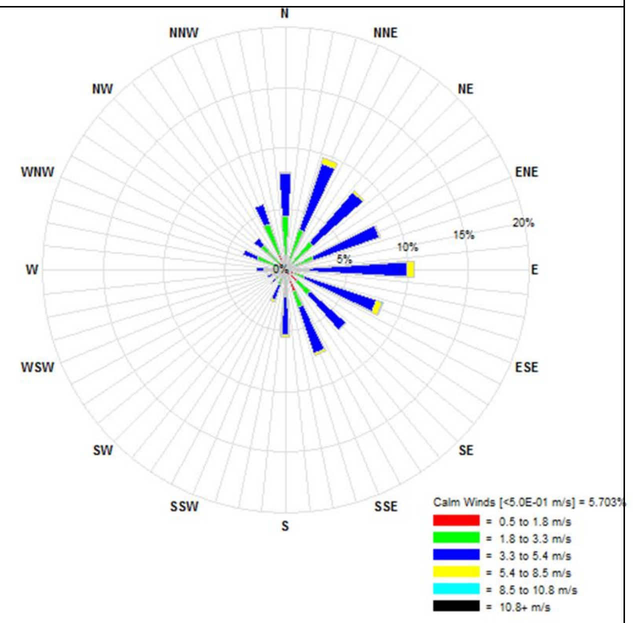
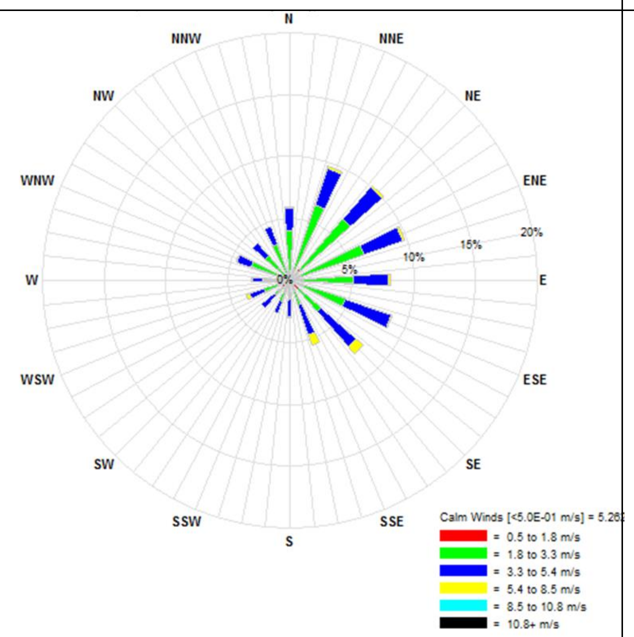
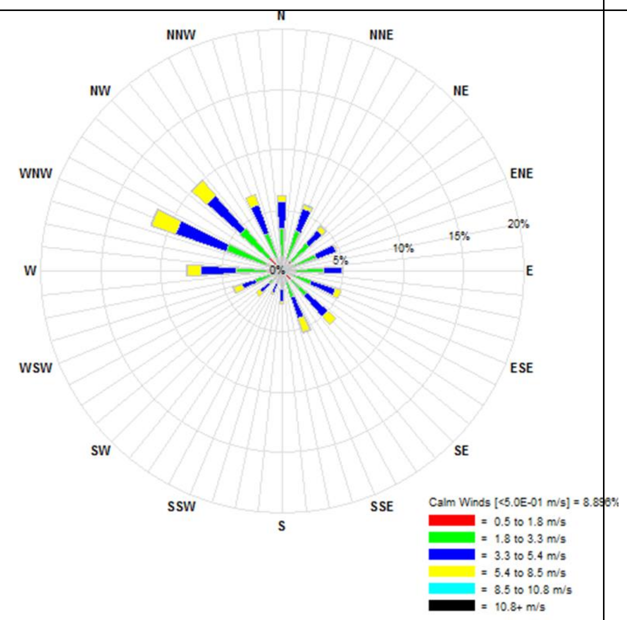
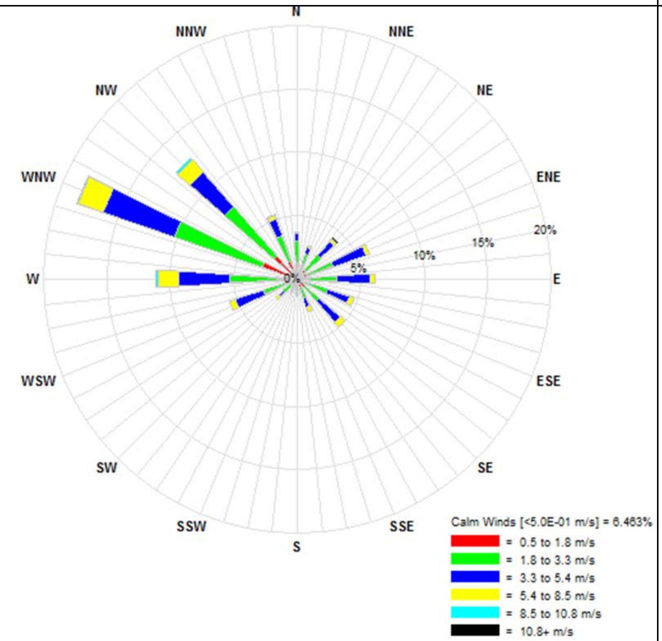
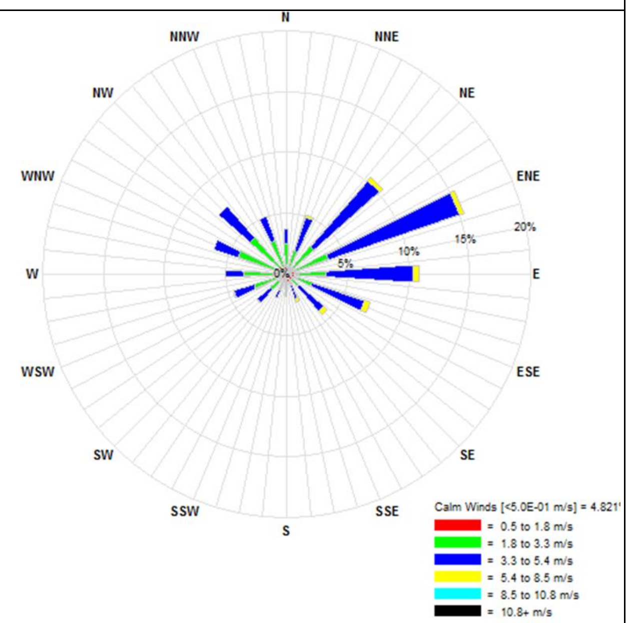
Location	Modelling period average	Summer	Winter
Majuba			

Table 25. Wind rose data by time of day

Location	Hours 01-06	Hours 07-12	Hours 13-18	Hours 19-24
Secunda				

Location	Hours 01-06	Hours 07-12	Hours 13-18	Hours 19-24
Witbank				
Majuba				

9 APPENDIX C: DAILY MODELLED VALUES

In evaluating the short term health impacts, daily modelled values are used in the calculation. The additional risk per day is evaluated and summed to obtain an increased annual risk. For illustrative purposes, the results for the daily modelled results are presented graphically in Figure 17 and Figure 18.

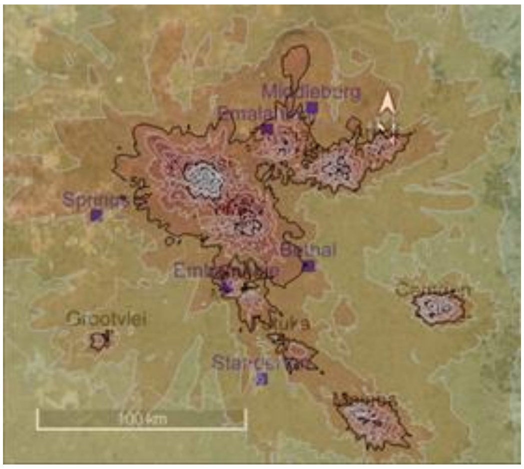


Figure 17. Modelled daily average (fourth highest value) modelled SO_2 concentration (µg/m^3) over the study area for baseline conditions

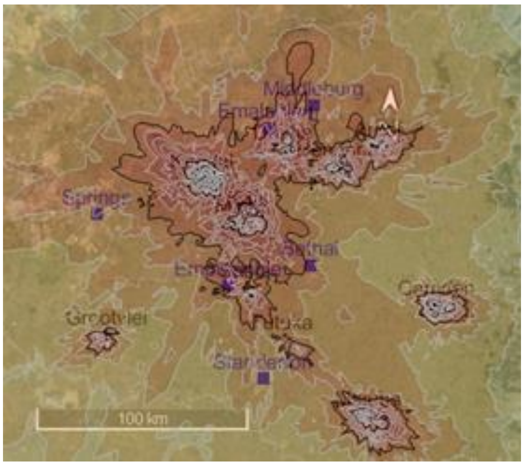


Figure 18. Modelled daily average (fourth highest value) modelled SO_2 concentration (µg/m^3) over the study area for compliance conditions

10 APPENDIX D: IMPACT OF GRANDFATHERING

It is common practice to distinguish between standards for new facilities and standards for older facilities, commonly referred to as grandfathering. Grandfathering can create unintended negative environmental consequences, as discussed in a report by Nash and Revesz (2007). In the United States, the Clean Air Act specifies that existing plants have to comply with new source standards, once modified. The extent and nature of the modifications that require facilities to meet new plant standards have been a source of debate and legal action for many years in the United States (Nash, 2007). The concern with grandfathering is that existing sources may choose to operate facilities past their economic lifetime in order to avoid compliance with strict new source standards. Facilities may further avoid maintenance activities to avoid classification as a new source. Stringent new source standards may discourage the building of new sources and increased reliance on old sources, which does not lead to environmental improvements and may lead to further deterioration. In areas where ambient air quality standards are exceeded, the less stringent treatment of existing facilities is problematic, as it impedes the construction of new facilities and expansion of others, as the capacity of the airshed has been reached.

There are mainly three ways in which to reduce or eliminate the impact of grandfathering – phasing out of grandfathering clauses to compel old sources to comply with new source standards, reducing the gap between new and old source limits by making new plant standards less stringent and by reducing the old source limits to values closer to the new source limits (Nash, 2007). South African legislation has taken the first approach, whereby there will be no distinction between new and existing source standards. In many cases, replacement of existing sources with new sources or significant retrofitting of abatement equipment to all sources will be required for full compliance, thereby aiming to eliminate the negative environmental consequences of grandfathering.

New South Wales, an Australian state, has taken the approach of gradually tightening old source standards closer to new source standards as new technologies become available. Several categories of sources are identified according to the age of the facility. Substantial modifications are required to comply with new source standards.

In order to determine the impact of grandfathering, the costs and benefits were re-calculated for only sources that have an expected lifetime exceeding 20 years from 2020. This would ensure that sources with a limited lifetime are not required to expend high capital cost to retrofit abatement equipment, rather phasing out such facilities. The results are shown in Figure 19.

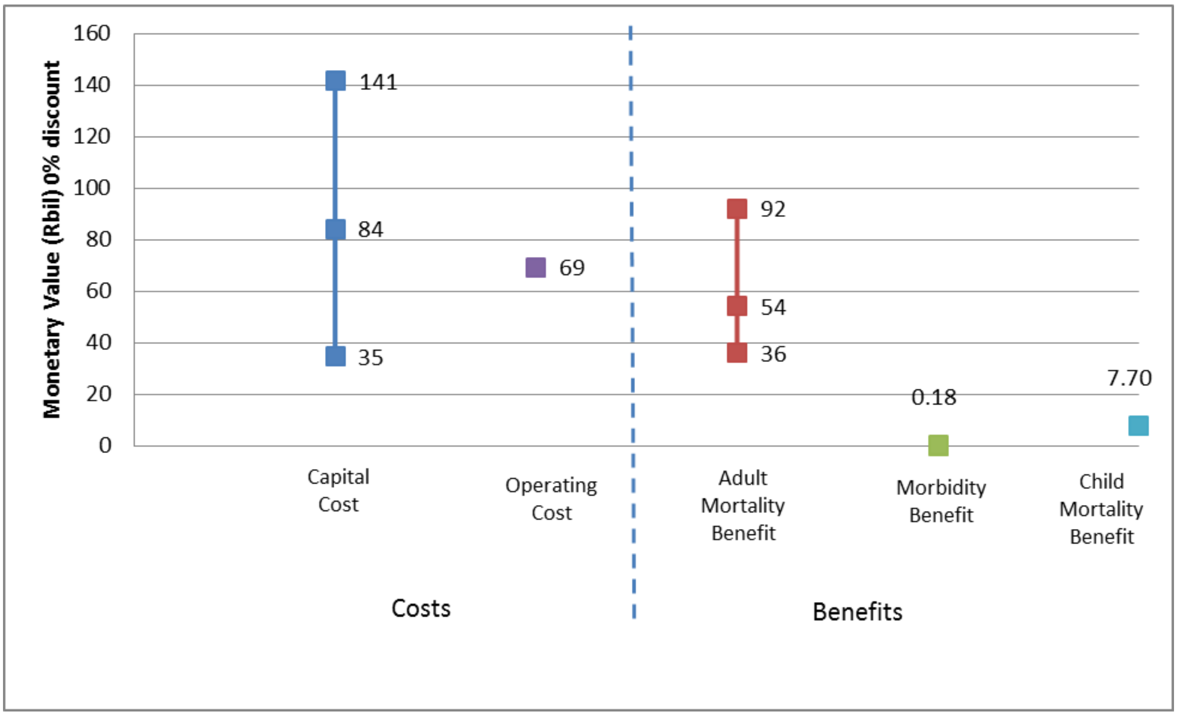


Figure 19. Retrofitting of facilities with an expected remaining lifetime exceeding 20 years from 2020

The results indicate that significantly lower capital cost is required, with a central estimate of R84 billion. In this scenario, the operating costs and mortality benefits are similar in magnitude, implying that discounting will not have such a significant impact on NPV. The results indicate that, due to reduced capital spend required, the economic impact will be lessened when compared to the retrofitting of all facilities. However, due to the large operating costs and significant capital spend still required, the basic financial analysis indicates that, even with limited grandfathering, the cost of compliance is likely to outweigh the benefit.

11 APPENDIX E: IMPACT OF USING A TOP-DOWN APPROACH

In order to determine the impacts of choice of CBA methodology, a top-down or damage cost approach was used to compare to the results from a bottoms-up or impact pathway approach. As part of the CAFÉ CBA, damage costs were calculated from the results. The value for the SO2 damage cost calculated did not include the direct impacts of SO2 on health or impacts related to ecosystems and cultural heritage. The impact of secondary particulates from SO2 emissions was taken into consideration. The study calculated the damage cost of SO2 as between €3700 and €11000 (2005 costs) depending on the VSL and VOLY estimates used.

The damage costs recommended by the UK Department of Environment, Food and Rural Affairs calculate damage costs of between £1208 and £1698 (2008 costs). These costs include direct health impacts of SO2 as well as damage to materials and health impact due to secondary particulates. (https://www.gov.uk/government/uploads/system/uploads/attachment_data/file/182391/air-quality-damage-cost-methodology-110211.pdf)

Using these DEFRA costs, a high and low benefit of R1200 billion and R1687 billion is calculated. The costs when using the CAFÉ estimates would be even greater (in excess of R3000 billion as a low estimate).

Care should therefore be taken when using damage costs from a study conducted elsewhere, as the results could yield an unrealistic view of the actual damage costs.