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NACA Courses 2018

4-Day CALPUFF and AERMOD Modeling Course

Centurion, Gauteng

16 - 19 July 2018

Who should attend:

Current or aspirant air quality practitioners in the public and private sector, specifically persons responsible for performing or evaluating applications for Air Emissions Licences, based on dispersion modelling. Intending participants should have at the minimum knowledge of the basics of meteorology and air quality management.

Lecturers and Course Content:

Two course-days are dedicated to the CALPUFF Module and two days to AERMOD Modeling. The course is presented by a senior lecturer from the Canadian company, Lakes Environmental.

Certification:

A certificate of attendance and satisfactory completion will be issued by NACA.

Course fees:

For more information on course fees, special discounts and group rates, please visit the NACA website.

5-Day Air Quality Management Courses

Cape Town, Western Cape

13 - 17 August 2018

Who should attend:

Current or aspirant air quality practitioners in the public and private sector, specifically persons responsible for implementing aspects of the NEM: Air Quality Act. Intending participants should have knowledge of the basics of air quality with a bachelor's degree, a national diploma or several years of relevant experience

Certification:

A certificate of attendance and satisfactory completion will be issued by NACA.

Course fee: R10,500.00 (Excl. 15% VAT)

Fee includes participation in the course, a full set of printed notes and CD disk and certificate. Morning and afternoon tea, and daily lunches will be provided.

Visit the NACA website at www.naca.org.za for more information on course content and registration forms.

Editorial
South African Air Quality Information System (SAAQIS) mobile application tool: bringing real time state of air quality to South Africans

Patience Gwaze and Sindisiwe H. Mashele
Department of Environmental Affairs, Pretoria

http://dx.doi.org/10.17159/2410-972X/2018/v28n1a1

In terms of Section 24 of the Constitution, as well as the National Environmental Air Quality Act (AQA, 2004), government is charged with the role to ensure that South Africans are breathing air that is not harmful to their health and wellbeing. Several spheres of government monitor the state of air quality across the country at over 130 fully automated air quality monitoring stations (Figure 1). These stations monitor a range of pollutants including ozone (O_3), particulate matter (both PM_{10} and $PM_{2.5}$), carbon monoxide (CO), sulphur dioxide (SO_2), and oxides of nitrogen (nitrogen dioxide NO_2, and nitric acid NO), lead (Pb), hydrogen sulphide (H_2S), black carbon (elementary carbon) and meteorological parameters. The stations form the National Ambient Air Quality Monitoring Network (NAAQMN) and are located in areas with the highest density of people in order to measure human exposure to air pollution. The stations provide critical information to assess compliance with ambient air quality standards and to assess the impact of intervention strategies aimed at addressing air pollution. In addition, data from these monitoring stations also provide valuable information regarding the state of ambient air quality to which the citizens of the Republic are exposed.

South Africans have a right to ambient air quality monitoring information monitored at all of the stations commissioned by government. This information is disseminated through the South African Air Quality Information System (SAAQIS), a partnership between the Department of Environmental Affairs (DEA) and the South African Weather Service (SAWS). SAAQIS is a 'one-stop-shop' for all air quality information, from monitoring to legislation, as well as notices, guidelines and contact information of air quality officials in different jurisdictions across the country. The SAAQIS has recently been upgraded to the second generation system that was launched in October 2017 during the Annual Air Quality Governance Lekgotla. At the centre of the SAAQIS is the ambient air quality monitoring module which provides the public with REAL TIME information on the state of air quality. This information is available on the SAAQIS website (https://saaqis.environment.gov.za) and the mobile application called SAAQIS on both Android and IOS platforms. To date, over 60 stations are reporting LIVE air quality to the SAAQIS. It is envisaged that all operational stations will be available online by the end of 2018.

Ambient pollutant measurements are complex for the general public, specifically issues such as how measured pollutant concentrations translate to the quality of air and associated health effects. DEA has development a country-specific Air Quality Index (AQI) in line with best international practices in order to simplify the reporting of air quality to the general public. Once sanctioned, this AQI will be the official manner by which air quality will be presented to the public. The AQI is derived from six (i.e. PM_{10}, $PM_{2.5}$, CO, O_3, SO_2 and NO_2) criteria pollutants, for good air quality (scale 1) to hazardous (10) based on National Ambient Air Quality Standards (NAAQS). The index has five bands indicating 'Low', 'Moderate', 'High' and 'Very High' and 'Hazardous' levels of air pollution. These bands are further divided into a ten-point scale to provide greater gradation of air pollution levels. The 'Low' bands indicate air pollution levels where it is unlikely that anyone will suffer any adverse effects of short-term exposure, including people with lung or heart conditions who may be more susceptible to the effects of air pollution. The 'Moderate' band represents levels of air pollutants at which there are likely to be minor effects for susceptible people only. Values for the 'High' bands are associated with significant effects in susceptible people. At 'Very High' levels of air pollution even healthy individuals may experience adverse effects of short-term exposure. The 'Hazardous' levels will trigger health warnings of emergency conditions as the entire population will likely be affected by serious health effects.

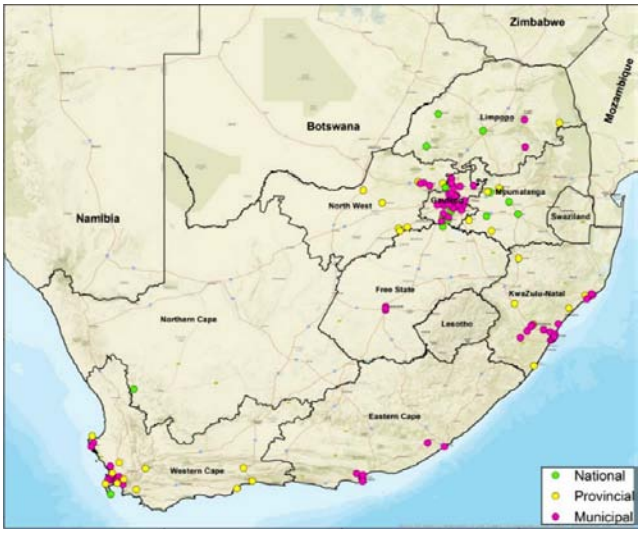

Figure 1: *National Ambient Air Quality Monitoring Network (NAAQMN) of South Africa. Green- national stations managed by the South African Weather Service (SAWS), yellow – provincial stations and magenta – district/metropolitan stations. (Map drawn in-house)*

The reported AQI is derived from the maximum of the pollutant-specific sub-index and associated health effects statements. This means that for each hour, the ambient pollutant concentrations are averaged, and assigned, according to the 10 bands for each pollutant to define the sub-indexes. The maximum of these sub-

indexes is then assigned to the respective hour as the AQI for each station. Because the AQI is defined for each hour, the index is intended to offer information for protection from those pollutants that have short-term health effects (acute effects) such as sulphur dioxide, oxide of nitrogen and ozone. Once the AQI is made available to the public, the AQI will also provide guidance and messages to be included when ambient air quality information is reported on several communication platforms such as the SAAQIS application, television, radio and the Internet.

Figures 2 to 5 show state of air quality in South Africa as of 24 May 2018 from 13h55 to 16h30 from the SAAQIS application screenshots. Users are able to display the AQI, actual pollutant measurements and meteorological data for each monitoring station. The default display is based on LIVE data averaged on the hour. The AQI display is also accompanied by emoticons which are colour-coded to the AQI of that hour (a pleasant green face when the air quality good, and an orange/red/purple sad face when the air quality is unhealthy) as shown in Figure 2.

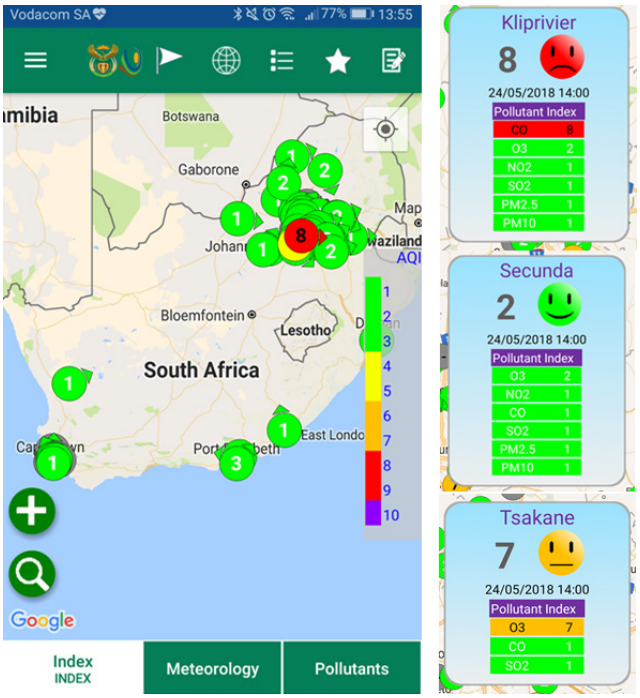

Figure 2: *Mobile device screenshot of SAAQIS application showing the state of air in South Africa as of 24 May 2018 at 13h55. Each monitoring station is represented by a circle colour-coded to the AQI of that station for that hour (greyed circles are stations that are not online at that moment).*

By clicking on a station, the application displays more information about that station, pollutants monitored, time series plots of pollutants and meteorological parameters (Figure 3). Time series plots can be presented over a day or two for any parameter as shown for SO_2 at Secunda. The SAAQIS application not only provides information on the state of air quality, but also educational information on pollutants, what the pollutants are, typical pollutant sources and associated health effects. This educational tool will further be enhanced on the SAAQIS website to support primary educational programmes and information for the general public.

Data presented in the SAAQIS application and website are quality controlled using default data management algorithms in SAAQIS

Figure 3: *Time series plot and general information about pollutants. For this display it is general information about SO_2 and SO_2 measured on 23 May 2018 at Secunda.*

to remove suspicious data spikes before display. All changes to data are noted in the database for verification by data specialists during additional data verification processes that are undertaken by ambient station managers before publishing monthly reports.

The SAAQIS application also includes these additional tools:

- Option for a user to customise display settings, select/unselect favourite stations and share information from the application (Figure 4).
- Ability for users to analyse data and generate a range of reports as shown on the right-hand side menu of Figure 4.

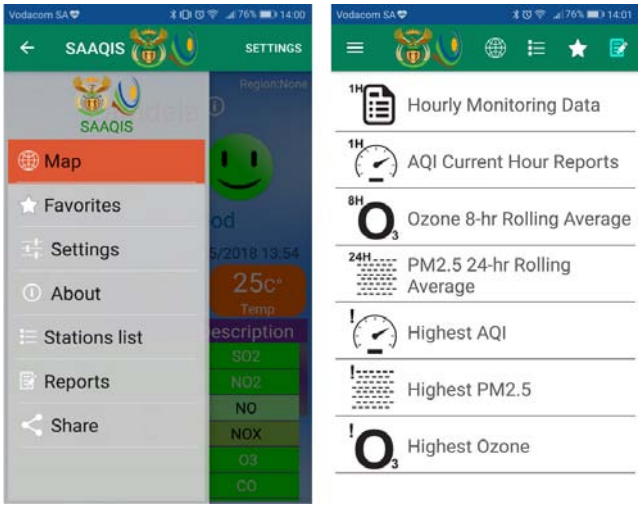

Figure 4: *SAAQIS Application menu of tools for data analyses and application customisation.*

While the SAAQIS application and website are currently displaying only government-owned monitoring stations, there are engagements with private network owners to provide their information into the system too. In addition, there are new initiatives underway to resuscitate non-operational government-owned ambient monitoring stations. By the end of 2018, more stations will be reporting to the SAAQIS, thereby improving coverage and access of information on the state of air quality to the general public. DEA is also undertaking roadshows with university students to educate them on SAAQIS application, air quality tools and how the country is disseminating air quality information to the public. During these roadshows, DEA is also inviting students to design visionary concepts on how the SAAQIS application can further be enhanced to meet community needs.

Save the Dates and Call for Papers

2018 Conference of the National Association for Clean Air
30 October - 1 November, Vanderbijlpark, Gauteng

CONFERENCE WEBSITE & REGISTRATION OPEN 14 MAY 2018

3 Three-day conference!

The 2018 Conference of the National Association for Clean Air will be held at the Riverside Sun in Vanderbijlpark from Tuesday, 30 October to Thursday, 1 November 2018. Day one of the conference will start with an afternoon session on Policy, followed by feedback from the 13th Air Quality Governance Lekgotla and conclude with a Technical Session. Days two and three of the conference will both kick-off with an early start and end at five o'clock. An informal braai on the banks of the Vaal River is also included in the programme, to take place on Wednesday evening.

2018 Conference Rates

The NACA Council has decided that the Early Bird conference rate will only be available to paid-up NACA Members. Should you not be a NACA Member, but still wish to pay the discounted Early Bird Rate, a R250.00 NACA Membership fee will automatically be included in the conference fee. Students presenting a poster or paper at the conference will again be able to do attend at a subsidised rate.

NACA Member Early Bird Fee *Until and due by 30 September 2018*	R5 900.00 (Excl. 15% VAT)
Non-NACA Member Early Bird Fee *Until and due by 30 September 2018* *(Includes 12 month, NACA Membership fee)*	R6 120.00 (Excl. 15% VAT)
Standard Conference Fee *From 1 October onwards* *Due before the start of the conference*	R6 500.00 (Excl. 15% VAT)
Student Fee *Due by 30 September 2018*	R3 500.00 (Excl. 15% VAT)

CALL FOR PAPERS

The Organising Committee of the 2018 NACA Conference invites submissions of papers for the annual conference. Presenters are requested to register and submit their abstracts on the electronic submission and evaluation system. Guidelines for papers and posters will be made available on the conference page of the NACA website.

DEADLINES

Abstract submission
30 May 2018

Notification of acceptance of abstracts
4 June 2018

Full papers due
30 July 2018

Notification of acceptance and comments on papers
20 August 2018

Authors resubmit papers if required indicating how reviewers' comments were addressed (by including a covering page to their paper)
17 September 2018

Reviewers indicate whether comments were sufficiently addressed
24 September 2018

Submission of final papers for inclusion in the electronic conference proceedings
8 October 2018

For more information on the 2018 NACA Conference visit the website at www.naca.org.za

Editorial
News on air pollution and health data and impacts on health from the World Health Organization

Sophie Gumy and Pierpaolo Mudu
World Health Organization

http://dx.doi.org/10.17159/2410-972X/2018/v28n1a2

In the recent months, the World Health Organization (WHO) launched important news related to air pollution and health. The first is the new global data on exposure and burden of disease from air pollution – both household and ambient. The second is the new version of WHO AirQ+ software that estimates the impacts of air pollution on health. The third is related to the BreatheLife communication campaign that was launched last year together with the United Nations Environment Programme and the Climate and Clean Air Coalition, and whose audience include the general public, city authorities, and health professionals.

Air pollution levels are at seriously high in several areas of the world. Major sources of air pollution from particulate matter include the inefficient use of energy by households, industry, the agriculture and transport sectors, and coal-fired power plants. In some regions, sand and desert dust is also a large source of $PM_{2.5}$ and waste burning and deforestation are additional sources of air pollution in some areas. Around 3 billion people – more than 40% of the world's population – still do not have access to clean cooking fuels and technologies in their homes, the main source of household air pollution, and Sub-Saharan Africa is lagging behind. WHO released new data in May 2018, that shows that 9 out of 10 people breathe air containing unsafe levels of pollutants and 7 million deaths every year are attributable to ambient (4.2 million deaths) and household air pollution (3.8 million) from particulate matter with a diameter of 2.5 micrometer or less ($PM_{2.5}$). Fine particles penetrate deep into the lungs and cardiovascular system, causing diseases including stroke, heart disease, lung cancer, chronic obstructive pulmonary diseases and respiratory infections, including pneumonia. More than 90% of air pollution-related deaths occur in low- and middle-income countries, mainly in Asia and Africa.

As custodial agency for the three air pollution related Sustainable Development Goals (SDG) indicators - access to clean energy, air quality in cities and mortality from air pollution -, WHO is hosting several databases related to exposure to air pollution. More than 4300 cities in 108 countries are now included in WHO's ambient air quality database, making this the world's most comprehensive database on ambient air pollution. WHO Data Integration Task Force developed the Data Integration Model for Air Quality (DIMAQ) to combine data from multiple sources in order to provide estimates of population exposures to $PM_{2.5}$ at high spatial resolution ($0.1° \times 0.1°$) globally. WHO also maintains a database storing information on the technologies and fuels used for major household energy end uses (e.g. cooking, heating, lighting) from over 1100 nationally-representative surveys and censuses. The calculation of the burden of air pollution on health is related to modelled data from monitoring stations and satellite estimates for ambient and modelled data from the percentage of population using polluting fuels and technologies for cooking. Estimating attributable deaths and diseases from air pollution are the essential starting point for developing or adjusting policies and interventions that safeguard people's health.

To support the quantification of the health effects of exposure to air pollution WHO has updated the software tool AirQ+ that was originally launched in May 2016 to replace the AirQ software already online for 12 years. The new tool calculates the health effects of exposure to air pollution from various pollutants, including estimates of reduction in life expectancy. In the 2 years since its launch, thousands of users from all over the world have used the software, which has been very recently translated in Russian and a French translation is ongoing.

In conjunction with the burden of disease data publication, global communications campaign BreatheLife has launched a challenge to encourage citizens to take action to reduce air pollution. The first in the series is "Marathon a month" which calls on people to pledge to leave their car behind and use alternative forms of transport for at least the distance of a marathon (42km/26 miles) during one month. These three news just cover the last month of activities of WHO on air pollution. The all range of work by WHO on air pollution, coordinated in a Global Platform on Air Quality and Health, covers many other aspects relevant for public health.

WHO is working to strengthen the health sector capacity to inform and protect against the adverse effects of air pollution and also provide tools to assess the health impacts of sectoral policies to engage with other sectors. In order to bring together global, national and local partners to share knowledge and take action for cleaner air and better health globally, WHO is organizing the first Global Conference on Air Pollution and Health, in collaboration with other UN agencies and international institutions (UNEP, WMO, CCAC, UNFCCC and UNECE) (http://www.who.int/airpollution/events/conference/en/). The conference is a chance to update the evidence on the health impacts of air pollution, the methods of monitoring pollution and health exposures, and tools for evaluating and implementing effective interventions. Air pollution is hitting the headlines in many different places and remains high on WHO agenda, and hopefully we will see improvements soon in areas of the planet that suffer for the consequences of bad air quality.

Commentary
Continuous air monitoring results laid bare: do we know what we know?

Patricia Forbes
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http://dx.doi.org/10.17159/2410-972X/2018/v28n1a3

As an analytical chemist based in academia, I am cautious about air monitoring results. Every analysis has some degree of uncertainty, both with respect to identification of the compounds present, as well as their concentrations. The reality is that even certified reference materials (such as those supplied by the US National Institute of Standards and Technology, NIST) which have been analysed rigorously by many laboratories using state-of-the-art equipment, require updating of their certificates of analysis ("accurate" concentrations) over time as technology progresses, resulting in lowered detection limits or improved precision. Many professionals working in the environmental sciences, however, take analytical results generated for them or by them, as absolute. Treating chemical analyses as a "black box" can lead to incorrect conclusions, unsuitable mitigation measures or management options…and ultimately, the environment which we are trying to protect may suffer.

A recent article which I read in a Royal Society of Chemistry publication (Steinmark, 2017) prompted me to write this commentary. It describes an incident that occurred on a Sunday afternoon at Birling Gap on the south coast of England in August last year. An acrid smelling haze suddenly appeared, people began vomiting and their eyes were streaming. Panic ensued; the beach was evacuated, people were hospitalised, some began driving to get away from the area and others were warned to stay inside. The thing is that the source and composition of this air pollution has still not been established, even though there were two operational continuous air monitoring stations in the area. The wind direction and back trajectory calculations ruled out some initial theories of emissions from potential sources such as adjacent countries and a sunken World War I ship. The monitoring data showed an apparent four times increase in ozone concentration during the incident, but the level that the ozone increased to was still only moderate, and importantly the health effects experienced were not consistent with the respiratory impacts associated with ozone.

The problem is that many volatile organic compounds can also be detected by ozone sensors. This cross-sensitivity arises from the fact that typical ozone monitors are based on a spectroscopic measurement and hundreds of organic compounds, in addition to ozone, absorb UV light. The ozone sensor compares absorbance at 254 nm in the air sample to that of the air sample after it has passed through a scrubber to remove ozone. The difference in absorption between the two samples is proportional to the ozone concentration. Organic compounds present in the air may be scrubbed along with ozone, and also absorb light from the source. This gives an overestimation of the concentration of ozone present in the air and a possible misidentification of the air pollutant(s) present in the sample.

In the end, the UK authorities stated that the most plausible theory for what happened at Birling Gap was that the source of the emission was a passing ship transporting organic compounds. This incident prompted UK scientists to highlight the complexity of determining specific compounds present in the air at any point in time: a complexity which cannot be fully addressed by standard routine continuous monitoring technologies. Monitoring results from a continuous time-of-flight mass spectrometer would have been needed at the time of the incident to assist with compound elucidation.

In South Africa, we are fortunate to have a network of continuous air monitoring stations, which has allowed us to better determine our air quality and to establish trends in this regard over time. The incident in the UK last year, however, highlights the importance of fully understanding the scientific principles on which our monitoring technologies are based, as well as their inherent limitations, when drawing conclusions from the data which they generate.

Reference

Steinmark, I.E. 2017, *Education in Chemistry*, 54 (6), 5.

Research brief
Estimates of CO_2 fluxes for Cape Town

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http://dx.doi.org/10.17159/2410-972X/2018/v28n1a4

Nickless et al. (2018) recently provided the results of an atmospheric inversion carried out for the city of Cape Town with the objective of obtaining estimates of weekly CO_2 fluxes at a spatial resolution of 1 km × 1 km. This approach incorporates the best information available on what the fluxes are believed to be from anthropogenic and natural sources, together with estimates of the uncertainty around these estimates, and uses measurements of CO_2 concentrations to improve on these estimates. CO_2 concentrations were measured, by means of Picarro Cavity Ring-down Spectroscopy (CRDS) analysers, from March 2012 until July 2013 at Robben Island and Hangklip lighthouses. These measurements allow the inversion to correct the prior estimates of the fluxes. The CO_2 fluxes can be converted into CO_2 concentrations by means of an atmospheric transport model – the inversion attempts to improve these modelled concentrations. Measurements at the Cape Point Global Atmospheric Watch Station were used to estimate the background CO_2 concentration.

The atmospheric transport model makes it possible to create a sensitivity matrix which converts the fluxes into concentrations at a location at a particular point in time. For this study a Lagrangian Particle Dispersion Model was used as the transport model, driven by climate inputs from the CCAM (Conformal Cubic Atmospheric Model) regional climate model in stretched grid mode (Engelbrecht et al 2013), zoomed in over Cape Town.

The prior information is key to performing a successful atmospheric inversion. To obtain prior estimates of the anthropogenic emissions, a spatially and temporally disaggregated inventory analysis was performed for Cape Town (Nickless et al. 2015). For each grid cell in our domain of interest, the direct emissions of anthropogenic CO_2 needed to be estimated. The sources considered were road vehicle emissions, domestic emissions from lighting and heating (not indirect emissions from electricity usage), airport and harbour emissions, and industrial point sources (which included power stations located within the domain). Emission factors reported by the IPCC and DEFRA, UK, were used to convert fuel use and other activity data into emissions. Error propagation techniques were used to obtain uncertainty estimates, and these ranged between 6.7% to 71.7% of fossil fuel flux estimate.

Natural fluxes over land were obtained from a land-atmosphere exchange model – CABLE (Community Atmosphere Biosphere Land Exchange model). This model can be used to calculate the fluxes of momentum, energy, water and carbon between the land surface and the atmosphere, based on process models of the major biogeochemical cycles of the land ecosystem. CABLE was dynamically coupled to CCAM, which meant that feedback between the climate and the biosphere could be explicitly modelled, instead of the land-atmosphere exchange model only reacting to the climate inputs. The net primary productivity was used as an estimate of the uncertainty around the net ecosystem exchange flux, therefore these uncertainty estimates were large relative to the nett natural fluxes. This meant that the atmospheric inversion was able make relatively larger corrections to the natural fluxes than to the anthropogenic fluxes. Ocean fluxes were obtained from Gregor et al (2013).

At the pixel level, the greatest uncertainty reduction achieved by the inversion of between 70 to 80% was made to the natural fluxes located near the Cape Town central business district and to fluxes from pixels located closest to the Robben Island measurement site.

Under the current inversion framework, the individual natural and anthropogenic fluxes are less reliably corrected by the inversion compared with the aggregation of these two fluxes. Under the prior flux estimates, the domain of the inversion over the measurement period from March 2012 until June 2013 was estimated to be a large sink of CO_2, but once the inversion was performed, the aggregated posterior flux indicated that it was closer to CO_2 neutral, with an overall uncertainty reduction achieved by the inversion of 25.6%. The results of the inversion indicated that the uptake of CO_2 by vegetation was overestimated by CABLE.

The limitation of the current framework is the large reliance of the inversion solution on the uncertainty estimates of the natural fluxes. By improving estimates of natural fluxes and reducing the uncertainties around these estimates, we can allow the inversion to make better corrections to the fossil fuel fluxes. We are exploring ways to improve the way natural fluxes are specified in the inversion to take advantage of knowledge associated with these fluxes. This is necessary if we wish to use the atmospheric inversion approach for monitoring, reporting and verification (MRV) of CO_2 emissions from a city. For example, if we wish to determine if climate change mitigation efforts are having the desired effect.

References

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Research brief
Leaf uptake of mercury lowers global air pollution
Cape Point GAW Station's research published in Nature Geoscience Journal

Casper Labuschagne
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http://dx.doi.org/10.17159/2410-972X/2018/v28n1a5

A new study, spearheaded by researchers from the University Grenoble Alpes, and international collaborators which included the South African Weather Service's Cape Point GAW Station, shows that the atmospheric pollutant mercury shows similar seasonality as the greenhouse gas CO_2. Atmospheric CO_2 levels fluctuate seasonally as vegetation absorbs the gas through leaves to produce biomass. Consequently, CO_2 levels are lower during summer compared to winter. By comparing mercury observations at 50 forested, marine, and urban monitoring stations, the study published in the highly esteemed Nature Geoscience (March 26, 2018) finds that vegetation uptake of mercury is important at the global scale. The researchers estimate that the biological mercury pump annually sequesters half of all global anthropogenic mercury emissions.

Annually industrial activities emit between two and three thousand metric tons of mercury into the atmosphere. Having a long atmospheric lifetime of about 6 months, mercury emissions are able to spread across the globe. However, this compound does not remain airborne indefinitely. It has been established that atmospheric mercury deposition is predominantly by rainfall and snowfall, and monitoring networks measure mercury wet deposition worldwide. A slowly increasing number of experimental, field and modeling studies has suggested

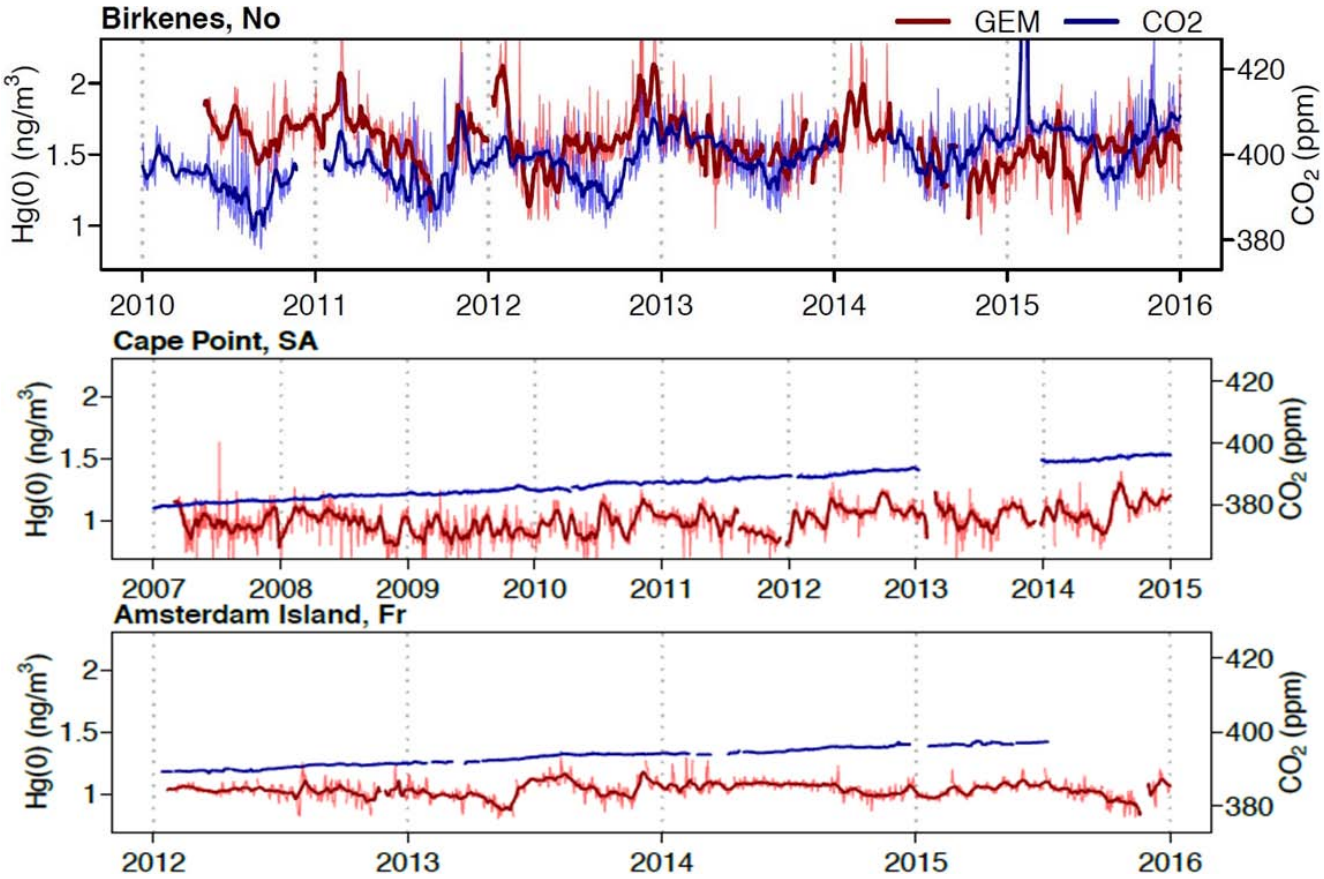

Figure 1: *Gaseous elemental mercury (GEM, red) follows seasonal CO_2 (blue) variations at the forested site of Birkenes (Norway), reflecting leaf uptake of atmospheric mercury. The effect is totally absent at the oceanic sites of Amsterdam Island in the Indian Ocean and the Cape Point GAW station. The study finds leaf uptake of mercury to be a globally important pathway of atmospheric mercury deposition.*

that plant leaves can also directly take up gaseous elemental mercury from the atmosphere. During autumn (otherwise known as fall), leaf mercury is then transferred to the underlying soil system by leaf senescence. Until now, the importance of this alternative deposition pathway, at the global scale, has never been fully appreciated.

To understand whether leaf uptake of atmospheric mercury is important on the global scale, Lead authors Martin Jiskra and Jeroen Sonke, from the Géosciences Environnement Toulouse laboratory (CNRS / UPS / IRD / CNES), teamed up with scientists who monitor atmospheric mercury and CO_2 levels across our planet. CO_2 has a well-known seasonality with concentration minima in late summer, at the end of the vegetation and leaf growth season, and higher levels during winter. To their surprise, the researchers found that mercury and CO_2 show similar seasonal variations at five forested monitoring stations in the Northern hemisphere (see Figure). Observations of mercury and CO_2 made at Amsterdam Island and Cape Point, both coastal sites, turned out to be key in identifying the role of vegetation. At the Cape Point GAW station, operated by the South African Weather Service, both mercury and CO_2 show near-zero seasonal variations (see Figure).

Consequently, the researchers turned to atmospheric monitoring databases from EMEP, AMNet and CAMnet, examining seasonal mercury observations for another 43 sites globally, but for which CO_2 observations were lacking. They found that the amplitude of seasonal atmospheric mercury variations is largest at inland monitoring sites away from the coast. At all of the terrestrial sites they found strong inverse correlations between satellite observed photosynthetic activity and mercury concentrations. At urban monitoring stations the correlations were absent, and mercury seasonality was essentially controlled by local anthropogenic mercury emissions. The researchers conclude that vegetation acts as a biological pump for atmospheric mercury and plays a dominant role in the observed atmospheric mercury seasonality. By comparing the 20% amplitude of seasonal mercury variations to the known amount of mercury in the atmosphere (~5000 metric tons), they estimate that each year about 1000 tons of mercury is sequestrated in vegetation via leaf uptake. This amount is equal to half the annual global anthropogenic mercury emissions. They also suggest that the documented 30% increase in global primary productivity over the 20th century has likely enhanced uptake of atmospheric mercury, thereby practically offsetting increasing mercury emissions. Although leaf uptake removes mercury from air, autumn litterfall transfers the sequestered mercury to soils. Soil mercury ultimately runs off into aquatic ecosystems including lakes and Oceans where the mercury bioaccumulates to toxic levels in fish.

Reference

Jiska M, Sonke J, Obrist D. et al. 2018. A vegetation control on seasonal seasonal variations in global atmospheric mercury.

Nature Geosciences, 11, 244-250. DOI: 10.1038/s41561-018-0078-8.

Company Profile

an EQH company

Originally formed in 1976, SI Analytics provides air monitoring solutions to industry, government and research organisations. Our analytical instrumentation offers continuous measurement of both surrounding air pollution and chimney emissions.

We design, manufacture, supply, install, commission, train and provide after sales service for either individual monitors, or integrated systems in southern Africa.

SI Analytics is currently a B-BBEE level 2 contributor.

Being a founding member of Europa Environmental gives us the expertise to supply and support advanced environmental monitoring technology.

Our aim is to bring you the world's best instrumentation, spare parts, service, technical support and training - at the most affordable pricing - to meet your environmental monitoring needs.

Services

Due to government regulatory requirements, our clients are under increasing pressure to collect emissions data. This is not part of their core business - their expertise lies in manufacturing the products they sell. Our expertise lies in assisting them to manage their data and compliance issues.

We offer a wide range of services such as ambient air quality and continuous emissions monitoring which include:

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In Memoriam
Professor George Djolov

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It is with deep regret that we announce the unexpected passing of a colleague and friend, Professor George Djolov.

Born in Bulgaria but travelling and studying extensively, George's life was characterised by a diverse and distinguished career in academia. For his studies in Canada and Russia, George received two doctoral degrees, his first in Physics and Mathematics, and his second in Mechanical Engineering. In the period leading up to 1990, he served as professor of Physics in the Bulgarian Academy of Sciences, founded the Institute of Ecology for the Bulgarian Academy of Sciences, served as Senior Advisor on Environmental Problems to the Bulgarian State President's Office, and was President of the WFEO's Committee on Engineering and the Environment (1990-1995). During this period George also had a close working relationship with the Energy and Climate Task Force of the International Institute for Applied Systems Analysis (IIASA). He was a member of the Editorial Board of the International Journal "Urban Climate" from 2012 – 2018.

An extraordinary period followed in Southern Africa. George first served as co-ordinator of the BTec Programme in Applied Physics at the University of Zimbabwe. In 1996 he was appointed as Professor and Chair of Physics, and thereafter Dean of the School of Mathematics and Natural Sciences at the University of Venda. In 2003, the University of the North (now Limpopo) appointed George as Professor and Director of the Faculty of Physical and Mineral Sciences. For two years thereafter, ending in 2006, he served as CEO of the National Community Water and Sanitation Training Institute in Polokwane (then Pietersburg). During this time, he co-supervised a Ph-D student at the North-West University and was appointed an extraordinary professor at the North-West University from 2004 -2007.

After his formal retirement, Professor George Djolov joined the University of Pretoria as a meteorologist and Extraordinary Professor in the Department of Geography, Geoinformatics and Meteorology in 2007. George continued to be highly active over the following decade. He taught each year into the meteorology programme and published and supervised postgraduate students in his favoured fields of meteorology, namely theoretical and numerical geophysical hydrodynamics, atmospheric physics and boundary layer dynamics, climate change, air pollution modelling and environmental physics. During his time in Africa he raised in excess of R17 million in research grants, co-authored and/or published four book and in excess of 100 journal articles. In 2015, the International Eurasian Academy of Sciences honoured George for his distinguished career and contribution to science.

In his final three years he managed the University of Pretoria's Laboratory for Atmospherics Studies. While liaising with industry, his primary role supported research and student supervision in the field of air quality management. George was more than a colleague at the University of Pretoria. He was a statesman for academia and for many also a friend and confidant. Always humble, he established an astounding rapport with his students and colleagues, and will be severely missed by all.

Originally published by the University of Pretoria, this piece has been edited and some content added by Professor JJ Pienaar, School for Physical and Chemical Sciences, North-West University.

In Memoriam
Johan van Heerden

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Johan van Heerden was born in Pretoria on the 16th of September and was the eldest of three children. He matriculated from Afrikaans Boys High in 1959 and attained the National Diploma for Meteorological Technicians. Between 1964 and 1967, Johan was seconded to the School of Technical Education. He was promoted to Lieutenant at this unit and during this time he married Sophie Pelser. Between the years of 1967¬ and 1972 he was a weather forecaster at various forecasting stations. In 1973, he relocated with his family to Pretoria after which he earned the following degrees: BSc, B. (Hons) and M. He was the first person to be awarded DSc (Meteorology) at the University of Pretoria (1984).

In 1976, he was appointed as Meteorologist and did research for the development of the Numerical Forecasting Unit and the Satellite Meteorological Group. In 1979, he became Head Meteorologist (Numerical Forecasting) and part-time lecturer in Dynamic Meteorology at the University of Pretoria. He represented the South African Weather Service (SAWS) at various local and international conferences and has published widely. Under his leadership, highly specialized satellite imaging and graphic systems were acquired for the SAWS.

In short, he was one of the first researchers who became an expert in the use of satellite data for weather forecasting. As a result, in 1980 he was invited to join the International Satellite Cloud Climatology Organization where he developed a system to archive Meteosat data for calibration purposes. In that year, he also published the first book about weather forecasting in the Afrikaans language and for this he was honoured by the ATKV (The Afrikaans Language and Culture Association). In 1984, he was promoted to Vice-Director for Research and he initiated and led research projects regarding long-term weather forecasting in South Africa.

In 1989, he became Professor and was the head of the Department of Meteorology at the University of Pretoria. This is, and has always been the only institution that offers a degree course in Meteorology in South Africa. During this time, he got more and more involved in field projects where, for example, he placed rainfall stations in the veldt in order to establish a comparison between satellite rainfall and measured rainfall. During this time he also started his endeavors in cloud and fog harvesting. This activity continued long after his retirement in 1999. In 2016, he participated in the research team that was tasked to examine the feasibility of cloud harvesting on the iconic Table Mountain in Cape Town, a very topical issue today. After Johan's retirement he stayed closely involved with his alma mater. He still supervised postgraduate students and took part in research projects. His contribution was mainly with BSc (Hons) level where he lectured in classical forecasting techniques and he was able to share his wealth of experience. Johan educated more than one generation of meteorologists and was mentor and friend for more than 30 postgraduate students.

During his illustrious career he visited many countries for research purposes and conferences and among them were Russia, Brazil, Chile, Australia and Antarctica. He was a member of 17 national and international research committees and he was president of the South African Society for Atmospheric Sciences (SASAS) for 6 years. He was also a member of the South African Scientific Committee for Antarctic Research and he was the representative for South Africa at the International Association of Meteorology and Atmospheric Sciences (IAMAS).

Apart from his pioneering research on the use of satellite images for weather forecasting, he was also one of the first scientists who noticed a connection between the El Nino phenomenon and South African rainfall. In his most recent research, not only did he focus on cloud harvesting but, with Ben van Rensburg, Bobby McEwan and Jack Mellet, they also designed and built a unique cloud harvesting system. During his retirement in Plettenberg Bay, Western Cape he was very involved in the local community and with his friend Paul Pretorius endeavored to make the Poortjies area safer with a camera security system. And finally, last but not least, Johan had the ability to inspire those around him and to cement eternal friendships. He will be missed.

Originally written by his daughter, Esbie van Heerden and published by the University of Pretoria and the South African Society for Atmospheric Science, this version has undergone minor editing for content.

Research article

The economics of flue gas cooling technology for coal-fired power stations with flue gas desulfurisation

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Abstract

Developments in heat exchanger technology, specifically in the use of polymers as tube material, have allowed the use of gas to water heat exchangers under conditions previously not viable. Two applications in the flue gas cleaning circuit of coal-fired power stations are described in this paper. In conventional pulverised coal-fired boilers, cooling of gas prior to the wet flue gas desulfurisation (WFGD) absorber reduces water consumption for evaporative cooling of the flue gas and can recover heat for feed water preheating or for use elsewhere in the plant. In another application, circulating fluidised bed boilers, which are currently proposed for a few independent power producers and may not require wet FGD, heat recovery is still feasible upstream of the bag filter typically used for particulate emission control. The extracted heat can again be recovered for use in other power plant processes, in this case most economically for pre-heating combustion air. This paper presents case studies for each of the above applications, showing that the power station efficiency is typically increased by approximately 1% of its pre-installation value. An economic analysis is provided for each, including additional power sales, reduced water consumption, or reduced fuel use with a reduction in carbon tax. For the larger installations with WFGD, payback time can be in the order of 6 years.

Keywords

flue gas desulfurisation, boiler efficiency improvement, carbon tax

Note: ***In view of an SI decision in this regard, we have used sulfur and desulfurization as the correct spelling for these terms, rather than the dictionary spelling.***

Introduction

In arid countries such as South Africa the reduction of water use in industry is especially important. Significant water savings have already been achieved by South African coal-fired power plants by using dry or air-cooled condensers instead of the conventional use of water cooling towers (Lennon 2011).

The emission requirements set for large coal-fired power stations under section 21 of the Air Quality Act (39/2004) (DEA 2014, DEA 2010) will require flue gas desulfurisation (FGD) for these stations from 2020 onwards, except where postponement has been granted. The present most economical option in South Africa is a wet absorption system using limestone installed downstream of the particulate bag filter (wet flue gas desulfurisation or WFGD). This imposes an additional water use for evaporative cooling of the absorber inlet gas and negates some of the savings achieved by dry cooling technology. However, if the inlet gas is cooled before entry into the WFGD by an external heat exchanger, and the heat is used elsewhere in the power circuit, there is an obvious benefit in water savings. Developments in materials of construction have allowed this possibility despite the corrosive nature of the gas.

Thermodynamically, the circuit efficiency improves when heat is recovered within the generating cycle, which leads to increased power sales or reduced fuel use. In the latter case, the emissions are also reduced. Even in power generation applications where WFGD is not practised, such as in circulating fluidised bed (CFB) boilers (with the addition of limestone directly to the boiler), the recovery of heat from the flue gas and the use thereof in other parts of the circuit may lead to similar benefits for these (usually smaller) applications.

These heat exchangers must be designed to cope with the extremely corrosive conditions encountered when the flue gas temperature approaches dew point, and therefore fluoropolymers are used for construction. This paper describes the benefits and cost impact of such heat transfer installations using a large WFGD and a smaller dry CFB application as case studies.

Technology description

Wet flue gas desulfurisation (WFGD)

In conventional WFGD systems, approximately 85% of the water used (~ 0.22 l/kWh) is required for quenching or cooling down the flue gas prior to, or on entry into, the absorber as indicated in Figure 1 below. The flue gas temperature is typically reduced from 150 °C to approximately 90 °C. If this temperature reduction is achieved by external heat exchange, the recovered energy could be used elsewhere in the process cycle, e.g. for absorber exit gas re-heating or for feed water pre-heating, and the following benefits could be achieved:

- an increase in power generation cycle efficiency, with a concomitant increase in electrical output or reduction in fuel use for constant output; or
- a reduction in carbon dioxide emissions by decreased fuel use per unit output concomitant with the increase in efficiency; and
- water savings as indicated above, which is especially important in arid countries such as South Africa.

External heat exchange of WFGD absorber inlet gas can be directly performed by the installation of an additional feedwater preheater in the absorber inlet ducting, but in the case of retrofitting this may entail considerable re-routing of feedwater piping. A more economical option for both feedwater heating and flue gas re-heating is to have a secondary water circuit removing the energy from the absorber inlet gas stream to either the feed water or the treated flue gas. In South Africa, where absorber outlet gas re-heating is less likely to be required than in countries with lower ambient temperatures or higher humidity, the feedwater heating option is more likely to be considered. A diagram for this layout is given in figure 2 below. Such equipment has been proven in commercial operation in Europe for more than 20 years with more than 20 units installed on power plants ranging in size from 400 MWe to 800 MWe, operating on both brown and hard coal.

Made of fluoropolymer, the mechanical design of the heat exchanger tubes compensates for the low thermal conductivity of the material, while avoiding the corrosion potential inherent in the high sulfur dioxide (SO_2) and sulfur trioxide (SO_3) contents of the gas. The design also allows for frequent online washing to reduce fouling by particulate matter. High temperature damage can be avoided by an emergency spraying system triggered by excessive temperature. The design also allows quick installation, typically during a normal boiler turnaround.

Systems without WFGD – Circulating fluidised bed

Circulating fluidised bed (CFB) boilers have rapidly evolved since the 1980s and are now available up to 800 MWe and with supercritical steam pressure, so that circuit efficiencies similar to those of pulverised coal (PC) power stations at 40% are achieved (Utt et al. 2011, Gauvillé et al. 2011). The ability of CFB plants to utilise a wide variety of solid fuels, including coal discard, and the direct capture of sulfur by the direct addition of limestone offers further benefits. A few such units have been proposed for local installation.

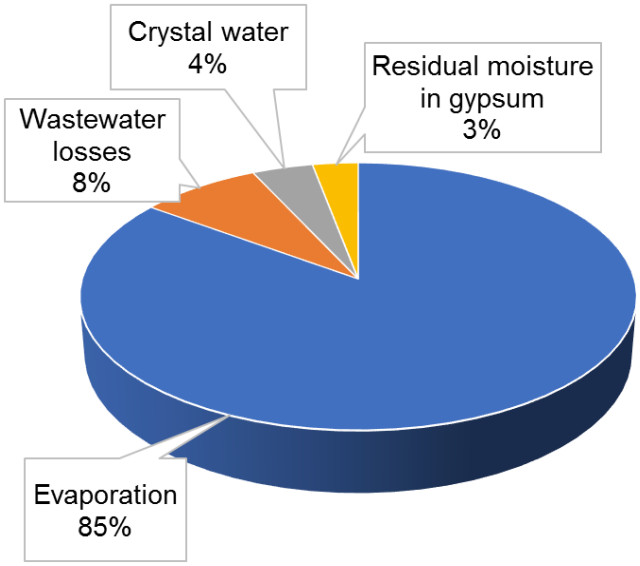

Figure 1: *Water use in WFGD system (Arif et al. 2015).*

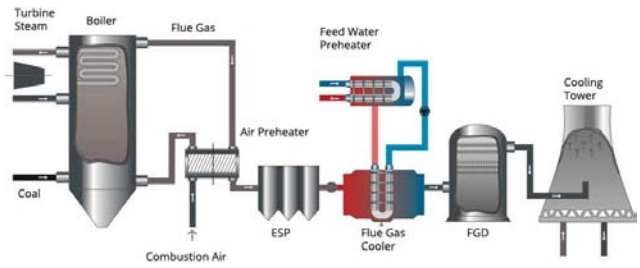

Figure 2: *Typical flow chart for a heat recovery system with feed water preheating (Wallstein 2015)*

Heat recovery in CFB systems is possible downstream of the bag filter usually used for fly ash collection. The temperature in the bag filter must be kept well above the dew point to prevent damage to the bags and the structure. However, the fluoropolymer construction of the heat recovery exchangers after the filter allows lower temperatures to be used there. Heat transfer then takes places via an intermediate closed loop (as for the WFGD) to the combustion air (both primary and secondary) of the fluidised bed, or to the feedwater. Such a flue gas heat recovery system will for example be installed at the 299MWe Teesside power station in the United Kingdom, which is the world's largest CFB utilising only biomass as fuel.

Economic analysis

Cost components to be considered include the capital and installation costs, as well as the running cost during the lifetime of the system. The latter consists of the power required to overcome the additional pressure drop in the flue gas ducting, the power required for pumping of the intermediate circulating heat transfer fluid (water) and the disposal of the washing liquid used to clean the tube bundles.

The capital cost for the heat exchangers and ancillary equipment for both case studies was taken from actual quotes and the estimates therefore have a high level of confidence. The confidence level of the installation cost is dealt with below.

Credit components that must be considered are the increased sales due to the increased efficiency at constant fuel use, or reduced fuel cost (if output is kept constant), and in all cases the reduction in water requirements for evaporative cooling in processes prior to the absorber. As it is highly probable that carbon tax will be introduced before commissioning of any of the projects discussed in this paper (South African Treasury 2017), the reduction in carbon tax associated with the reduced fuel use must be considered. The analysis carried out in this paper assumes similar inflation rates for costs and for additional income. History has shown that electricity prices tend to increase at a rate higher than general inflation and in that sense the analysis can be considered to be conservative.

A similar economic analysis for the Medupi dry cooled pulverised fuel station was previously presented by Stephen et al. (2016). However, Stephen et al. did not consider carbon tax. In addition, in the analysis presented here, a different Lang factor was considered. The Lang factor is defined by

$$\mathrm{TPC} = f\ x\ \mathrm{TEC}$$

where TPC is the total plant cost, TEC the major capital equipment cost and f a multiplier known as the Lang factor, after Hans J Lang who first introduced it in the 1940s (Wain 2014). The multiplier obviously depends on the type of plant and the location, while it also includes the labour cost of the installation. Stephen et al. (2016) used an f-value of 4.10 that is commonly assigned to a liquid handling plant. However, this value was considered too conservative due to the simplicity of the heat exchanger installation (it has only two major plant items). Wallstein (2015) indicates a factor of 1.425 for a retrofit on a 550 MWe PC boiler in the USA. Taking into consideration the perceived lower labour productivity in South Africa, a separate installation cost estimate using methodology developed and tested for local conditions was made and used to calculate a local Lang factor of 2.10.

Retrofit on a major South African pulverised coal-fired power station

Four scenarios are considered here:

- A retrofit during a scheduled boiler overhaul/inspection, assuming that the six-week period is sufficient for the installation and with the published Lang factor of 1.425
- A retrofit during a scheduled boiler overhaul/inspection, assuming that the six-week period is sufficient for the installation and using the local Lang factor derived above
- A retrofit during a scheduled boiler overhaul/inspection, with an additional two weeks allowed for the installation and considering a loss in electricity sales during this period and the 1.425 Lang factor
- As for c, but with a Lang factor of 2.1.

For each of these cases, savings can accrue either by selling more power using the same amount of fuel, or by keeping the output constant and reducing the carbon emission from the reduced fuel use.

The capital cost of the major plant items (mainly the heat exchangers and casings) is approximately R 113 million (Wallstein 2015) at an exchange rate of R14.5 to the € (as in February 2018), the rate used in all the calculations.

For all scenarios, the same technical assumptions given in table 1 below are made. In all cases, the annual maintenance time was taken as 42 days and availability during run-time as 99.6%.

Table 1: *Technical assumptions for WFGD*

Design electrical output	**MW**	**800**
Increase in efficiency[1,2]	%	40.0 to 40.53
Water savings[2]	l/kWh	0.21 to 0.15
Additional pressure drop (flue gas side)[1]	mbar	0.38
Additional fan power[1]	kW	375
Liquid pumping power[1]	kW	125
Water cost[2]	R/ m^3	15.7
Carbon tax[3]	R/ton CO_2	120
Carbon emissions[4]	ton/MWh	0.9
Power income[2]	R/kWh	0.42
Coal cost[5]	R/kWh	0.26

Notes:
[1] Wallstein 2015
[2] Stephen et al. 2016; water use is reduced from the existing 0.21 l/kWh to 0.15 l/kWh
[3] South African Treasury 2017
[4] Assumed 10% lower than Eskom average (Eskom 2015)
[5] Eskom 2016 (pre-retrofit figure)

For each of these scenarios, the payback period and the yearly Return on Investment (ROI as % over 50 years) were calculated as given in Table 2 below (Furey 2017).

Operating costs, i.e. sodium hydroxide for circuit water treatment, the cost of wash water for the bundles, and 1% per year of capital equipment cost were included in the annual cost calculations. The fraction of the additional income due to each of the savings components is given in Table 3 below.

Table 2: *Economic metrics for WFGD scenarios (R14.5=1€)*

Scenario	**a**	**b**	**c**	**d**
Payback period (years)				
Additional electricity sales	4.5	6.6	7.6	9.8
Coal savings	5.1	7.5	8.7	11
Return on Investment (ROI) (%)				
Additional electricity sales	21	14	13	10
Coal savings	19	13	12	9

The results in Table 2 are obviously sensitive to the exchange rate and to the assumed lifetime of the plant. The latter does not have a major impact; at the discount rates under discussion, the present value factors differ very little between 40 years and 50

years. As an example, at 14.5% hurdle rate the cumulative cash flow factors are 6.40 and 6.41 times the annual flow. An 'better' exchange rate obviously improves the payback period and the ROI; Table 4 provides comparative figures for the economic metrics at an exchange rate of R16.5 to the € as seen in October of 2017.

Table 3: *Components of additional income*

	Water savings	Coal savings	Carbon tax	Add. sales
Increased power	15%	0%	0%	85%
Constant power	17%	59%	24%	0%

It is evident from Tables 2 and 4 that an attractive rate of return, exceeding the 14.5% given as the hurdle rate for Eskom by Stephen et al. (2016), can be achieved even for some of the cases where 2 additional weeks of power sales is lost during installation, given the 40 or 50-year lifetime of the large SA coal-fired power stations.

Table 4: *Economic metrics for the WFGD scenario (R16.5=1€)*

Scenario	a	b	c	d
Payback period (years)				
Additional electricity sales	5.1	7.6	8.3	10.7
Coal savings	5.8	8.6	9.4	12
Return on Investment (ROI) (%)				
Additional electricity sales	19	13	12	9
Coal savings	17	11	11	8

Finally, the reduction in greenhouse gas emissions for the constant output case over a 50-year lifetime is in the order of 3.5 million ton; the income per ton of carbon dioxide saved is in the order of R 456 per ton – this is therefore an opportunity to reduce emissions at a profit. The figure compares favourably with similar figures given in the South African Long-Term Management Strategy (ERC 2007) and figures given by McKinsey and Co (2011) for global opportunities.

Original installation in a CFB station

For this case study, the intermediate liquid loop taking heat from the flue gas downstream of the final bag filter and heating the primary and secondary air feed to the fluidised bed has been used. The installation is assumed to occur during the construction of the plant so that no sales are lost during the installation and the sensitivity analysis is only done for the Lang factors of 1.425 and 2.1 as above.

The capital cost of the major heat exchange plant items for a 300 MWe boiler is R 96 million. The assumptions for this case are given in table 5 below. Results indicated that additional income is obtained from additional electricity sales (80%) and carbon tax benefit (20%). Payback periods and ROI values for the two Lang factors are given in table 6.

Clearly this application will require careful analysis and a more detailed design before an investment decision can be made. Particularly the case for fuel saving requires further consideration, as the rationale for the use of CFBs includes the availability of previously unusable fuel sources for which the cost may well be lower than the figure used here.

Table 5: *Technical assumptions for CFB*

Design electrical output	MW	300
Increase in efficiency[1]	%	38.0 - 38.42
Additional pressure drop (flue gas side)[2]	mbar	0.38
Additional fan power[2]	kW	197
Liquid pumping power[2]	kW	110
Carbon tax	R/ton CO_2	120
Carbon emissions[3]	ton/MWh	1.05
Power income[4]	R/kWh	0.42
Fuel cost[5]	R/kWh	0.26

Notes:
[1] Lockwood 2013
[2] Provided by Wallstein GmbH
[3] Assumed to be 5% higher than Eskom average
[4] Stephen et al. 2016[5] Eskom 2016 (pre-retrofit figure)
[5] Eskom 2016 (pre-retrofit figure)

Table 6: *Economic metrics for CFB scenarios*

Lang factor	1.425	2.1
Additional electricity sales		
Payback time (years)	8.6	12.6
ROI (%)	11.6	7.8
Fuel savings		
Payback time (years)	8.8	12.9
ROI (%)	11.2	7.7

Conclusions

Two options for flue gas cooling downstream of the dust filter with concomitant heat recovery have been technically and economically considered:

- Feed water pre-heating for pulverized coal-fired power stations with the typically used wet desulfurization of the flue gas
- Combustion air pre-heating for fluidised bed boiler stations with instant desulfurization.

The following benefits have been identified:

- Reduced water consumption during flue gas wet desulfurization for option 1
- Increased power plant efficiency expressed
- either as gain in power output and additional electricity sales
- or as fuel savings with decreased carbon dioxide emissions.

In case of wet FGD technology the water savings due to flue gas cooling are significant and particularly valuable for arid countries like South Africa. The calculated payback times in the range of 4-8 years have additionally confirmed the feasibility of the technology, illustrating why this technology has been applied for several decades worldwide, helping owners to increase overall energy efficiency and meet more stringent environmental standards for their plants.

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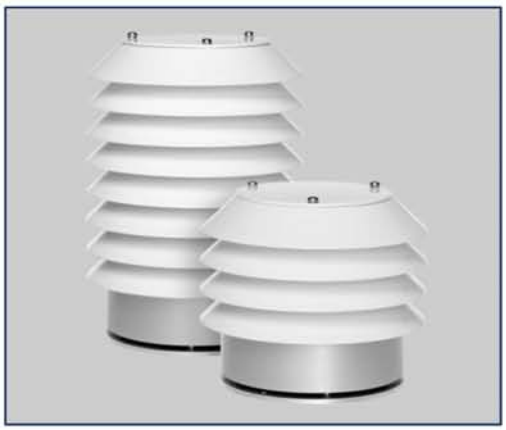

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Research article

An economic assessment of SO_2 reduction from industrial sources on the highveld of South Africa

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Abstract

The costs and benefits associated with the implementation of an SO_2 point source standard for solid fuel combustion installations (Category 1.1 sources, National Environmental Management: Air Quality Act: s21 List of Activities 2013) were evaluated to assess the desirability of implementation of the standards from an environmental as well as economic point of view. The study used a bottoms-up or impact pathway approach to analyse the impact of emission reduction. To reach the new plant (2020) SO_2 emission standard of 500 mg/Nm^3, the installation of wet flue gas desulfurisation (FGD) is the likely technology as it is a widely installed and well-developed technology. Costs and benefits associated with the installation of FGD were 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. A sensitivity analysis was conducted on the costs and benefits with the largest impact on NPV (net present value) 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 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 (value of statistical life) estimate used (high R115 billion; low R36 billion). The choice of appropriate concentration response functions and the applicability thereof in the South African context are important considerations, likely requiring further study. The capital cost of FGD installations was found to be the most significant cost and was sensitive to the evaluation method (central R187 bil; high R306 bil; low R80 bil). Failure to account for operating costs would significantly impact the economic evaluation. The results of the study indicate that, given the information currently available, it is unlikely that the benefit of reducing SO_2 emissions from existing sources to the required standard outweighs the cost of implementation on the Mpumalanga Highveld.

Keywords

cost benefit analysis, flue gas desulfurisation, Mpumalanga Highveld, sulfur dioxide

Introduction

Air quality management in South Africa has undergone significant legislative reform in recent years. 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 (MES) for industrial point sources. 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 purpose of the investigation was to assess the desirability of implementation of one of these standards from an environmental as well as economic point of view. The research assessed compliance with the Category 1.1 standards for SO_2 emissions (DEA, 2013) 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.

Methodology

The study used a bottom-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 operational at the time of study within the study area expected to have a significant impact on ambient SO_2 concentrations were included in the study.

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 were therefore used in the study. The process flow methodology is shown in Figure 1.

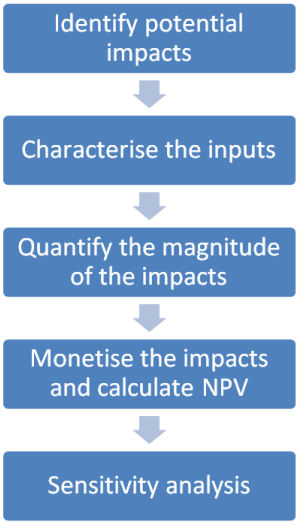

Figure 1: *Process methodology (Jalaludin, Salked, Morgan, Beer & Bin Nisar, 2009; Van Horen, 1996)*

Externalities were identified and classified as follows, based on the expected benefit and availability of information:

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.

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.

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 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.

Evaluation of Cost

Capital Cost

The costs associated with FGD include capital cost, lime or limestone costs, cost of additional water, additional electricity usage and greenhouse gas emissions.

The installation of FGD will require direct capital cost. FGD has been widely installed internationally and capital cost estimates based on actual costs are available in literature (World Bank, 1999; Orfanoudakis, Vakalis, Krallis, Hatzaipostolou & Vlachakis, 2005; Cleetus, 2012). Eskom provided reviewed cost estimates for the installation of FGD in their postponement report (Eskom, 2014), which was used as the central estimate.

The direct capital cost estimates per kW were adjusted to the base year (2020) and compared for consistency. The time-adjusted Eskom estimate is 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 as the mean 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 (2020). The overall 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 at Secunda.

Water Cost

The operation of FGD on all the facilities will require an estimated 98 million m^3 of water per annum (Sasol, 2014; Eskom, 2014). The most likely adequate source of additional water for the Vaal River system is the Lesotho Highlands Phase 2 (LHP2) Project. A study by Basson, Combrinck, Schroder and Rossouw (2010) calculated the cost of water from LHWP Phase

2, including royalties, at 6.14 R/m^3 of raw water. This cost was used to calculate the costs of additional water supply as it takes the capital cost and operating costs of providing the additional water into consideration. The estimated total cost of providing water was calculated as R 32 billion (present value 2020) over the lifetime of the facilities (up to 2050).

Lime Cost (Direct)

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). The limestone 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 limestone to R300/ton. The cost for limestone was calculated as R63 billion over the study period.

Waste Disposal Cost

It was assumed that the waste generated would be disposed of in a similar manner to fly ash currently, 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. 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).

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, assuming that current waste disposal facilities are compliant or that additional complaint facilities will be required to address the additional load. 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.

Electricity and additional CO_2 cost

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. Energy consumption for wet limestone FGD is 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 ($SO_2 + CaCO_3 \rightarrow CaSO_4 + CO_2$). 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.

Evaluation of Benefits

The expected benefits associated with the reduction of SO_2 include health benefits (mortality and morbidity), impact on ecosystems and water resources due to acid deposition, impact on vegetation through respiration, impact on structures and visibility impacts.

Impact on soils

Although acid deposition can be quantified, the impact thereof on ecosystems has not been quantitatively proven (Josipovic, Annegarn, Kneen, Pienaar & Piketh, 2011). It is possible that a reduction in deposition would result in increased biodiversity, as has been the case in the United States and Europe (NAPAP, 2005; EEA, 2014). 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.

Impact on vegetation

A number of studies (Scorgie & Thomas 2006, Josipovic, Annegarn, Kneen, Pienaar & Piketh 2009) found exceedances of the UNECE-CLRTAP Lichen standard of 10 ug/m^3 and 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. The actual impact on ecosystems cannot at this stage be accurately quantified due to a lack of quantifiable damage functions. This impact is therefore classified as a Class 2 impact.

Impacts on buildings, materials and visibility

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.

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.

Health impacts

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, 2000). Concentration

response functions that relate changes in concentration to increased risk are derived from epidemiological studies. Two types of epidemiological studies are commonly employed. Cohort studies are long term studies that follow study subjects to relate exposure status to a health outcome by means of a relative risk (RR). Confounders can be more easily controlled for; such studies are however conducted over long time periods and have significant costs. Two such large studies have been conducted, one by the American Cancer Society (ACS study) (Pope, Burnett, Thun, Calle, Krewski, Iyo & Thurston, 2002; Krewski, Jerret, Burnett, Ma, Hughes, Shi, Turner, Pope, Thurston, Calle & Thun, 2009) and the Harvard Six Cities study (Dockery, Pope, Xu, Spengler, Ware, Fay, Ferris & Speizer, 1993; Krewski, Burnett, Goldberg, Hoover, Siemiatycki, Abrahamowicz, White & Others, 2000). Both studies indicated increased mortality associated with air pollution.

Time-series studies correlate the incidence of health outcomes, such as mortality or hospital admissions with fluctuation in daily pollutant values to establish an increased percentage of adverse health outcomes for a certain change in ambient pollutant concentrations. As time-series studies are easier and less cost intensive, many more of these studies have been conducted world-wide. Several meta-analyses using data from different studies have also been conducted. Detailed analyses of these studies have been conducted as part of European and US health reviews (US EPA, 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 (Sunyer, Atkinson, Ballester, Le Terte, Ayres, Forastiere, Forsburg, Vonk, Bisanti, Anderson, Schwartz & Katsouyanni, 2002, Sunyer, Ballester, Le Terte, Atkinson, Ayres, Forastiere, Forsburg, Vonk, Bisanti, Tenias, Medina, Schwartz & Katsouyanni, 2003; Dominici, McDermot, Daniels, Zeger & Samet, 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; Ayres 2009). 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. A study by Atkinson, Mills, Walton and Anderson (2014) reviewed epidemiological time-series studies investigating the association between mortality and hospital admissions and fine particle components. 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, Demerijan, Ebi, Favarato, Greenbaum, Mehta, North, O'Keefe, Pandey, Pope, Smith, Speizer, Walsh & Zhang, 2010).

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

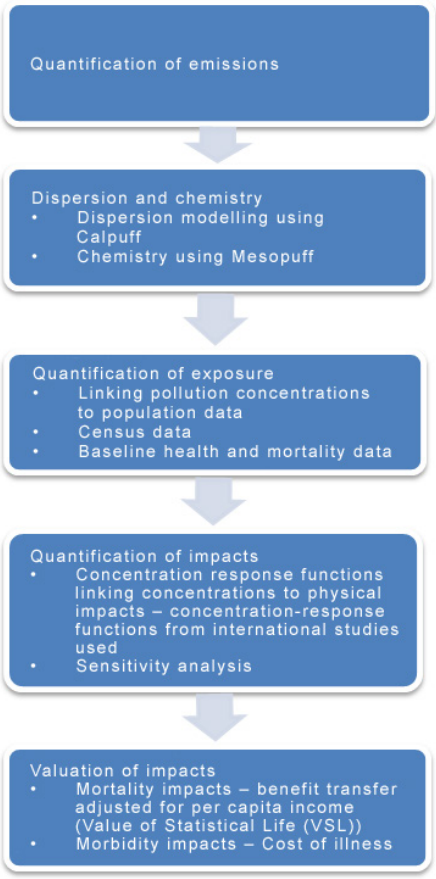

Figure 2: *Evaluation of health benefits (Based on: USEPA, 2011)*

daily concentrations of PM_{10}, SO_2 and NO_x and respiratory, cardiovascular and cerebrovascular mortality in the city of Cape Town. The study found associations between the criteria pollutants and increased mortality.

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. The largest of these studies is the ACS study (Pope et al., 2002) which included the most participants and ambient stations in the analysis (with 552 138 adults using data from 151 monitoring stations). 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 was therefore used for adult mortality calculation. As these studies were conducted in a developed country (United States), a sensitivity analysis was 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). The concentration-response function derived in the Lin, Pereira, Nishioka, Concecao, Braca and Saldiva (2004) study was used, as the study was conducted in an area with relatively higher SO_2 ambient concentrations. In order to determine the morbidity impacts, COMEAP data for SO_2 respiratory hospital admissions was used (Ayres, 1998). For morbidity impacts of sulfates, the

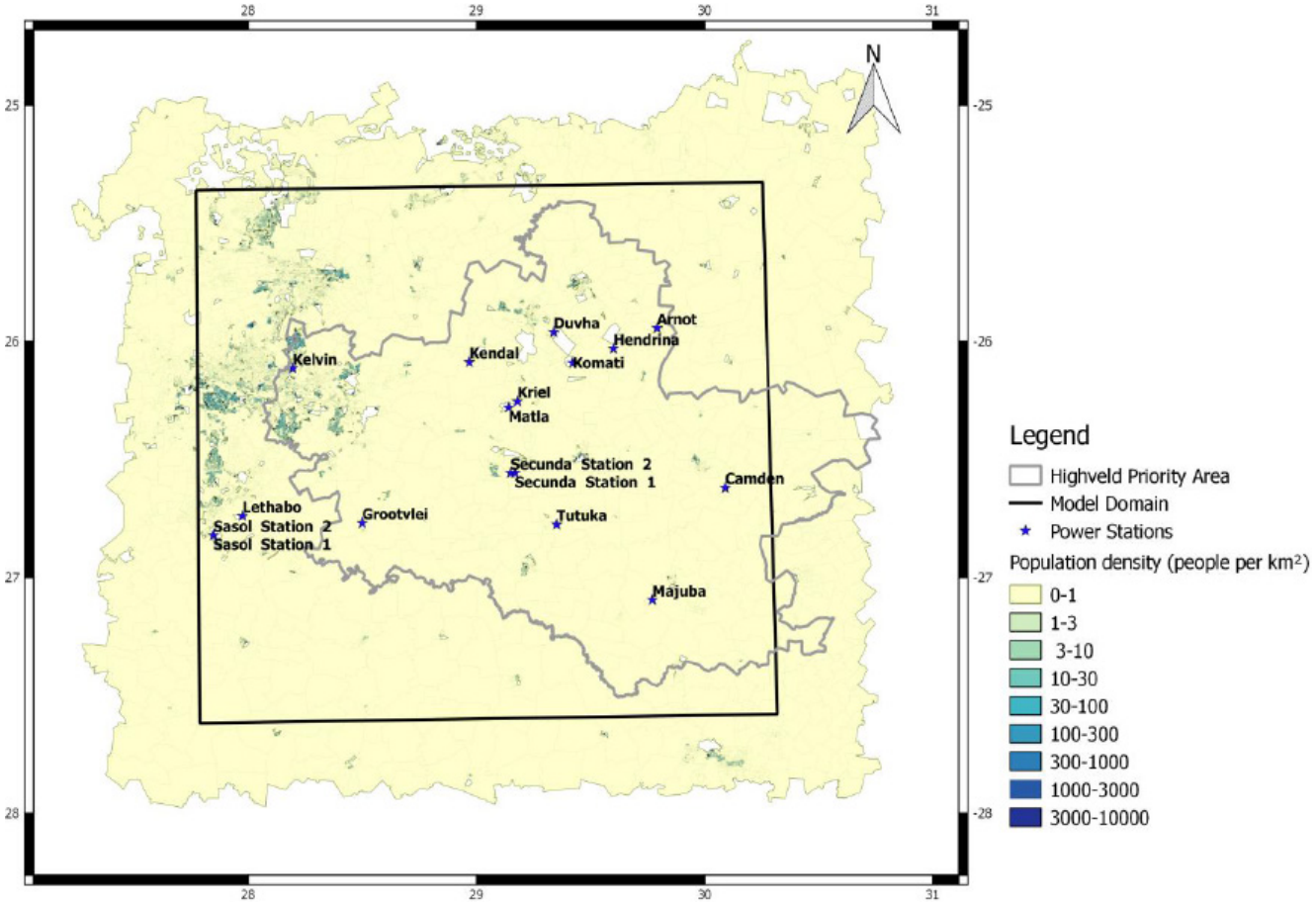

Figure 3: *Highveld priority area with major sources (Pretorius 2015).*

Table 1: *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)

Table 2: *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)

Table 3: *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

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, Bobak, Gorynski, Wotyniak, Danova, Celko, Kriz, Briggs and Elliot. (2005) was used to quantify asthma risk.

The methodology followed for calculating health benefits is shown in Figure 2. The final dose-response functions, as well as the baseline mortality and morbidity values used are given in tables 1 to 3 overleaf.

Dispersion modelling

The study area was taken as the Highveld priority area, as this is where most coal-fired power stations are located. The study area, location of sources population density used in the study are shown in Figure 3 below. The Sasol 1 and 2 stations and the Kelvin station (western border of the study area) were not included.

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 must be accounted for in the study (refer to Health impacts on page 25). 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 uses 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 Figures 4 and 5.

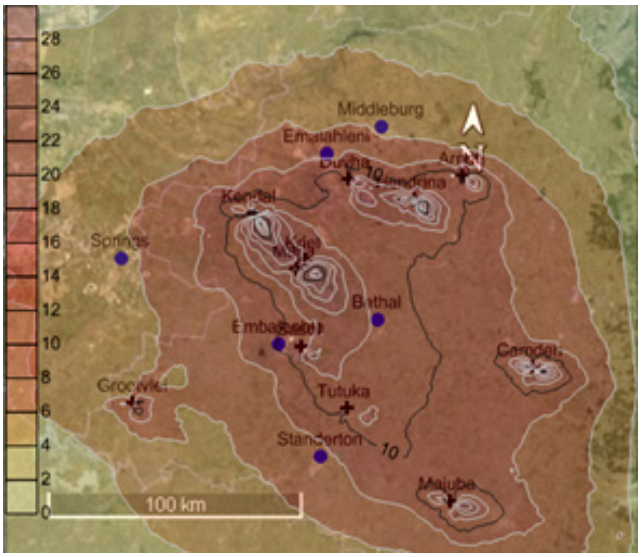

Figure 4: *Modelled annual average SO_2 concentration (µg/m³) over the study area for baseline conditions*

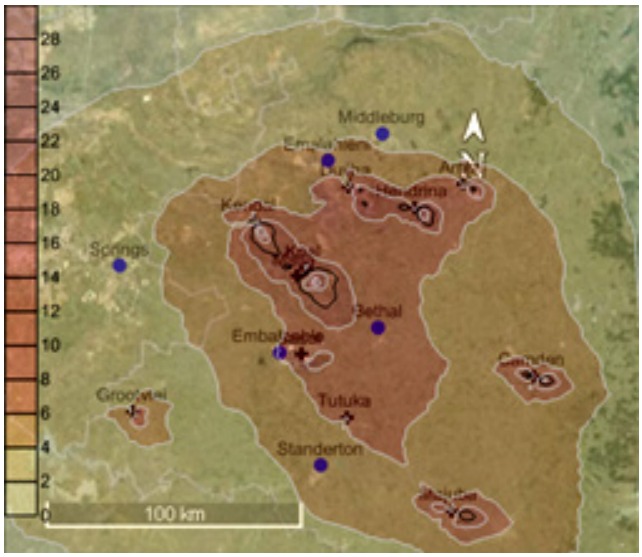

Figure 5: *Modelled annual average SO_2 concentration (µg/m³) over the study area for compliance with the new plant standards*

Discussion

In the evaluation of costs and benefits, 2020 was used as the base year for all calculations, based on the timeframe in which the requirements for existing plants to reach new plant standards comes into effect. The study further assumes that all retrofits could be completed by 2020, with water supply and lime supply infrastructure in place. This assumption ensures that the maximum benefit intended resulting from the regulations is accounted for. The study area was taken as the Highveld priority area, as this is where the majority of coal-fired power stations are located. Costs and benefits were escalated to adjust for inflation.

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.

Calpuff was selected for the dispersion modelling as the model includes chemical transformation and can handle calm conditions often experienced on the Highveld. The model is recommended for long-range transport (>50 km) and for multiple sources (DEA 2013b). 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.

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 methodology for the health impact assessment was as follows:

- The changes in ambient concentration of SO_2 and sulfates between the baseline (current operations) and compliance scenario (emissions at 500 mg/Nm³) were extracted from the dispersion modelling results for short-term (daily average) and long-term (annual average) impacts.
- 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).
- Population data was overlaid on the dispersion modelling results and health impacts were calculated using concentration response functions.
- 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 (SADHS 2003) and the Chronic Disease of Lifestyle in South Africa Study (Ehrlich & Jithoo, 2006). Baseline hospital admissions were calculated using data from the HASA Study (Da Costa, 2009)

The WHO-recommended methodology (Ostro, 2004) was used to calculate the mortality and morbidity impacts. Various studies have shown that the use of threshold values have a significant impact on the outcome (Rowe, Lang, Chesnut, Latimer, Rae, Bernow & White, 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.

Adjusting the United States Value of Statistical Life (VSL) (Viscusi & Aldy, 2002), a VSL of R53 million (2020) was calculated. The

income elasticity was taken as 1 to obtain a high estimate.

The data obtained from the Medscheme information (FRIDGE 2004) 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.

Results

The results of the analysis at a zero discount rate are shown in Table 5.

Table 5: *Estimates for costs and benefits without discounting*

Impact	Valuation mean (Rbil)	
Costs		**% of Total cost**
Direct Capital Cost	187	67
Water	29	10
Limestone cost	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 a nett negative NPV of R 166 billion, calculated using the high estimate for benefits and the central estimate for costs.

On the cost side, the most significant cost was the capital cost at 67% of the total. Cumulative operating costs, including water, lime and waste disposal costs, were calculated at 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 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, largely due to the high capital cost of FGD implementation (low and high estimates from –R17 billion to -R5 billion).

Sensitivity Analysis

Due to the significant impact, sensitivity analyses were conducted on the capital cost, mortality benefit and discount rate used. The high capital cost estimate was calculated using the data from Cleetus et al. (2012) and the low estimate using Orfanoudakis et al. (2005). The range of calculated direct capital cost was R306 billion to 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 concentration response functions used and the monetisation of the benefit (VSL estimates).

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 and VSL estimate used were shown to have a large impact on the results and sensitivity analysis is recommended to obtain a range of estimates, as shown in Figure 3.

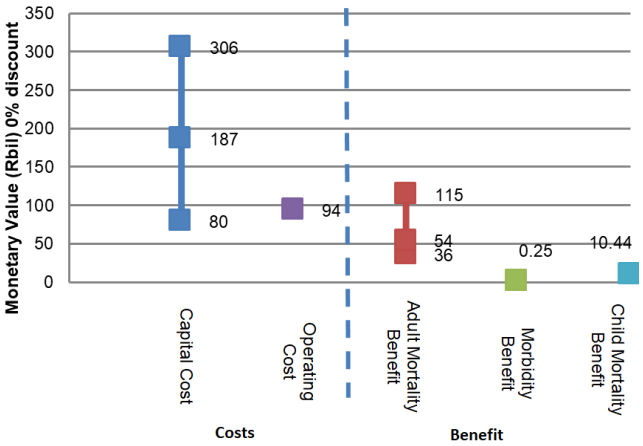

Figure 3: *Comparison of the magnitude of undiscounted costs and benefits*

The use of a discount rate in this case was found to have 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 (annually from 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.

Conclusions and Recommendations

The results of the study indicate that costs of the implementation of the category 1.1 new plant (2020) SO_2 standard exceed the likely quantifiable benefits due to the high capital and operating cost associated with the implementation of FGD. The analysis indicated that operating costs are a significant input and exclusion of these costs will undervalue the true implementation cost.

The study evaluated mortality benefits in terms of VSL, as it is a widely used methodology. The results indicate that the choice of VSL estimate has a significant impact on the mortality benefit. 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. A standardised South Africa-specific approach to VSL (or alternatively the Value of a Life Year) would ensure that air quality-related CBAs have comparable results. Due to the uncertainty regarding the applicability of health related data derived in other parts of the world, sensitivity analysis using various concentration response functions is required.

The ecological impact of acid deposition could potentially be significant on the Highveld. Further study to apply international experience with improved ecosystem functioning resulting from pollution reduction initiatives, is required.

The impact of the regulation on the broader economy 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. In this case, the basic financial CBA already indicates that it is unlikely that the benefits of regulation outweigh the costs.

The evaluation of costs and benefits are estimates based on the best available information and are most useful in informing decision making, prioritisation of initiatives and for illustrating the potential benefits of trade-offs. The results of the study indicate that, given the information currently available, it is unlikely that the benefit of reducing SO_2 emissions from existing facilities to the required standard outweighs the cost of implementation on the Mpumalanga Highveld.

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Research article

Fine PM emission factors from residential burning of solid fuels using traditional cast-iron coal stoves

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Abstract

Residential burning of solid fuels is a major source of fine particulate matter ($PM_{2.5}$), which degrades indoor and ambient air quality in low-income settlements. The adverse impact of fine particulate emissions on the environment and human health is well-documented in other countries such as China and India; however, there is need for local studies to report on emission factors from residential burning of solid fuels. An emission factor quantifies the total mass of a pollutant emitted per amount of fuel burned. Emission factor is an input parameter in air quality modelling to forecast a pollutant concentrations over time and when calculating total emissions from a specific source. Local emission factors are central to managing air quality for they give results that are representative of the source compared with international emission factors. Quantifying emissions, understanding household fuel use patterns and interaction with the stove (stove operation behaviour) during a burning event is fundamental when designing emission control strategies. The aim of the study is to quantify fine particulate matter emissions from residential coal burning using systematic field measurements. The objectives of the study are (i) to characterize stove operation behavior effect on the emissions and (ii) to quantify $PM_{2.5}$ emission factors using field measurements. Isokinetic (2015) and direct (2014) stack sampling tests were done to observe how PM emissions profiles change with stove operation behavior and to quantify $PM_{2.5}$ emitted per kilogram of fuel burned. Fine PM emission profiles change with stove operation behavior with an emission factor ranging 6.8 $g.kg^{-1}$ and 13.5 $g.kg^{-1}$. The study results implies that residential coal burning is a major source of fine particulate matter in the residential area. As demonstrated that stove operation behaviour affect stove to fuel combination emissions; it is therefore suggested that those factors leading to increase emissions should be kept minimum.

Keywords

Residential burning, emission factor, low-income settlements, particulate matter

Introduction

Solid fuels (coal, wood and dung) are the most common and frequently used energy sources in low–income settlements. This is due to their multi-purpose functionality, availability and affordability. Low-income households burn solid fuels in inefficient stoves resulting in increased emissions of incomplete combustion products (Worobiec et al., 2011). In kwaDela, a low income residential area in the Highveld, the annual solid fuel consumption in kwaDela is about 512 tons (Nkosi et al., 2017). Study conducted by Nkosi et al. (2017), revealed that solid fuel as the most common energy source with a total annual consumption of 512 tons.

Emissions from residential burning degrade ambient air quality and cause indoor air pollution leading to adverse impacts on human health (Smith et al., 2013). Fine particulate matter has been specifically identified as the most hazardous to human health due to its ability to infiltrate deeper parts of the respiratory system (Lee, 2010). There is a well- documented literature linking emissions of fine PM to cardiovascular and respiratory diseases (Seaton et al., 1995; Rehfuess et al., 2006; Albers et al., 2015). In 2007 emissions from residential combustion of solid fuels accounted for up to 75 % of the ambient concentration of fine particulate matter in Soweto, with a peak concentration of 110 $\mu g.m^{-3}$ (Nuwarinda et al., 2007). Similarly also 2007, residential burning accounts for up to 62.1 % of fine particulate matter ambient concentrations in Qalabotjha (Nuwarinda et al., 2007).

An emission factor (EF) is used to quantify the total amount of a pollutant emitted per amount of fuel burned (Amaral et al., 2016). Emission factors from residential burning are highly variable because they are governed by many factors can that be classified into (i) appliance characteristics (size of the stove, combustion air supply the design and size of the burning chamber), (ii) stove operation factors (lighting method used, and refueling intervals) (Schmidl et al., 2011).

Tissari et al., (2007) investigated the impact of stove operation on particulate and gaseous emissions using Mansory heaters and sauna stoves. Findings were that fine particulate emissions decreased with time after the ignition and there was a direct proportional relationship between fuel mass load and fine PM emissions. When doubling the fuel quantity the total amount of PM emitted was 1.9 times higher. Similar PM emission trends were observed by Johansson et al. (2004), large fuel batches had PM emissions that were 4 times higher when compared with small batches.

High ash content from previous burning events blocks air from entering the combustion chamber and also inhibits smokes from escaping, thus resulting in indoor air pollution (Garcia-Maraver, 2014). Ash also makes it difficult to light and for fuel to catch the fire thus results in combustion in low temperatures. Combustion conditions and fuel composition are primary determinants of the PM emission factors measured (Campbell, 1908).

Using emission factors that are not local and do not consider spatial difference in fuel type, composition, and stove operation behavior when quantifying the emissions may lead to an overall underestimation/ overestimation of PM emissions in developing countries (Sheng et al., 2013). For example, the amount of ash produced in South African coal is higher by a factor of 3 when compared with coal from the United States (Pretorius et al., 2015).

There is a need for local emission factors to reduce uncertainties related to data availability and EFs variation. One of the factors that causes variation in EF measured is the experimental method used (Shen et al., 2010). Laboratory derived PM EFs may be different from EFs derived in the field due to the difference in the burning conditions. According to Sheng et al., (2013) laboratory generated emission factors underestimate environmental conditions which are often uncontrolled, such includes wind speed, temperature and humidity. PM emissions from two laboratory experiments were 2 to 4 times lower than PM EFs measured from the field Roden et al (2006 and 2009). Important variables that have an effect on PM emissions such as the material and method used to start the fire, and uncontrolled refueling were excluded in a laboratory environment (Shen et al., 2013). Therefore, ignition method and fuel used when sampling flue gas in the field were the same with that of households used daily. Therefore, a better representation with actual ambient conditions compared with laboratory generated PM EFs can be drawn.

In contrast studies conducted in the laboratory (Tissari et al., 2009) and field environment (Tissari et al., 2007) reported that there is no significant difference on fine PM emissions measured from the field or laboratory environment. Fine PM from the field were less (2.7 $g.kg^{-1}$) when compared to laboratory measurements (5 $g.kg^{-1}$).

The aim of the study is to quantify fine particulate matter emission factors from residential coal burning using systematic field measurements. The results are essential in determining the amount of PM emitted and is essential for emission inventory and can also be used as an input parameter during modelling and source apportionment studies.

Material and Methods

The methodology used in the study is similar to Sheng et al., 2013. Figure 1 shows a gas and PM sampling train that was set up in two kwaDela randomly selected houses.

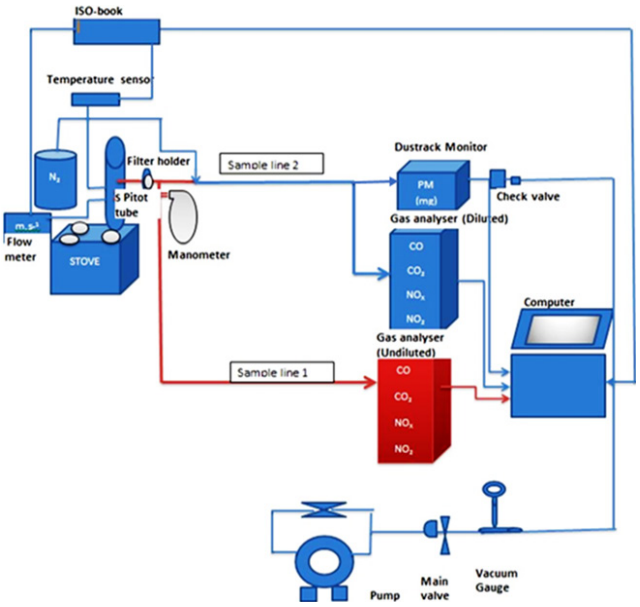

Figure 1: *Chimney sampling set up in kwaDela, showing experiment set*

The emission test were conducted using coal and wood purchased from local stores, exactly the ones where hops and coal sites where the residents usually buy. For ignition the residents used paper and wood, then after the fire is established, coal was added into the stoves. Traditional cast-iron coal stoves are the commonly used solid fuel burning devices in the residential area therefore were used for the field tests experiments. The coal stoves are primarily used for cooking, boiling water and space heating (Nkosi et al., 2017). The stove consist of a combustion chamber, cooking plates, a baking area and a chimney extending to the outside through the roof. Inside the chamber just below the fire bed are openings that allow ash from burned coal to be transferred to the ash-tray. The plates are also used as openings where the fuel is loaded.

The stoves are sold with a steel extension used to reshuffle the fire or remove ash that falls on the sides of the ashtray. Before each test experiment, the stoves and chimney were thoroughly cleaned to remove ash from previous burning events.

The amount of fuel used during the field experiments was measured with a calibrated digital Adam GFK floor scale. Stove operation (Stove cleaning, refueling times) and burning patterns (burning events start and end time) were recorded in a field note book. The notes were used to understand the DustTrak™

concentration profiles for each experiment. Table 1 shows the type and amount of solid fuels used for each experiment test.

Table 1: *Fuel mass used for each test (Test 1-5)*

Test	Coal mass (kg)	Wood and Paper (kg)	Total fuel
1	4.1	0.9	5.0
2	4.9	1.5	5.4
3	5.0	0.8	5.8
4	4.6	1.7	6.3
5	7	1.3	8.3

For 2014 field experiment, gravimetric PM concentrations were used instead of real time DustTrak™ measurement due to instrument malfunction during the sampling tests. 86R Teflon tubing (43 mm diameter, 1 µm pore size) and quartz (47 mm diameter, 2.2 µm pore size) membrane filters were used to collect PM emitted. Before use, borosilicate thimbles were washed with acid to decrease their trace metal content. Quartz thimbles were preheated at 900 °C for about 2 hr, also washed with acid to stabilise their weight and reduce their metal content. The filters used have about 99.9% efficiency with approximated 0, 2 heating loss. A Mettler Toledo balance was used to weigh the mass of the filters before and after sampling. A factory calibrated TSI DustTrak™ II Aerosol Monitor 8530 was set to zero then used to measure particulate concentration ($\mu g.m^{-3}$) at averaged over a minute.

An S type Pitot tube pressure meter with an accuracy of ± 5 %was used to measure the flow pressure. The velocity is calculated from the gas flow pressure (P_2 - P_1), difference between the static and direct pressure flow is recorded as the velocity of the flue gas. The velocity measured was used to determine the sampling tunnel gas flow $m^3.s^{-1}$, for every PM concentration measured (Equation 2).

A High-Volume Air Sampler monitor was adjusted at 5 minute interval to control flue gas flow during the sampling experiment. The flow rate ranged between 3 to 5 $L.min^{-1}$, it was slightly increased when the concentrations were low and decreased when they were high. A calibrated digital pressure Calc TM flow meter reader measured the flow rate averaged over a minute. Two gas analysers Horiba, (PG 350, PG 250) equipped with a Non-dispersive Infrared Absorption sensor measured diluted and undiluted gases at 10 seconds interval (CO, CO_2, O_2, SO_2 and NO_x). Prior each test the analysers were initialized (zero) and span calibrated. The dilution ratio was calculated and multiply the PM concentration measured as shown in equation 1.

$$\frac{CO\ u}{CO\ d}$$ ***Equation 1***

COu Refers to the undiluted carbon monoxide concentration and *COd* refers to the carbon monoxide diluted concentration.

To account for dilution, the calculated dilution ratio was applied to the DustTrak™ II PM concentrations before emission rates calculations. An ISO BOOK equipped with a flow meter and K–type thermocouple monitor was used to measure flue gas temperature inside the chimney. The diameter of the diluter used was 11, 5 cm therefore, a single sampling point was used to extract the flue gas.

The requirement for stack particulate sampling techniques is that the flue gas measured is representative of the gas stream inside the sampling tunnel. The velocity homogeneity, number of sampling points and isokinetic withdrawal determines if the sampled flue gas is a good representative (Hildermann et al., 1984). To reduce uncertainties associated with flue gas sampling, a dilution system that complies with EPA direct chimney sampling conditions and the code of Federal Regulations, isokinetic sampling method in accordance with US EPA and heating systems were used. The flue gas sampled was diluted with pure nitrogen gas to simulate ambient conditions, avoid condensation of the particles, gases and capture secondary particles. Nitrogen gas was chosen due to its ability to reduce about 60-80 % of artifacts that may occur under stove combustion conditions (Wien et al., 2001). Isokinetic system was implemented for all tests sampling to prevent particles deposition or deviation from the gas flow pattern sampling. This was achieved by monitoring the nozzle gas sampling rate and the velocity of the gas flowing inside the sampling duct. A filter heating system (80 °C) was used to remove any moisture that could cause particle size growth. Using a heating system also prevents particle loss along the inlet line, high temperatures along the heated line give the particles kinetic energy (Hildermann et al., 1984). Gaseous and particulate matter measurements began immediately after ignition and until the last second of the burning event duration. The sampling tests in 2014 lasted for 2 hours however 2015 tests lasted for 3 hours. The extension of the burning duration in 2015 was due to the slight increase in the PM concentration observed just before the dying stage of the fire in 2014. Therefore 2015 field test sampling duration was extended to 3 hours to capture the concentration increase during the dying stage of the fire. Test 1, represent a resident who would light the fire, leave it undisturbed until the fire burns out. Test 2 is a resident who lights the fire with 2 refueling events while test 3 pokes the fire after ignition. To account for DustTrak™ over-estimation, a correction factor calculated by comparing online measurements with exposed filters that were gravimetrically analysed was applied to the concentration before calculating the emission rates (Language et al., 2016). PM mass flow rate was determined from the equation 2.

$$Flowrate = \left(\frac{\pi}{4}\right) * (chimney\ diameter)^2 * velocity$$

Equation 2: *PM volumetric flow rate*

Where $(\pi/4)*(\text{chimney diameter})^2$ accounts for the chimney circumference (m^3). π is the constant value for pie equals to 3.14. Velocity (s^{-1}) refers to the flue gas speed inside the chimney

measured by the Pitot tube. The flow rate is product of both the chimney circumference and velocity. The DustTrak™ PM concentrations mg.m^{-3} were multiplied with the corresponding flow rate (m^3.s^{-1}) to determine the total PM emitted per second as shown in Equation 3.

$$PMEmission\ \left(mg.s^{-1}\right) = ([PMMeasured]) * (Volumetric\ flow)$$

Equation 3: *Emissions calculations*

Where *PMEmission* is the amount (mg) of particulate matter emitted for the period of the burning event and is the product of the (*[PMMeasured]*) (mg.m^{-3}) particulate matter concentration measured by the DustTrak™ II and the *Volumetric flow* (m^3.s^{-1}). Equation 4 was used to determine the emission factor.

$$PM\ Emission\ factor\ \left(\frac{mg}{kg}\right) = \frac{total\ PM\ emitted\ (mg)}{Amount\ of\ fuel\ burned\ (kg)}$$

Equation 4: *Emission factor for* $PM_{2.5}$

PM Emission factor is the total amount of PM emitted per kilogram of fuel burned. It is calculated by dividing the total emissions (*Total PM emitted*) from each test by the total amount of fuel burned (*Amount of fuel used*) during the event.

After sampling the filters were placed in a container and kept in a zip lock bag for less than 48 hours to avoid contamination and loss of particles. To remove any moisture accumulated during sampling, a laminar flow was used to dry the filters. Field blanks were used to account for the filters PM loss and gain during the field experiment tests. The total amount of PM collected from the tests and field blanks was gravimetrically analyzed with a high precision digital balance (Mettler Toledo). The filter mass and flow rate readings difference before and after sampling were used to determine total fine PM sampled in grams Equation 5. The obtained PM gain from field blanks was subtracted from the total PM mass derived from the field tests experiments.

$$PM\ total\ emission = Filter\ mass\ (mg)\ \ (before\ sample - after\ sample)\ (1000)/volume\ of\ dry\ gas$$

Equation 5: *PM volumetric flow rate*

Where Filter mass (mg) (before sample- after sample) is the filter mass difference calculated from mass recorded before and after the test experiment. (1000) converts the mass from mg to g. volume of dry gas is the flow meter reading difference of the respective experiment test.

Quality control

A total of six field tests were done but due to instrument malfunction, only five tests were used to estimate fine PM emission factor. The number of measurements is constrained by the fact that the sampling of field measurements have a huge impact on the household's themselves. When considering the ethical implication of the experimental results. It is better to limit the number of observations since this is the first time PM emissions were sampled from the households. Daily household energy use patterns were used in order to carry-out experiment under real use scenarios. Before each test the instruments (sampling lines) were cleaned with compressed air and checked for leaks.

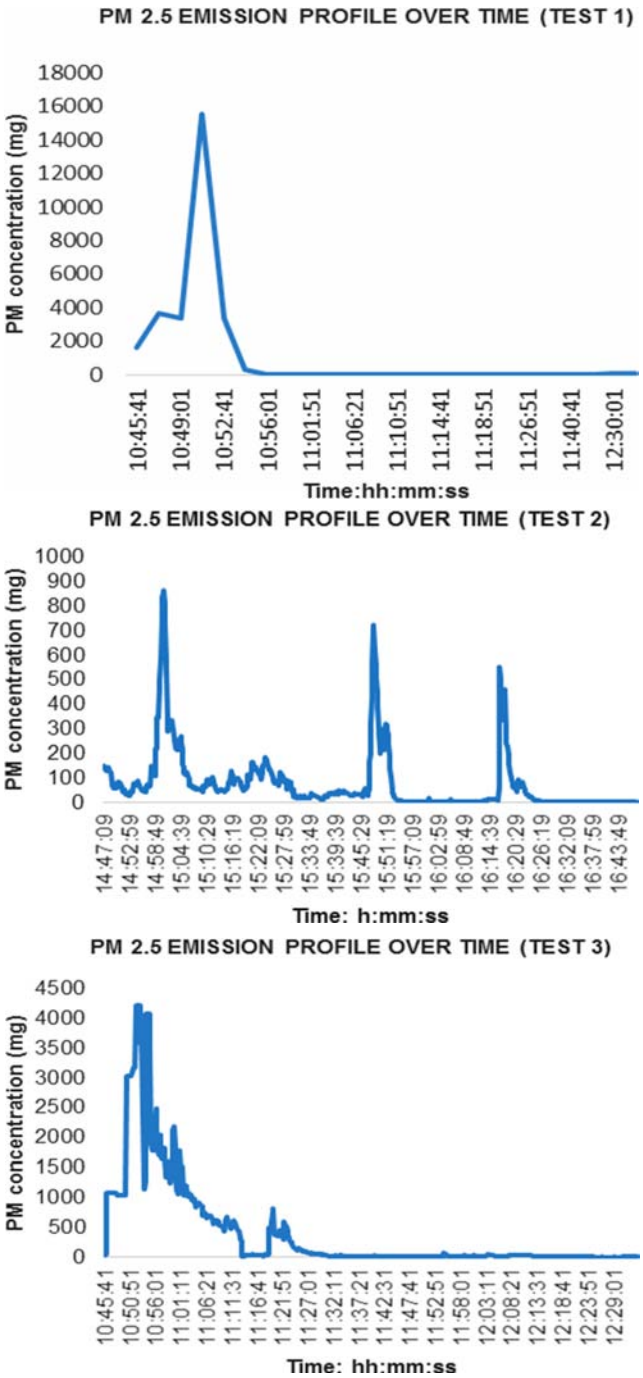

Figure 2: *Fine particulate emissions tests results in KwaDela, Test 1–3.*

Results and Discussion

Particulate matter emissions peaks were associated with the

Table 2: *Fine particulate matter ($PM_{2.5}$) filters Isokinetic direct chimney sampling results showing emissions of $PM_{2.5}$ test (4–5) PM=Particulate Matter*

Field test experiment number	Date and duration of burning event	Fuel used (kg)	Mass of filter before sampling (g)	Mass of Filter after sampling (g)	Flow rate $m^3.s^{-1}$	Total mass of PM in mg	$PM_{2.5}$ emission factor ($g.kg^{-1}$)
4	01/04/2015 3 hours	6.32	3.4	3.5	0.48	189,432	30
5	31/03/2015 3 hours	8.26	2.7	2.8	0.38	107,208	13

following fire stages, (i) the beginning of the burning event (ignition), (ii) when the fire was being reshuffled (iii) when fuel is reloaded, Figure 2. Test 1 has a single spike, test 2 has three and test 3 has only two. Above 80 % of the total emissions from test 1 and 3 were emitted during the 30 minutes of the burning event. The DustTrak™ real time PM concentrations showed that emissions on test 2 were distributed throughout the burning event with the ignition phase having the highest emission. Spikes on test 4 and 5 could not be identified because emissions were not measured with a DustTrak™ but collected with filters. High emissions of fine particulate matter were identified at the beginning of the burning event and during a refueling event. Test 1 and 3 had high emissions when compared to test 2.

Emissions from ignition stage result from combustion of wood, paper, and emissions from refueling result from coal combustion.

Findings of our study are similar to those of Tisarri et al. (2009) where during refueling, fuel combustion occurs at high temperatures producing large amount of ash which is a source of fine particulate matter. High emissions of fine particulate matter at high combustion rate are caused by oxygen deficiency in the combustion chamber and inadequate mixing of fuel and air (Tisarri et al., 2007). Results from a separate study on emissions of PM from traditional stoves indicated that about 98 % of total PM is emitted during ignition and refueling (Maddalena et al., 2014).

The amount of fuel used for test 1 and 2 is almost the same with a difference of 0.4 kg however, test 1 PM emissions are almost two times higher than test 2. The difference in the amount of fuel used in test 1 and 3 is 0.8 kg but the amount of PM emitted from both tests do not vary significantly. For test 2 the emissions are distributed throughout the burning event with high emissions occurring on the first 30 minutes (Ignition phase) and least emission during the last hour (refueling) of the event. Table 2 presents filter based emission factors, amount of fuel used, and filter mass before and after test 4 and 5 field experiment. Less fuel (6.32 kg) was used for test 4 compared with test 5 (8.26 kg) however, test 4 has almost 2 times more emissions compared with test 5. Studies done by Shen et al. (2013) on the effect of fuel mass load on PM emission factors reported that fuel mass load does not have a significant impact of amount of PM emitted and PM size distribution. Lamberg et al. (2011) conducted a study in a laboratory environment, his findings were that there is a negative correlation between fuel load and PM emissions. Decreasing fuel quantity decreases the flue gas residence time in the combustion chamber due to high air flow, therefore, results in insufficient mixing of flue gas and combustion air. PM mass concentration emitted increases with increase in excess air inside the combustion chamber (Johansson et al., 2003). This may explain why emissions for test 4 where higher than test 5 even when more fuel was used for test 4 compared with test 5.

In agreement with the current study a field study done in Malawi quantifying fine particulate emissions from burning of solid fuels using traditional combustion methods reported an emission factor of 7.1 $g.kg^{-1}$ (Wathore et al., 2017). The reported emission factors were highly variable with a standard deviation of ±1.3 $g.kg^{-1}$. Jordan and seen (2005), Schmidl et al., (2011) confirmed that the difference in supply and regulation of combustion air results in variation of the emission factors.

The large variation in the current study emission factors may have also resulted from simultaneously using PM concentration measured with DustTrak™ and filters, also using a small number of field tests experiments. Moreover not considering other factors known to have influence on the emission such as particle size and moisture may introduce uncertainties in the emissions factors.

One disadvantage of using a dilution system is that it causes the condensed material to go unmeasured. This explains why in the current study the emissions measured with the filter are more than DustTrak™ emissions. Similar observations were made by Tissari et al. (2008a) when he measured fine PM emissions from sauna stoves using filters and an electronic low-pressure impactor. The emission measurements from the impactor were 8 % lower than filter-based measurements.

A two-tailed chi-square test (at the 95% confidence level) was used to determine the Pearson Chi square and likelihood ratio for gravimetric and real-time DustTrak based calculated emission factors. The P- value for gravimetric emission factors was 0.157 with a likelihood ratio of 0.096. For the real time DustTrak calculated emission factors the P value was 0.199 with a likelihood ratio of 0.159.

The emission factors calculated from DustTrak measurements were not significantly variable with a standard deviation of ±.2.1 $g.kg^{-1}$. Whereas the filter derived emission factors had a standard deviation of ±8.5 $g.kg^{-1}$. The filter based emissions were up to 7 times higher when contrasted with DustTrak™ II emissions.
The error from using five field tests was minimised by using a "bootstrapping" simulation method to extrapolate the calculated emission factor to 500 samples. Bootstrap simulation was used quantify the uncertainty in the mean emission factor of all test results at 95% confidence interval.

Table 3: *$PM_{2.5}$ Emission factors from all tests in $g.kg^{-1}$*

Field test experiment	Test 1	Test 2	Test 3	Test 4	Test 5
$PM_{2.5}$ ($g.kg^{-1}$)	8.1	4.1	7	30	13

Conclusion

Fine PM emission profiles change with stove operation behavior with an emission factor of 6.8 $g.kg^{-1}$ to13.5 $g.kg^{-1}$. The study results suggest that stove operation behaviour such as fire poking and refuelling lead to high emissions during a burning event. Even though emissions were not calculated based on the different fire stages in the current study, all the tests graphs show high emissions to be occurring from the ignition phase of the fire followed by refuelling. Having a small sample and significant variation within the sample introduces uncertainties to the final emissions value. Therefore there is a need to conduct field studies using large sample size to account for the emissions variability. The study suggest that real time stove mass and combustion chamber temperature readings during the burning event should be reported with the emissions to better understand the emissions profiles.

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Research article

Household air pollution exposure and respiratory health outcomes: a narrative review update of the South African epidemiological evidence

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Abstract

One of the greatest threats to public health is personal exposure to air pollution from indoor sources. The impact of air pollution on mortality and morbidity globally and in South Africa is large and places a burden on healthcare systems for treatment and care of air pollution-related diseases. Household air pollution (HAP) exposure attributed to the burning of solid fuels for cooking and heating is associated with several adverse health impacts including impacts on the respiratory system. The researchers sought to update the South African evidence on HAP exposure and respiratory health outcomes from 2005. Our quasi-systematic review produced 27 eligible studies, however, only four of these studies considered measures of both HAP exposure and respiratory health outcomes. While all of the studies that were reviewed show evidence of the serious problem of HAP and possible association with negative health outcomes in South Africa, no studies provided critically important information for South Africa, namely, local estimates of relative risks that may be applied in burden of disease studies and concentration response functions for criteria pollutants. Almost all of the studies that were reviewed were cross-sectional, observational studies. To strengthen the evidence of HAP exposure-health outcome impacts on respiratory health, researchers need to pursue studies such as cohort, time-series and randomised intervention trials, among other study designs. South African and other researchers working in this field need to work together and take a leap towards a new era of epidemiological research that uses more sophisticated methods and analyses to provide the best possible evidence. This evidence may then be used with greater confidence to motivate for policy-making, contribute to international processes such as for guideline development, and ultimately strengthen the evidence for design of interventions that will reduce HAP and the burden of disease associated with exposure to HAP in South Africa.

Keywords

environmental health, air quality, household emissions

Introduction

While episodes of unusually high air pollution attract attention and public health concern, the greatest damage to public health is associated with long-term exposure to air pollution (HEI 2017). Outdoor and indoor personal exposure to air pollution, combined, comprise the largest environmental risk factor for mortality, responsible for 6.4 million deaths in 2015 (11% of global deaths) (Cohen et al. 2017).

The costs of air pollution in Africa are high - estimated at around USD450 billion in 2013 (Roy 2016). The economic impacts include life years lost, increased healthcare (and subsequent demand on government) and lost worker productivity due to air pollution impacts on health.

Globally, epidemiological studies and systemic reviews have shown associations between exposure to household air pollution (HAP) and a variety of diseases and symptoms (Jedrychowski et al. 2017; Tanaka et al. 2012; Koo et al. 2011; Pope et al. 2010). The indoor environment represents an important microenvironment in which people spend approximately 90% of their time each day (WHO 2014a).

According to the World Health Organization (WHO), HAP is responsible for more than 1.6 million annual deaths globally, and 2.7% of the global burden of disease (WHO 2014b). HAP is reported to increase irritation of the airways, coughing, irregular heartbeat, difficulty breathing and premature death in people with heart and lung disease (Gurley et al. 2013; Laumbach and

Kipen, 2012; Ritz and Wilhelm, 2008). Exposure further worsens existing respiratory diseases such as bronchitis, cardiovascular disease and emphysema (Laumbach and Kipen, 2012; Fisk et al. 2010; Rinne et al. 2006). Exposure to solid fuel burning indoors has also been associated with tuberculosis (TB) (Jafta et al. 2015; Lin et al. 2014), cataract (Ravilla et al. 2016) and adverse birth outcomes (Wylie et al. 2017; Pope et al. 2010).

HAP is derived from multiple indoor sources (Colbeck and Nasir 2010) varying from one building to another and depending on fuels used for heating and cooking, smoking habits and use of a wide variety of consumer products and building materials (Shezi et al. 2017; Jafta et al. 2017; Tanaka et al. 2012; Verma et al. 2010). This is also influenced by time activity patterns i.e. cooking period, number of meals cooked, single or multiple fuel use and number of cigarettes smoked and further compounded by confounding variables such as outdoor air pollution sources near the homes (Shezi et al. 2017; Colbeck and Nazir 2010). In developing countries, the most significant indoor air quality issue is exposure to pollutants released during combustion of solid fuels used for cooking and heating in the home (Wylie et al. 2017; Pope et al. 2010).

The measured mean concentrations of the HAP vary depending on the sources, for example, average particulate matter with an aerodynamic diameter of 2.5 ($PM_{2.5}$) concentration in households using solid fuels has been reported to range from 133.5 µg/m^3 to 670 µg/m^3 (Balakrishnan et al., 2015; Balakrishnan et al., 2013; Clark et al., 2010), while the carbon monoxide (CO) average concentrations have been reported to range from 2.7 ppm to 14.3 ppm. The average $PM_{2.5}$ concentrations in households using cleaner fuels have been reported to range from 10 µg/m^3 to 38 µg/m^3 (Shezi et al. 2017; Tunno et al., 2015; Evans et al. 2000). Studies conducted in South Africa have also reported high mean concentrations in households using both clean and dirty fuels (Jafta et al. 2017; Shezi et al. 2017; Wernecke et al 2015; Rollin et al. 2004) which exceed the guidelines set out by the WHO for indoor air quality in households; South African indoor air quality guidelines pertaining to household air pollution are under review and have not yet been promulgated.

South Africa is a middle-income country burdened by poverty and inequality where many South Africans are exposed to biomass fuels used for cooking and heating indoors. Many families live in close proximity to industrial areas and major roads (Albers et al. 2015; StatsSA 2012; Norman 2007) where outdoor pollutants may also be transported indoors (Colbeck and Nasir 2010). Vulnerable groups such as people who are elderly, women, young children, people with pre-existing diseases and people living in poverty are more susceptible to air pollution health impacts (Barnes 2014).

The WHO comparative risk assessment determined the mortality burden and Disability Adjusted Life Year (DALYs) due to HAP. HAP associated with household solid fuel use for cooking and heating caused 0.5% of all deaths in South Africa in 2000 (uncertainty 0.3% -0.6%) (Norman et al., 2010). More than 10 years ago, Wichmann and Voyi (2005) published a review of the air pollution and health epidemiological studies in South Africa, focussing on methodological issues and the need for quantitative intervention studies. In 2014, Barnes (2014) described behavioural change studies for reduction of HAP in developing countries, calling for more rigorous study design and interventions grounded in behavioural change theory. In recent years, anecdotal evidence suggests that studies on the impact of HAP and associated respiratory health outcomes has been growing in South Africa (Barnes et al., 2009). In this narrative review, evidence from the recently published South African studies that considered HAP exposure and associated respiratory health outcomes to augment our current knowledge is collated. An attempt to identify research gaps and suggest directions for further studies is made.

Review methods

A quasi-systematic (i.e. following the guidelines for systematic review but with slight differences in methods to accommodate all available evidence) review of the South African evidence on HAP (term group 1) and associated respiratory health outcomes (term group 2) was conducted using the PRISMA guidelines (Moher et al. 2009). PubMed, Web of Science, Science Direct and Google Scholar were searched for studies with full text in English, published between 2005 and 2017. The term groups listed in Table 1 were used for the separate searches and in various combinations. The reference lists of included papers were searched to ensure that no studies were omitted.

Table 1: *Search strategy applied to retrieve published articles reporting on household air pollution and respiratory health outcomes associated with household air pollution in South Africa.*

Term group 1	Term group 2	Combinations
Household air pollution OR indoor air pollution OR air pollutants OR biomass OR solid fuel OR cooking fuel OR heating fuel OR wood smoke OR wood burning OR paraffin OR dung OR coal burning OR indoor air OR particulate matter OR carbon monoxide OR nitrogen dioxide OR sulphur dioxide	Respiratory health outcomes OR asthma OR bronchitis OR wheeze OR pneumonia OR runny nose OR cough OR lower respiratory tract infection OR upper respiratory tract infection	Term from Group 1 plus Term from Group 2 until all combinations exhausted

To be eligible, studies had to have been carried out in South Africa. All epidemiological study designs including cross-sectional studies, cohort studies, longitudinal studies, case-cross-over studies, intervention studies etc. were eligible. All studies that provided estimates of HAP exposure concentrations (by indicator, proxy or actual measurements) as well as respiratory health outcomes found to be associated with HAP exposure were included. We noted whether the correlate and health outcome was measured at the level of the individual, household or community. The review was not restricted by defining a minimal study sample size and studies of all sample sizes were included.

Results

After removing all ineligible articles, mostly since they were studies unrelated to South Africa, our searches using both formal methods (described above) and informal means (such as personal communication with researchers) produced a total of 27 articles. Of this total, only 4 studies measured HAP exposure (mainly by proxy/indicators of air pollution exposure) and respiratory health outcomes. Ten studies assessed HAP but no respiratory health outcome(s) and the remaining 13 studies did not measure HAP, instead used ambient AQ monitoring station data (or other) for exposure assessment.

Table 2 describes in brief the four studies that included HAP exposure monitoring and associated respiratory health outcomes by type of study, setting, sample size, methods, exposure information and outcome results. None of the studies measured HAP, instead, they used indicators or proxies for exposure, such as presence of environmental tobacco smoke (ETS) and fuel used for cooking and / heating in the home. Elf et al. (2017) used passive samplers to measure ETS. All of the studies were cross-sectional epidemiological study designs and the largest sample size was ~ 3 000 schoolchildren (Shirinde et al., 2014) with the other three studies having sample sizes of less than 1 000 individuals (no sample size calculations were provided). One study did not set out to measure HAP but rather focussed on ETS exposure.

Inter-comparison of the findings of the studies is difficult due to the heterogeneity in the study design, target population and health outcomes of interest. Both Albers et al. (2015) and Shirinde et al. (2014) found that among schoolchildren (albeit of different ages) there was an association between respiratory outcomes (most notably wheeze) and use of non-electrical heating sources. Elf et al. (2017) and du Preez et al. (2011) consider TB as a health endpoint and ETS and solid fuel use, and ETS, respectively, in relation to TB.

Tables 3 and 4 summarise the remaining studies that the researchers reviewed as part of this review exercise. These studies did not meet the study inclusion criteria; however, they do provide useful information on the levels of various air pollutants in the indoor and outdoor environments of different parts of South Africa. Since they did not mention respiratory health outcomes, the researchers do not discuss them in detail here, but included them as a reference for future research.

Of the twenty-three studies listed in Table 3 and 4, two studies were review articles, six studies used questionnaires and interviews which are methods of assessment that are based on self-report and recall. Either indoor and / or outdoor pollutants were measured in 15 studies and some of the studies measured multiple pollutants. Particulate matter with an aerodynamic diameter of 10 (PM_{10}) was measured in twelve studies while $PM_{2.5}$ and particulate matter with an aerodynamic diameter of 4 (PM_4) were measured in four and two studies, respectively. Two studies measured respiratory particulate matter (RPM). NO_2 was measured in five studies; NO_x in one study and volatile organic compounds were also measured in one study. Three studies measured CO and seven studies measured sulphur dioxide (SO_2). Ozone (O_3) was measured in one study and total reduced sulphur was also measured in one study. Indoor undisturbed dust samples were undertaken in one study to analyse for dichlorodiphenyltrichloroethane (DDT), dichlorodiphenyl-dichloroethylene (DDE) and dichlorodiphenyl-dichloroethane (DDD), while settled dust and airborne fungal sampling was also conducted in one study.

Discussion

Few published studies conducted in South Africa in recent years have sought to show associations between HAP and respiratory health outcomes. All of the studies that did so used cross-sectional study design. This is unfortunate since most international systematic reviews for HAP exposure and health outcome association only use data from cohort, time series, long-term panel (longitudinal) and case-crossover studies. Our studies are therefore not contributing to international evidence on this important topic, nor are they providing reliable evidence that is generalizable to others parts of South Africa. While cross-sectional studies are typically less expensive and easier to implement compared to other epidemiological study designs, the data produced from cross-sectional studies is not as useful and the lack of randomisation, among other shortfalls, prohibits generalisation. Researchers need to work towards larger, more complex epidemiological studies and also use of existing, high quality (where possible) data in South Africa that will help to address the problem of HAP and respiratory health. While it will likely cost more, other study designs beside cross-sectional studies, will provide opportunities for HAP monitoring over substantial periods of time to provide exposure information that can then be associated with health outcomes, preferably diagnosed by a health professional (and not self-reported or parent-reported) for more precise results, in a more meaningful way.

It is also important to account for potential biases and critical confounders, including age, sex, and individual socio-economic status, among others, when planning a HAP and respiratory health study, and also when reporting study results as failure to control for confounding variables can lead to erroneous

associations between HAP and respiratory outcomes.

While the reviewed studies predominantly report on combustion generated variables (type of fuel used for cooking or heating and active or passive smoking) with PM, CO and NO_2, being products of incomplete combustions, other sources of indoor air pollutants not necessarily emitted by incomplete combustion such as building materials, ventilation characteristics and cleaning agents may not be ignored.

While, exposure data misclassification and validity of exposure are often overlooked or underestimated and not critically discussed (Wichmann and Voyi, 2005), the lack of direct exposure measurement such as the use of home monitors and personal monitors present results that are in some part questionable.

None of the studies reviewed provided concentration response functions for the criteria air pollutants in South Africa (DEA, 2009). This point was made by Wichmann and Voyi (2005) and it still holds true in 2017. This is still a major gap in our knowledge in both South Africa and on the continent. South African researcher continue to use the international literature and WHO evidence, without making contribution to this important body of knowledge.

Another shortfall is that only one study reviewed in this exercise was an intervention study (under real life conditions). The WHO calls for the support of research that is driven by interventions and searching actively for solutions, in particular for urban settings, in relation to air pollution and health (WHO 2014). In South Africa, research partnerships and consortia may assist to create large research teams to lead intervention studies to address HAP and adverse health impacts in areas of greatest concern.

Our study had some limitations. There were very few studies to critically review hence the researchers opted to describe them descriptively instead. Wichmann and Voyi (2005) provided a critical synthesis of the evidence in this field up to 2005. The authors did not find a substantial number of studies to add to that body of literature beyond 2005. The authors set out to find studies that had measured HAP and simultaneously measured respiratory health outcomes. The authors may have not identified all studies relevant to the review inclusion criteria, although they tried to avoid this by searching the literature regularly and speaking with South African researchers in the air pollution and health fields. The authors did not consider mortality as an end-point. Wichmann and Voyi (2006) found that exposure to cooking and heating smoke from polluting fuels was significantly associated with 1–59-month mortality, after controlling for mother's age at birth, water source, asset index and household crowdedness (RR=1.95; 95% CI=1.04, 3.68).

Conclusions

The South African studies on HAP and respiratory health outcomes do provide some evidence of the serious impacts of HAP and especially the use of solid fuels in the home on respiratory health in the country, but the studies are few and limited. South African and other researchers working in this field need to work together and take a leap towards a new era of epidemiological research that uses sophisticated methods and analyses to provide the best possible evidence. This evidence may then be used with greater confidence to, for example, motivate for policy-making, contribute to international systematic reviews for guideline development and other purposes, and ultimately strengthen interventions that will reduce HAP and the burden of disease associated with exposure to HAP in South Africa.

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Table 2: *Findings from studies on household air pollution exposure (most often using a proxy) and associated respiratory health outcomes in South African settings.*

Author(s) (year)	Study design	Population/Setting	Sample size	Exposure	Reported associations between HAP and respiratory outcomes
Elf et al (2017)	Convenience sample of homes and household members participating in an ongoing active case-finding study	All adults (≥ 18 years of age) and children between 7-17 years of age living in the same household as the index tuberculosis (TB) case, including the index cases themselves in Matlosana district townships surrounding Klerksdorp were included in this air pollution study.	124	The household survey assessed the primary and secondary fuel use, duration of use for both heating and cooking in the households. Individual questionnaire collected information on tobacco use and environmental tobacco smoke (ETS) exposure. Passive and nicotine monitors were placed in the common living space of each home for a period of 14 days.	A high prevalence of air pollution from second hand smoke, solid fuels, and kerosene among individuals in homes with a case of prevalent active tuberculosis disease was observed. Adults in 40% of homes reported a daily smoker in the home, and 70% of homes had detectable air nicotine. In homes with a history of previous TB (prior to but not including the index case) as compared to those without previous tuberculosis, both second hand smoke (83% vs. 65%, respectively) and solid / kerosene fuel use for more than 1 h/day (27% vs. 21%, respectively) were more prevalent.
Albers et al (2015)	Cross-sectional study	Children 9-11 years old in grades 4 and 5 in six randomly selected primary schools in eMalahleni and Middelburg, Mpumalanga.	627	Type of energy sources and associated respiratory outcomes were collected using a structured questionnaire completed by parents/guardians of the children.	Past 6 months doctor-diagnosed bronchitis 66/422 (15.6 %); 'Ever' during childhood to date doctor-diagnosed asthma 39/551 (7.1 %); self-reported chest wheeze 59/519 (11.4 %); chest cough 45/445 (10.1 %); phlegm on the chest 123/481 (25.6 %); any respiratory health condition 214/627 (34.1 %). Overall prevalence of respiratory ill-heath was 34.1%. The prevalence of respiratory ill-health conditions was significantly elevated among children from households using non-electrical fuels v. electricity for cooking (43.9% v. 31.6%; adjusted p-value 0.005). The same was noted among those using non-electrical fuels for heating (37.8% v. 29.0%).
Shirinde et al (2014)	Cross-sectional study	Children (12 - 14 years) from 16 randomly selected schools in Thembisa and Kempton Park in the Ekurhuleni Metropolitan Municipality.	3 424	Information on active and passive smoking at home and at school, type of energy sources for cooking and heating at home, transportation mode to school and frequency of trucks passing near residences was collected using a questionnaire.	Exposure to ETS at school was associated with wheeze ever (OR 1.22 95% CI: 1.03-1.45) and current wheeze (OR 1.33 95% CI: 1.08-1.64). In households that reported frequent use of gas for residential heating, the likelihood of wheeze ever increased by 47% (OR 1.47 95% CI: 1.15-1.88). An increased likelihood of wheeze ever (OR 1.32 95% CI: 1.01-1.73), current wheeze (1.61 95% CI: 1.15-2.24) and current severe wheeze (OR 2.22 95% CI 1.28-3.77) was associated with trucks passing near homes for almost the whole day during weekdays.
du Preez et al (2011)	Cross-sectional study	Children based in Cape Town, aged 3-15 years who had contact with an adult starting clinic-based anti-TB treatment in the preceding 3 months were recruited.	196	Questionnaire was used to collect information on ETS exposure.	From the 65.3% children that reported ETS exposure, 49.5% were *M. tuberculosis* infected of these 63.3% were exposed to ≥2 household smokers.

Table 3: *Findings from studies that assessed indoor pollution exposure in South African settings but did not quantify respiratory health outcomes and / or investigate the association between household air pollution exposure and respiratory health outcomes.*

Author(s) (year)	Study design	Population/Setting	Sample size	Exposure	Reported associations between HAP and respiratory outcomes
Jafta et al. (2017)	Cross-sectional study	Children (<15 years of age) who participated as cases or controls in the TB study within eThekwini Municipality.	246 house-holds	Homes were assessed using a walkthrough checklist and indoor monitoring was conducted in 105 homes over a period of 24 hours for PM with an aerodynamic diameter of 10 (active) and 82 homes were monitored for SO_2 and NO_2 (passive) for a period of 2-3 weeks.	Mean indoor concentrations of PM_{10}, NO_2 and SO_2 were 64 µg/m³ (range 6.6–241.0); 19 µg/m³ (range 4.5–55.0) and 0.6 µg/m³ (range 0.005–3.4), respectively.
Shezi et al. (2017)	Cross-sectional study	Households of pregnant females from the north and the south of Durban participating in the mother and child in the environment (the MACE cohort study).	300 house-holds	Walk-through indoor assessment and post-activity questionnaire were used to collect information on the household building materials, occupant activities and outdoor sources such as industries and major roads in the vicinity of the homes. Indoor $PM_{2.5}$ levels were measured in 300 homes for a period of 24 hours.	The $PM_{2.5}$ levels ranged from 1.4 to 162.0 µg/m³. The mean (SD) of these levels was 38.3 (31.1) µg/m³, and the median was 28.0 µg/m³.
Nkosi et al. (2017)	Cross-sectional study	Children, including 10 asthmatics 13-14 years of age, a subset of the 2012 International Study of Asthma and Allergies in Children (ISAAC) from 10 schools in Gauteng and the North-West Province were included. Five schools were within 1- 2 km from a mine dump in Gauteng or the North-West Province and 5 were 5 km or further from a mine dump in these provinces.	100 children	Personal air sampling was performed in the breathing zone of 10 asthmatics learners randomly selected from each school, outdoor SO_2 and PM_{10} were measured for 8-h at each school.	Indoor respiratory dust in the classroom differed significantly between exposed (0. 17mg/m³) and non-exposed (0. 01 mg/m³) among children with asthma. Outdoor SO_2 levels were 0.002 ppb for exposed children and 0.01 ppb for unexposed children ($p<0.001$). Outdoor PM_{10} was 16.42 mg/m^{-3} and 11.47 $mg.m^{-3}$ for exposed and unexposed, respectively.
Language et al. (2016)	Panel study	KwaDela, Mpumalanga, South Africa	20 house-holds	Indoor (PM_4) and ambient ($PM_{2.5}$, PM_{10}) air pollution monitoring in 20 households over two years: two summers and two winters (10-12 weeks each); 207 household questionnaires were administered to determine household energy use and perceived quality of life.	The majority (97.10%) of households had access to electricity, however, there was still a high rate of solid fuel use – coal (75.36%) and wood (63.28%). 40.57% of households used a combination of these fuels. Ambient PM_{10} concentrations were 102.1 ± 76.96 and 99.29 ± 61.39 (µg/m³), respectively, and summer concentrations were 50.43 ± 29.59 and 66.03 ± 25.86 (µg/m³). $PM_{2.5}$ concentrations showed less seasonality compared to PM_{10}. Finer aerosols were lower than the coarse fraction by a factor of 4 to 5 (winters) and 2 to 3 (summers).

Table 3 continued ...

Author(s) (year)	Study design	Population/Setting	Sample size	Exposure	Reported associations between HAP and respiratory outcomes
Wernecke at al. (2015)	Case study	KwaDela, Mpumalanga, South Africa.	1 household	Indoor (PM_4) and ambient ($PM_{2.5}$, PM_{10}) air pollution monitoring in 20 households in winter 2013 and 2014 and summer 2014 and 2015; data presented for winter 2014.	Mean outdoor $PM_{2.5}$ and PM_{10}, indoor PM_4 and personal PM_4 concentrations were 27±18 and 48±122 and 17±23 and 16±7 μg/m^3, respectively.
Gaspar et al. (2015)	Cohort study	Households of women from the VHEMBE study on indoor spraying for malaria and health effects.	50 households	Indoor undisturbed dust samples analysed for dichlorodiphenyltrichloroethane (DDT), and its degrading products, dichlorodiphenyl-dichloroethylene (DDE) and dichlorodiphenyl-dichloroethane (DDD) to determine dust loading levels and compared these levels to paired serum concentrations of p,p′-DDT-and p,p' DDE in women residents.	p,p′-DDT and p,p′-DDE had the highest detection frequencies in both dust (58% and 34% detection, respectively) and serum samples (98% and 100% detection, respectively). Significantly higher detection frequencies for o,p′-DDT, p,p′-DDE, and p,p′-DDD were observed in dust samples collected in buildings that had been previously sprayed for malaria control. Significant, positive association between dust loading and serum concentrations of p,p′-DDT and p,p′-DDE (Spearman's rho=0.68 and 0.54, respectively) was observed.
Vanker et al. (2015)	Prospective cohort study	Pregnant women (20–28 weeks' gestation) from the Drakensberg Child Health study (DCHS) an African birth cohort.	633 women	Indoor air monitoring of PM_{10}, CO was conducted for a period of 24 hours, while indoor NO_2 and volatile organic compounds (VOC) were conducted for a period of 2 weeks.	31% were active and 44% passive smokers. Of HAP measured, benzene (VOC) was significantly above ambient standards with median 5.6 μg/m^3 (IQR 2.6–17.1). There were significant associations between the use of fossil fuels for cooking and increased benzene [OR 3.4 (95% CI 2.1–5.4)], CO [OR 2.9 (95% CI 1.7–5.0)] and NO_2 [OR 18.6 (95% CI 3.9–88.9)] levels. A significant seasonal association was found with higher benzene and CO levels in winter.
Jafta et al. (2012)	Cross-sectional study	Households of children (20 children from each of 7 schools, Grades 3-6) (mainly asthmatics) who took part in the South Durban Health Study.	135 households	Observation (walk through survey), sampling and analyses of settled dust and airborne fungal sampling.	Asp f1 allergen was detected in all homes, and Bla g1 allergen was detected in half of the homes. Detection frequencies varied from 51% for Bla g1 to 100% for Asp f1. House dust allergens, Der f1 and Der p1 exceeded concentrations associated with risk of sensitization and exacerbation of asthma in 3% and 13%, respectively, of the sampled homes, while Bla g1 exceeded guidance values in 13% of the homes. Although airborne fungal concentrations in sleep areas and indoors were lower than outdoor concentrations, they exceeded 1 000 colony forming units per cubic meter of air in 29% of the homes.

Table 3 continued ...

Author(s) (year)	Study design	Population/Setting	Sample size	Exposure	Reported associations between HAP and respiratory outcomes
Pelzer et al. (2011)	Cross-sectional study	Secondary analysis of the Global Youth Tobacco Survey conducted in South Africa among current non-smoking school going adolescents (aged 11-18 years). A two-stage cluster sample design was used to produce representative data.	6 412	Exposure assessment data on active and passive smoking was extracted from the Global Youth Tobacco Survey for analysis.	25.7% of students were exposed ETS at home, 34.2% outside of the home. Parental and close friend smoking status, allowing someone to smoke around you and perception that passive smoking was harmful were significant determinants of adolescent's exposure to both ETS at home and outside of the home.
Barnes et al. (2011)	Intervention study	The study took place in two poor rural villages in the North-West Province.	Intervention group (n=36) and control group (n=38)	The study employed a quasi-experimental before and after study design with a control group. Baseline data was collected during winter in both the intervention and control group. Follow-up data was collected from both groups 12 months later. The intervention was implemented immediately after the baseline data collection in the intervention group only. The following were promoted (burn outdoors when possible, if fires are burned indoors, open at least two sources of ventilation during peak emission times, reduce the amount of time that children spend in the burning room while fires are burning).	Both groups reduced indoor air pollution indicators between baseline and follow-up. In the intervention group, the median before-after reduction in PM_{10} equalled 17% (p<.01), CO equalled 11% (p=.15) and CO (child) equalled 47% (p=.02). In the control group, the median reduction in PM_{10} equalled 28% (p=.01), CO equalled 21% (p=.46) and CO (child) equalled 57% (p=.09).

Table 4: *Findings from other recently published studies relevant to our understanding of air pollution and health in South Africa but excluded from this review for reasons stated in the right-hand column. Mostly, these studies did not measure household air pollution exposure but are included here for completeness.*

Author(s) (year)	Study design	Population/ Setting	Sample size	Exposure	Estimates	Missing information
Amegah and Agyei-Mensah (2017)	Narrative review paper	Sub-Saharan Africa.	-	Ambient and indoor air pollution.	South Africa and Accra reportedly have the best air quality programmes in the Sub-Saharan region. Lack of political will is likely to be a major challenge and requires World Health Organization and United Nations Environment Programme to continually engage governments to view air pollution as a major environmental and public health problems.	No exposure measurements or health outcomes included in the review.
Morakinyo et al. (2017)	Human health risk Assessment	Pretoria West area situated near the coal-fired power station, metallurgical industries such as a coke plant and a manganese smelter.	-	Ambient air monitoring of PM_{10}, NO_2, SO_2, CO and O_3, was performed using US Environmental Protection Agency human health risk assessment framework.	Mean annual concentrations for PM_{10}, SO_2 and NO_2 were 48.3±43.4, 18.68±25.4 and 11.50±11.6 µg/m^3, respectively. The mean 24-h CO and O_3 concentrations were 618.30±618.30 and 22.15±7.96 µg/m^3, respectively.	Ambient air pollution monitoring data. No IAQ measurements.
Shirinde et al. (2015)	Cross-sectional study	Children (12-14-years) selected from 16 Thembisa and Kempton Park schools in the Ekurhuleni Metropolitan Municipality.	3 424	Questionnaires to determine the association between eczema and exposure to ETS.	The likelihood of eczema ever was increased by exposure to ETS.	Eczema is not a respiratory health outcome.
Nkosi et al. (2015)	Cross-sectional study	Study on wheeze, asthma, and rhino conjunctivitis associated with community proximity to mine dumps. The study focussed on 13-14-year old pupil who attended schools located between 1-2 km from mine dumps and those living >5 km away from the mine dumps in the Gauteng and North-West province.	3 641	Self-administered questionnaire on asthma and rhinitis (among others) was completed by the adolescent living at a certain distance (within 1-2 km -exposed and >5 km -unexposed) from five pre-selected mine dumps.	Living close to mine dumps had an increased likelihood of current wheeze OR 1.38 (95% CI: 1.10-1.71), rhino-conjunctivitis OR 1.54 (95% Cl 1.29-1.82) and a protective association with asthma OR 0.29 (95% CI: 0.23-0.35).	No exposure measurement. Proximity to mine dump proxy for exposure.

Table 4 continued ...

Author(s) (year)	Study design	Population/ Setting	Sample size	Exposure	Estimates	Missing information
Nkosi et al. (2015)	Cross-sectional study	Elderly persons situated near mine dumps 1-2km (exposed) and 5km (unexposed).	2 397	Questionnaire was used to collect information on respiratory outcomes.	Elderly people living 1-2km away from the mine dumps had a significant association with asthma (OR= 1.57, 95% CI: 1.20-2.05), chronic bronchitis (OR=1.74; 95% CI:1.25-2.39), chronic cough (OR=2.02; 95% CI: 1.58-2.57), emphysema (OR=1.75; 95% CI:1.11-2.77), pneumonia (OR=1.38 95% CI 1.07-1.77) and wheeze (OR =2.01; 95% CI: 1.73-2.54).	No exposure measurement. Proximity to mine dump proxy for exposure.
Jafta et al. (2015)	Review and meta-analysis of case-control studies of children <15 years with TB	For years 1953 – 2014, included articles of children below 15 years exposed to indoor air pollution in relation to TB.	11 studies	Studies published in peer reviewed journals and written in English or translated into English related to laboratory-confirmed childhood TB and exposure to ETS and bio-mass fuel smoke were searched from PubMed, Science Direct and Web of Science.	Exposure to ETS was found to be associated with tuberculosis.	Only two South African articles were reviewed and the main author was based in South Africa.
Reddy et al. (2015)	A multistage stratified cluster South African representative sample of households selected from the South African National Health and Nutrition Examination survey.	Adults (>18 years of age) from 10 000 households in all 9 provinces who participated in National Health and Nutrition Examination Survey (2012).	13 897	Information on smoking habits was extracted from the South African National Health and Nutrition Examination Survey.	17.6% (95% CI: 6.3 - 18.9) currently smoked tobacco. Males (29.2%) had a higher prevalence than females.	No IAQ measurements. Active smoking as a proxy for exposure.
Ayo-Yusuf et al. (2014)	Cross-sectional study	Adults aged ≥16 years who participated in the 2010 South African Social Attitudes Survey.	3 094	Secondary data analyses of samples drawn from the 2010 master sample was conducted. Information on the active and passive smoking was drawn from the survey data.	Overall, 55.9% of all non-smokers reported exposure to second hand smoke from at least one source (i.e., in the home, workplace or at a hospitality venue).	No IAQ measurements. ETS as a proxy for exposure.

Table 4 continued ...

Author(s) (year)	Study design	Population/ Setting	Sample size	Exposure	Estimates	Missing information
Thabethe et al. (2014)	Human health risk assessment	Residents of a low-income community near Secunda in Mpumalanga, South Africa.	Community	PM_{10} monitoring for one month during winter and summer.	The residents were exposed to higher concentrations of PM_{10} during winter (24-h exposure of 157.37 µg/m^3).	Ambient air pollution monitoring data. No IAQ measurements.
Naidoo et al. (2013)	Cross-sectional study	Children from randomly selected Grade 4 classrooms in four areas in South Durban (industrial) and three in North Durban (non-industrial) plus all children from grade 3 - 6 in the same schools who were asthmatic between 2004 and 2005.	341	Interviews with children and their caregivers; spirometry, methacholine challenge tests and skin prick tests were conducted by experienced respiratory technicians on all participants. Those without a baseline obstructive pattern underwent methacholine testing. Ambient monitoring of SO_2, NO_x, $PM_{2.5}$ and PM_{10} was conducted.	The adjusted odds ratios (AORs) were elevated ($p<0.05$) for children in the south for 5 of the 13 outcomes investigated: doctor-diagnosed chronic bronchitis (AOR 3.5, 95% confidence interval (CI) 1.6 - 7.7), as well as bronchitis by symptom definitions; watery/ itchy eyes; wheezing with shortness of breath; and marked airway hyper reactivity (AHR). In addition, marked AHR was associated with SO_2 exposure. The prevalence of symptoms consistent with asthma of any severity was 32.1%. Covariate-adjusted prevalence were higher among children from schools in the south than among those from the north for persistent asthma (12.2% v. 9.6 %) and for marked airway hyper reactivity (AHR) (8.1% v. 2.8%), while SO_2 resulted in a two-fold increased risk of marked AHR (95% CI 0.98 - 4.66; $p=0.056$).	Ambient air pollution monitoring data. No IAQ measurements.
Wichmann and Voyi (2012)	Case cross-over study	All deaths that occurred in the entire Cape Town metropolitan municipality during 2001–2006.	149 667	Investigated all deaths during 2001-2006 in Cape Town to determine the association between average ambient PM_{10}, SO_2 and NO_2 levels and daily respiratory diseases, cardiovascular diseases and cerebrovascular diseases mortality in Cape Town.	Association (excess mortality risk) between PM_{10}, and respiratory disease, cardiovascular disease and cerebrovascular disease mortality of 1.1% (CI −1.1, 3.3), 1.7% (CI −0.1; 3.5 and 3.2% (CI 0.3; 6.2) following a 10 µg/m^3 increase (entire year data) respectively; association between NO_2 and respiratory disease mortality of 1.7% (CI −1.3; 4.7), following a 10 µg/m^3 increase (entire year data) between NO_2, and cardiovascular disease and cerebrovascular disease mortality of 2.6% (CI 0.2; 5.0) and 6.6% (CI 2.4; 11.0) following a 10 µg/m^3 increase (entire year data), respectively.	Ambient air pollution monitoring data. No IAQ measurements.

Table 4 continued ...

Author(s) (year)	Study design	Population/ Setting	Sample size	Exposure	Estimates	Missing information
Reddy et al. (2012)	Cohort study	African children (between 9–11 years old) who formed part of the South Durban Health Study.	129	Questionnaires was administered, information included the presence and severity of asthma, wheezing, coughing, chest tightness, shortness of breath, exposure to cigarette smoke. GSTM1 and GSTP1 genotypes were determined. Intensive monitoring of SO_2, NO_2 and PM_{10} was conducted over a three-week period.	Relatively modest, and not entirely consistent, interaction effects of the GSTM1 and GSTP1 polymorphisms were found. Among the 24 pollutant-lag combinations examined, for GSTM1, in only one of the four that showed statistically significant pollutant-genotype interactions, was the effect in the expected direction of those with GSTM1 null having greater pollution associated increases in FEV1 intraday variability.	Ambient air pollution monitoring data. No IAQ measurements.
Kistnasamy et al. (2008)	Cross-sectional study	The study was conducted at the primary schools situated near industrial areas, the prevalence of symptom-defined asthma and nonspecific bronchial hyper-reactivity was examined.	248	Ambient monitoring of SO_2, oxides of nitrogen, and PM_{10}. Questionnaire was also used to collection information on the number of smokers in the participants' households.	The odds of developing bronchial hyper-reactivity and persistent asthma was 0.64 (95% CI: 0.32 - 1.28) and 1.13 (95% CI: 0.61 - 2.09), respectively among the children with one or more smokers compared to children who lived in households with non-smokers.	Ambient air pollution monitoring data. No IAQ measurements.