Clean Air Journal

ISSN 1017 - 1703
Vol 33 No 2
November / December 2023

Official publication of the
National Association for Clean Air

CLEAN AIR JOURNAL

ISSN 1017-1703
November / December 2023
Volume 33, No. 2

Published twice yearly by the National Association for Clean Air, Republic of South Africa

www.cleanairjournal.org.za | info@cleanairjournal.org.za

Contents

Our vision for a Just Energy Transition enables job creation and growth

Our JET strategy has five underlying principles

Balancing all elements of energy trilemma: energy security, access, and sustainability

Decoupling Repurposing and Repowering initiatives from decommissioning schedules

Embodying principles of Distributive, Restorative and Procedural Justice

5 Principles of Eskom's Just Energy Transition

Making decisions flexibly and dynamically to respond to changing context

Aligning with both national and Eskom priorities to ensure success

Our JET vision will enable growth, create jobs and increase competitiveness in a future net-zero world

Demand side response

Distribution grid enabled for consumer responses to fluctuations in supply, and for sale of consumer-owned RE back to the grid

Firm capacity

Wind and PV constructed with storage, and grid network adapted to increase stability

Seasonal peaking power

Gas used as transitional peaking fuel until cleaner technologies emerge (e.g. SMR nuclear)

Daily peaking power

Batteries and pumped hydro for short term power storage

Trade

Export industries receive low-carbon power, allowing them to complete globally despite globally increasing carbon-border taxes

Older generation

Residual traditional nuclear and newer coal plants maintained until replaced, but ultimately phased out

Decommissioned coal plants **repurposed** (e.g. agrivoltaics, training facilities, new manufacturing centres etc.) or **repowered** (into new RE plants) bringing new, jobs to communities

Industry

Industry and commerce grows with access to reliable, abundant and affordable power

Socio-Economic Development

New training centers prepare the workforce for new jobs created in the economy, supporting innovative small businesses creating local value

Our JET strategy will create new, high quality jobs in South Africa, such as:

RE plant operator operates and maintains power producing equipment

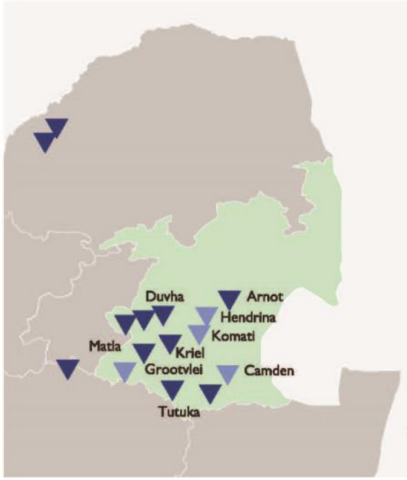
Solar water system plumber installs, repairs, and maintains water systems

Laboratory assistant supports tests and research for bioenergy solutions

Horticulture specialist helps develop agrivoltaics, aquaponics, hydroponics

Eskom is making progress towards a Just Energy Transition for South Africa

We will repower and repurpose to create local opportunities



Starting with Mpumalanga's oldest coal plants, we will...

Repower coal plants by leveraging existing infrastructure to build new generation capacity including solar, wind, gas turbines, or synchronous condensers

Repurpose coal plants into new centres of economic activity including training centres, water treatment facilities, manufacturing plants, microgrid assembly and modern farms

Unlock transmission and distribution capacity by investing in sub-stations and transformers in a province where solar resources are high by global standards

Ensure a Just Transition for local communities through the Repurposing and Repowering activities that will help to retain economic activity, creating jobs and new economic opportunities

▼ Coal plant operation post 2035 ▼ Coal plant to be R&R'ed by 2035

The largest just repowering and repurposing project in the developing world is already underway at Komati Power Station

Three projects in progress

- **Komati Training Facility** capacitates local community and Eskom with renewable energy capabilities and diversified jobs
- **Agrivoltaics Plant** combines solar power, battery storage, and agricultural activities, acting as an incubator for agriculture SMMEs
- **Containerised Microgrid Assembly** creates a broad array of new jobs, including welders, painters, electricians and more

New project in development

- **Copper Recycling Facility** extracts copper from existing coal plants to recycle and beneficiate for use in renewable energy

Impact by 2030

~660 **Direct, sustainable jobs** created in repowering and repurposing work at Komati Power Station

~8700 **Additional temporary jobs** created mainly in work related to construction and decommissioning

~300 **MW Generation Capacity** added, replacing 100 MW of Komati operational capacity when decommissioned

~R1.3b **Total contribution so South Africa's economy** from Komati Repurposing and Repowering can be achieved

~$0.5b **Funding** secured from the World Bank for Komati Repurposing and Repowering initiatives

***Note:** Precise figures may vary between now and 2030*

Eskom

Editorial for special issue
Introduction to special issue ICAQ'AFRICA2022 Conference: Trends and perspectives in air quality research in Africa

Bertrand Tchanche[1], Khanneh Wadinga Fomba[2], Wahid Mellouki[3]

[1]Alioune Diop University, Senegal
[2]Leibniz Institute for Tropospheric Research, Germany
[3]University Mohammed VI Polytechnic (UM6P), Morocco

https://doi.org/10.17159/caj/2023/33/2.17373

Air pollution is an emerging concern in many African countries transcending geographical boundaries and affecting the well-being of its communities due to the increase in population and industrial activities that control different pollution sources as well as the impact of climate change on various landscapes affecting traffic, residential combustion, and power generation emissions (Mead et al, 2023). African cities are growing at a fast pace (around 5% per annum urbanization rate in many places) and this growth is like those seen in emerging regions like South America and Southeast Asia. Africa now hosts megacities such as Kinshasa in the Democratic Republic of Congo, Lagos in Nigeria, and Cairo in Egypt. However, this development is followed by several challenges in water and electricity supply, municipal waste management, a growing need for infrastructure (hospitals, schools, etc.), and deteriorating air quality with adverse consequences on health.

In this light, the African Society for Air Quality (ASAQ) organized its first International Conference on Air Quality in Africa (ICAQ'22) to serve as a catalyst for collaborative research and knowledge exchange and called for papers that could highlight the state-of-the-art in the different specific research areas within the continent (Tchanche et al., 2022). The event convened researchers from various parts of the world delivering around 70 talks covering several topics including atmospheric pollutant monitoring and forecasting models, microplastics, the impact of atmospheric pollution on climate and economy, remote sensing, air pollution and health effects, urban heat island, and mitigation strategies.

The collective knowledge shared at the conference contributes to a deeper understanding of air quality challenges in Africa and fosters collaborative efforts toward sustainable solutions. The articles presented in this Special Issue delve into the intricate dynamics that shape the air quality landscape across the continent and shed light on some emission problems, key management challenges, and potential solutions that could empower both the private and public sectors in developing air pollution mitigation strategies.

Nana et al. (2023) examine the carbon dioxide emission factors stemming from power generation focusing on the year 2018, which is used as an example to evaluate the contribution of renewable energy sources in mitigating CO_2 emissions. The findings emphasize the need for a nuanced understanding of emission factors and highlight the role of renewable energy in fostering sustainable power generation practices.

Faboya et al. (2023) use Lagos as an example of a specific urban area, where anthropogenic activities strongly contribute to the exceedance of $PM_{2.5}$ levels, and further quantify specific chemical components in these particles, especially, aliphatic hydrocarbons, evaluating source diagnostic indices, offering crucial insights into the local air pollution challenges. The conclusions drawn pave the way for targeted management strategies tailored to address the specific dynamics of air quality in Lagos.

Finally, a holistic approach is provided by Borge et al. (2023), on how air quality issues in five major African cities are managed. Through an exploration of urban air quality, monitoring capabilities, emission inventories, and governance structures, the study emphasizes the urgent need for comprehensive air quality management strategies. It further highlights the

intertwined nature of poverty, social equity, and environmental health, calling for collaborative, multi-level interventions in addressing air quality issues.

While these articles contribute significantly to the dialogue on air quality in Africa, it is imperative to highlight certain overarching themes emerging from ongoing research in the field of atmospheric science on the continent involving:

Source apportionment and chemical composition: Many studies have presented particulate matter concentrations, revealing widespread pollution across African cities. However, few studies have provided the sources and their chemical composition. Future research could delve more into source apportionment and the chemical composition of collected samples, and those of gases.

Data availability challenges: Ongoing atmospheric science studies in Africa consistently indicate poor air quality in major cities, however, data from many locations are still unpublished. The high upfront cost of air quality monitoring, coupled with low awareness among decision-makers and the public, and a shortage of well-trained personnel, pose significant obstacles to the availability of air quality data.

Role of low-cost sensors: A notable number of studies revolve around low-cost sensors as a potential solution to overcome the lack of air quality data in Africa. These sensors offer a promising avenue for cost-effective monitoring, potentially increasing data availability across the continent.

Integrating remote sensing data: Although air quality data retrievals via remote sensing using satellites are available, very few works have been published on this topic. Future research can encourage their application especially due to their ability to provide valuable insight's into air quality data and dynamics currently limited.

Health implications: Atmospheric pollutants are significant contributors to several diseases, including asthma, cancer, stroke, and dementia, leading to morbidity and premature mortality. Despite progress, more research in this direction will benefit public awareness of curbing air pollution.

Policy and legislation: While legislation related to air pollution exists in many African countries, enforcement remains a challenge. The need for more countries to adopt and rigorously enforce air quality policies is important to enhance the collective effort in combating this challenge.

Urban heat islands: Heat islands negatively affect productivity and the economy in most African cities, particularly those located in tropical areas. There is a need for researchers to study the effects of heat islands in African cities and explore strategies, such as cool roofs, to mitigate these impacts.

Collaboration and capacity: Most African researchers are isolated and work with limited access to funds. South-South collaboration and more training in the field of air quality are highly recommended for capacity building. Nevertheless, the interest is growing in air quality research in Africa. The number of research groups as well as the number of published papers is increasing rapidly. More and more scientific events are being organized on the topic in various parts of Africa. After the ICAQ'Africa 2022, more than ten events were organized on the continent focusing on different aspects of air quality.

Collectively, the interdisciplinary nature of the ongoing research underscores the pressing need for policymakers, researchers, and stakeholders to collaborate in devising sustainable strategies for addressing air quality challenges in Africa. The complexity of the issue also necessitates international cooperation to build a healthier and sustainable future for the continent. It is anticipated that air quality research in Africa will continue to grow and improve in the upcoming years.

References

Mead, M.I., Okello, G., Mbandi, A.M. and Pope F. (2023): Spotlight on air pollution in Africa, *Nat. Geosci*. 16, 930–931. https://doi.org/10.1038/s41561-023-01311-2

Tchanche, B., Fomba K.W., Mellouki W., Westervelt D.M. and Giordano M. R. (2022): Scientists discuss the state of air quality research in Africa during the First International Conference on Air Quality in Africa – ICAQ'AFRICA2022, *Clean Air Journal* 32 (2). https://doi.org/10.17159/caj/2022/32/2.15243

Nana, B., Zalle, H., Ouarma, I., Daho, T., Yonli, A. and Bere, A. (2023): Assessment of carbon dioxide emission factors from power generation in Burkina Faso, *Clean Air Journal* 33 (2). https://doi.org/10.17159/caj/2023/33/2.15701

Faboya, O.L., Fomba, K.W., Ezeh, C.G. and Herrmann H. (2023): Assessing anthropogenic contribution to $PM_{2.5}$ from an urban residential area of Lagos, Nigeria using aliphatic hydrocarbon compounds as indicators, *Clean Air Journal*, 33 (2). https://doi.org/10.17159/caj/2023/33/2.15790

Borge, R., Lange, S. and Kehew, R. (2023): Analysis of air quality issues and air quality management status in five major African cities, *Clean Air Journal*, 33 (2). https://doi.org/10.17159/caj/2023/33/2.15605

REGISTRATION WITH

SACNASP
South African Council for Natural Scientific Professions
Integrity in science

01 LEGAL COMPLIANCE NSP ACT (27 of 2003)

02 RECOGNITION AS A PROFESSIONAL

03 PUBLIC CONFIDENCE IN YOU AS A SCIENTIST

04 MARKETABILITY (EMPLOYERS REQUIRE REGISTRATION)

05 CODE OF CONDUCT – TRUST FOR ETHICAL VALUES

06 INPUT TO GOVERNMENT – SACNASP VOICING SCIENTISTS' INPUT AT MINISTERIAL LEVEL

07 NETWORKING – WEBINARS, WEBSITE NETWORKING PORTAL

08 POTENTIAL FAVOURABLE RATES FOR PROFESSIONAL INDEMNITY INSURANCE

09 CAREER ADVERTISEMENTS – EMPLOYERS ADVERTISE VACANCIES ON SACNASP WEBSITE AND SOCIAL MEDIA

10 VOLUNTARY ASSOCIATION EVENTS – NETWORK WITH FIELD OF PRACTICE PEERS AND GAIN VOCATIONAL CAREER ADVICE

11 FACILITATES LIFELONG LEARNING THAT IS CRUCIAL TO A PROFESSIONAL'S CAREER PATH – CANDIDATE MENTOR PROGRAMME

12 CONTINUING PROFESSIONAL DEVELOPMENT – ONLINE LEARNING

Suite L4 and U13, Enterprise Building
Innovation Hub, Pretoria
Gauteng

Tel: +2712 748 6500

www.sacnasp.org.za

Commentary
Imagining an air quality framework that works: How do we mainstream offsets?

Phaṭhutshedzo Mukwevho[1], Roelof Burger[1], Stuart Piketh[1], Niké Jacobs[1], Brigitte Language[1], Henno Havenga[1] and Daniël Düring[1]

[1]Climatology Research Group, School of Geo- and Spatial Science, Unit for Environmental Science and Management, North-West University, Potchefstroom 2520, South Africa

https://doi.org/10.17159/caj/2023/33/2.17252

Air quality management (AQM) in South Africa faces significant challenges in balancing the constitutional right to clean air with economic development pressures. Civil organisations and communities demand their constitutional right to clean air while local governments face other pressing priorities, such as basic services and economic development (Naiker et al. 2012; Garland 2023). Therefore, imagining an air quality management framework that works within the civil and political context of South Africa is deemed imperative.

Current air quality management relies heavily on policy and legislation with robust compliance and enforcement measures dominated by Command and Control (CaC) approaches. This approach lacks innovation and does not encourage incentive-driven thinking (Mukwevho et al. 2022). There is an excessive reliance on legal recourse as the primary incentive to comply, which may not be sustainable or encourage proactive measures from industry. While recognising the significant role of CaC approaches, it is recommended to explore and adopt additional approaches beyond CaC for further enhancement (Mukwevho et al. 2022), which may lead to the industry sharing the responsibility with the government of ensuring clean air for all. The government has initiated adopting alternative approaches like market-based instruments, including a carbon tax. However, these measures can intensify financial burdens on industries, thereby complicating efforts to reduce emissions in a manner that is considerate of socio-economic development needs.

In the context of sustainable development, air quality offsetting emerges as a promising approach to incentivise emission reduction while supporting economic growth. In South Africa, offsetting is implemented by polluting industries as a condition of postponement of compliance with the Minimum Emissions Standards (Langerman 2019). This takes the form of interventions to reduce poor air quality in low-income settlements by focussing on counterbalancing adverse environmental impacts without requiring that the offset reduces the same pollutant emitted by the industry (DEA 2016; Langerman 2019). It is argued that organisations are more inclined to engage in environmental initiatives when these offer incentives for specific objectives, and they are open to modifying abatement activities of the facilities and behaviours within the business to align with these objectives (Frondel et al. 2008). An unintended consequence is that offsets are typically seen as stopgap measures with fixed and limited time frames as opposed to control technology projects with significant capital investment for development and implementation lifetimes in the order of decades. Control technology subsequently disincentivises the industry to take offsets seriously.

Therefore, the questions that we are asking are: 1) What are the benefits of air quality offsets and 2) To what extent should offsets be integrated into our existing CaC instruments?

First, we must appreciate the complexity of air pollution in its totality. We need to acknowledge that air quality management is not only about industrial air pollution, as is commonly perceived, but we also need to consider the significance of other major pollution sources such as waste, domestic combustion and traffic. Many households in South Africa, especially those in low-income settlements, are exposed to indoor PM exceeding ambient PM concentrations (Language et al. 2016; Morakinyo & Mokgobu 2022; Wernecke et al. 2015). Adesina et al. (2020) illustrated that increased indoor and ambient PM_4 concentrations during the winter primarily result from solid fuel combustion and not from nearby power plants.

Secondly, we need to recognise the potential benefits of air quality offsets and explore practical examples that have proven beneficial. Adesina et al. (2020), Langerman et al. (2018) and Murray et al. (2023) argue that the introduction of air quality offset initiatives is a valuable augmentation to the arsenal of strategies aimed at reducing emissions, as it brings in additional contributors and financial resources to support the drive towards clean air. In addition, offsets represent a cost-efficient means of directing limited resources toward mitigating exposure to harmful pollutant levels.

The complex nature of air pollution and the necessity to evaluate the impact of sectoral mitigation measures on pollution levels and socio-economic factors calls for an integrated approach to air quality management. Such an approach should be underpinned by a rigorous scientific method, particularly in the prioritisation of offsets and the determination of activities that are eligible for offsets, ensuring optimisation across several dimensions, including:

- Reduction of emissions,
- Improvement of ambient air quality,

- Minimisation of economic costs,
- Generation of co-benefits, and
- Enhancement of health outcomes.

A key tool for such optimisation emerges as Integrated Assessment Modelling (IAM). Ideally, through IAM, the intricacies of management measures and their impacts, including offsets, can be assessed and optimised. In doing so, activities qualifying for offsets can be identified and directed toward the focus of interventions.

In practical terms, this approach would enable a strategic reallocation of resources to areas where the most significant environmental and health benefits can be realised, concurrently ensuring economic efficiency. The industry could then opt to mitigate their emissions to the techno-economically viable extent and utilise offsets to compensate for any shortfall in meeting minimum emissions standards.

Adesina et al. (2020) argue that full implementation of the air quality offset programme could yield significant advantages for local communities, including reducing indoor PM concentrations and enhancing thermal comfort, particularly during the winter months. Evidence suggests that these mitigation methods driven by offsets may enhance their overall quality of life (Langerman 2019).

Our commentary suggests that the current process undertaken by the Department of Forestry, Fisheries and the Environment (DFFE) to review the *National Framework for Air Quality Management* presents the best opportunity to integrate air quality offsets into the CaC legal framework. The core aim of the framework is to identify mechanisms, systems, and processes that promote an integrated approach to air quality management. This encompasses tactics for preventing and decreasing pollution at its origin and managing its environmental consequences, extending from local to global issues. Integrating air quality offsetting into the national framework, therefore, is advisable, seeing the current South African air quality management system faces various challenges, including a fragile economy, limited lifetime of infrastructure, the short timeframe of extensions of compliance with the minimum emission standards and the focus on reduction of emissions. Air quality offsets provide an opportunity to holistically enhance quality of life whilst combating poor air quality.

How can offsets become a legitimate part of an air quality management framework? Offsets integration can be done in a manner that requires it to prioritise emission reduction, maintenance, co-benefits, and contribution to ambient air quality improvement. For offsets to be viable, there needs to be clear strategies which will ensure that realistic programs, long-term implementation, regular maintenance (funded by the implementing industry), monitoring, and independent evaluation of their performance are conducted. Lastly, the option to implement air quality offsets, particularly by industry, should not take away their responsibilities to meet minimum emission standards but provide an additional tool to ensure clean air for all.

References

Adesina, J.A., Piketh, S.J., Qhekwana, M., Burger, R., Language, B. and Mkhatshwa, G. 2020. Contrasting indoor and ambient particulate matter concentrations and thermal comfort in coal and non-coal burning households at South Africa Highveld. *Science of the Total Environment*, 699: 134403.

Adesina, J.A., Piketh, S.J., Qhekwana, M., Burger, R., Language, B. and Mkhatshwa, G. 2020. Quantifying the effect of air quality offsets on household air pollution and thermal comfort on the South Africa Highveld. *Clean Air Journal*, 30:1.

DEA (Department of Environmental Affairs). 2016. Air quality offsets guideline. *Government Gazette* no.39833.

Frondel M., Horbach J., Rennings K. 2008, What triggers environmental management and innovation? Empirical evidence for Germany. *Ecological Economics*, 66: 153 – 160.

Garland, R. 2023. South African court rules that clean air is a constitutional right: what needs to change. https://theconversation.com/south-african-court-rules-that-clean-air-is-a-constitutional-right-what-needs-to-change-179706 Date of access: 26 October 2023.

Langerman, K.E. 2019. An Environmental Justice Perspective on Air Quality Offsets. *Clean Air Journal* 29(2), pp.1-14.

Langerman K., Pauw CL, Smith HJ, and Piketh SJ. 2018 Moving Households to Cleaner Energy through Air Quality Offsets. *International Conference on the Domestic Use of Energy (DUE)* (pp. 1-8). IEEE

Language, B., Piketh, S.J., Wernecke, B. & Burger, B. 2016. Household Air Pollution in South African Low-income Settlements: A Case Study. *In* WIT Transactions on Ecology and the Environment. WIT Press.

Mukwevho, P., Retief, F. and Burger, R 2022. Conceptualising air quality management instruments in South Africa. *Clean Air Journal*, 32(2), pp.1-15.

Murray M., Pauw C., and Smith H 2023. Evaluation of a Household Intervention Implementation in the Sasol Secunda Air Quality Offset Programme. *Annual Conference of the National Association for Clean Air 2023*.

Naiker, Y., Diab, R.D., Zunckel, M. & Hayes, E.T. 2012. Introduction of local air quality management in South Africa: overview and challenges. *Environmental Science & Policy* 17:62-71.

Morakinyo, Q.M. & Mokgobu, M.I. 2022. Indoor Household Exposures and Associated Morbidity and Mortality Outcomes in Children and Adults in South Africa. *International Journal of Research and Public Health* 19(15): 9471.

Wernecke, B., Language, B., Piketh, S.J. & Burger, R.P. 2015. Indoor and outdoor particulate matter concentrations on the Mpumalanga highveld – A case study. *Clean Air Journal* 25(2).

MINTEK - Analytical Chemistry Division

Analytical Chemistry Division (ACD) is a world-class leader in research, development & innovation (RD&I) which provides capabilities for elemental analysis and solutions using ultra-modern instruments in the field of minerals. ACD provide specialized solutions to metallurgical, mining, environmental and related industries.

ACD is an ISO/IEC 17025 accredited laboratory for several methods that supports research and commercial work with complex analysis for metallurgical, environmental and geochemical materials such as:

- Base metals (i.e. Hg, TCLP, S)
- Water Analysis
- Rare Earth Elements (REEs)
- Precious Group Metals (PGMs) by Novel Fire Assay Technology

Global leader in the Production of Certified Reference Materials (CRMs) under South African Reference Material (SARM) program.

Contact us:

Ms. Dorcas Hanyane

asdenquiries@mintek.co.za

+27 11 709 4209/4046

Commentary
Alternative carbon-based fuels

Hein Neomagus[1*], John Bunt[1] and Nthabiseng Leokaoke[1]

[1]Centre of Excellence in Carbon-Based Fuels, North-West University, Potchefstroom Campus, Private Bag X6001, Potchefstroom 2520, South Africa

*Corresponding author: hein.heomagus@nwu.ac.za

https://doi.org/10.17159/caj/2023/33/2.17302

The need for alternate carbon-based fuels

The pressing global challenge of climate change has intensified the search for sustainable and environmentally friendly energy sources. Among the potential solutions, the need for alternate carbon-based fuels has gained prominence as a critical avenue for reducing greenhouse gas emissions and mitigating the impact of fossil fuel consumption. Traditional carbon-based fuels, such as coal, oil, and natural gas, have long been the primary drivers of economic development and industrialization. However, their combustion releases vast amounts of carbon dioxide (CO_2) into the atmosphere, contributing significantly to the greenhouse effect and global warming.

To address this issue, scientists, researchers, and policymakers are exploring alternative carbon-based fuels that offer a more sustainable energy future. One promising category of alternate carbon-based fuels includes biofuels derived from organic materials, such as plant biomass or algae. Unlike fossil fuels, biofuels can be considered carbon-neutral because the plants used in their production absorb CO_2 from the atmosphere during their growth. This closed carbon cycle helps offset the emissions released when the biofuels are burned for energy. Another avenue involves developing synthetic fuels, often referred to as e-fuels or electrofuels, which are produced by using renewable energy to convert CO_2 and water into hydrocarbons or other fuel molecules. These synthetic fuels can be used in existing combustion engines or fuel cells, providing a potential bridge between current infrastructure and a more sustainable energy system.

The need for alternate carbon-based fuels is underscored by the limitations and challenges associated with widespread adoption of renewable energy sources like solar and wind. These sources are intermittent and dependent on weather conditions, making energy storage and distribution challenging. Carbon-based fuels, whether derived from biomass or synthesized through renewable processes, offer a more reliable and flexible energy solution that can complement intermittent renewables. Moreover, transitioning to alternate carbon-based fuels can help reduce dependence on geopolitically sensitive regions for traditional fossil fuel sources. This diversification enhances energy security and reduces the geopolitical tensions associated with access to conventional energy resources. This commentary provides some insight into our focus on alternate carbon-based fuels development at the North-West University.

Green coal

A new formulation of 'green coal' has been developed, and its application has been tested at the coal research laboratories of North-West University. Bio-oil and biomass char were produced from sweet sorghum bagasse using hydrothermal liquefaction (HTL) at temperatures between 280 and 300°C. The resulting char was mixed with medium-rank C bituminous discard coal fines and calcium carbonate ($CaCO_3$) in different ratios, and the mixtures were pelletized and gasified using CO_2 at temperatures up to 1000°C. The experimental data revealed that the reaction rate of the biochar was significantly higher than that of raw coal.

To further examine the impact of temperature and catalyst on elemental sulphur retention during pellet combustion, a combustion setup was used, showing low sulphur retention for raw coal and biochar blends. However, the addition of a metal catalyst/sorbent increased sulphur retention to 56-86%, decreasing with rising temperature. A simulation using FactSage™ predicted that operating the fixed-bed gasifier in catalytic gasification mode at 800°C would remove over 50% of the pyritic sulphur as insoluble $CaSO_4$ from the gaseous phase. This aligns with the experimental results, and six international patents have been filed for this innovative process.

Fine discard coal briquetting

South Africa produces approximately 30 million tonnes of coal fines annually, which can lead to dust release, acid mine drainage and spontaneous combustion. Agglomeration technologies present a unique solution, allowing for fine coal use in conjunction with other waste materials. The North-West University has evaluated briquetting and extrusion as possible agglomeration techniques, looking at co-utilisation, waste valorisation and binder optimisation. Binder costs are often the cause for cost impracticality in agglomeration, hence binders derived from waste plastics along with low concentration binders (poly-acrylic materials) have been tested.

Agglomerates produced were found to have compressive strengths double that of the binderless briquettes, and in some cases surpassing the compressive strength of run-of-mine coal (14 MPa). These agglomerates can potentially be used in fixed-

bed gasification or chain grate stoker fired boiler combustion applications. Over the course of the years, the testing facilities at the university have grown enough to allow for the continuous production (25 – 200 kg/hr) and testing (mechanical as well as thermal) of the manufactured agglomerates.

Waste valorisation

The green coal initiative has opened avenues for innovative fuel formulations. One promising approach involves extruding coal fines with recycled low-density polyethylene (LDPE) and polypropylene (PP) as binders. The binder content varied from 5 to 100 wt%, and the mixtures were processed in a co-rotating twin-screw extruder at 220°C. This novel method, not explored previously, resulted in extrudates with varying mechanical strength. Those containing 10% or more binder content exhibited strength and homogeneity, while a 5% binder proved insufficient. Under load, the extrudates with 10% LDPE and 10% PP displayed compressive strengths of 17.3 and 5.9 MPa, respectively, before breaking. The extrudates, regardless of binder content, showed consistent compressive strength after exposure to water. Moreover, they absorbed less than 5% water after 24 hours of submersion. As the binder concentration increased, the calorific value of the extrudates significantly rose due to the higher calorific values of LDPE and PP compared to coal fines. Additionally, higher binder content led to a decrease in sulphur content. Thermogravimetric analysis (TGA) revealed synergy between plastics and coal fines during pyrolysis. The study suggests that co-extrusion of recycled plastic with coal fines can yield carbonaceous fuels with enhanced hydrophobicity, heating value, and strength compared to using coal fines alone. This approach not only addresses environmental concerns but also utilizes waste coal fines and plastics for industrial applications.

In a related study, the slow co-pyrolysis behaviour of extrudates from South African Highveld coalfield coal fines and recycled LDPE and PP was investigated. Plastic fraction varied from 10 to 100% during extrusion. Slow pyrolysis yielded up to 83% char from coal fines and over 90% condensable products from plastics. The slow heating rate favoured condensable product production, increasing with plastic concentration. Equations were developed to predict pyrolysis yields and characteristics based on coal and plastic composition. The results indicate potential applications in stand-alone pyrolysis plants for downstream chemicals production and char gasification, marking a step forward in alternate fuel technology.

Cook-stove emission reduction

The use of coal as a domestic fuel source still occurs and is linked to poor health due to exposure to household air pollution. Alternative carbon-based fuels have been considered for use in a newly designed cook-stove, with the aim of increasing the air quality in households where electricity driven equipment is not always feasible. Coal-biomass pellets and lump coal were combusted in this newly designed semi-continuous cook-stove at the North-West University. Biomass addition generally leads to lower sulphur emissions and ash generation, and at low concentrations (up to 25 wt%), a reduction in particulate matter emissions was also observed. The use of torrefied wood was shown to increase the potential energy content of the fuel, approaching the calorific value of low-grade coal. To improve handleability, binders such as starch and PVA were utilised, and were found to have no significant influence on the NO_x or SO_x emissions of the pellets. Finally, it was found that all the fuel batches in combination with the improved semi-continuous coal stove delivered significantly lower NO_x, SO_x, and CO emission levels than when compared to conventional coal stoves using lump coal. A roll out of 50 coal stoves in a low-income settlement has been concluded, and the improvement of air quality is now being evaluated. This stove can also be fed with blended coal/biomass pellets, which can potentially make the process more sustainable. Modifications to the stove design are currently underway.

Conclusion

The imperative need for alternative carbon-based fuels arises from the urgency to address climate change, reduce greenhouse gas emissions, and transition towards a more sustainable energy landscape. Biofuels and synthetic fuels represent promising pathways that align with the goals of mitigating environmental impact, fostering energy security, and promoting technological innovation for a cleaner, more resilient future. This commentary has provided some insight into our focus on alternative carbon-based fuels at the North-West University with the view to lower emissions and improve process efficiency and sustainability.

GAS ANALYSIS

The National Metrology Institute of South Africa is mandated by the Measurement Units and Measurement Standards Act, 2006 (Act No. 18 of 2006) to provide accurate and internationally recognised national measurement results. The NMISA Gas analysis laboratory enables compliance to legislation and reliable reporting of emission measurements through the provision of traceable and reliable primary reference gas mixtures. The laboratory is responsible for the dissemination of primary reference gas mixtures, provision of reference measurements and calibration services, including training, proficiency testing schemes, assessments and consultation.

Our Services

- **Primary reference gas mixtures**
  - Criteria air pollutants
  - Greenhouse gases
  - Reactive gases
  - Volatile organic compounds
  - Oxygenated volatile organic compounds
  - Energy gases
  - Stack emission gases
  - Automotive exhaust gases
- **Purity Analysis for high purity gases**
- **Reference measurements**
  - Certification of gas mixtures from local gas manufacturers (Secondary reference materials)
  - Analysis of tedlar bags and other sampling vessels
- **Proficiency Testing Schemes**
  - Ambient air monitoring
  - Stack emission monitoring
- **Calibration of ozone instruments**
- **Calibration of gas monitors**
  - Personal monitors
  - Gas leak detectors
- **Calibration of evidential breath analyser (EBA) for law enforcement**

Please contact us at **gas@nmisa.org** or **+27 12 947 2818**

nmisa
National Metrology Institute of South Africa

Research Projects

Environmental and ambient air gases

Development of the following reference gas mixtures:

- **Greenhouse gases**
  - Carbon dioxide, methane, Sulphur hexafluoride, nitrous oxide
- **Low amount fraction reactive gases**
  - Nitrogen monoxide, nitrogen dioxide, hydrogen sulphide
  - Preparation of calibration gas mixtures using dynamic volumetric methods
- **Volatile organic compounds**
  - Non-methane hydrocarbons (NMHCs), Hazardous Air Pollutants (HAPs), oxygenated VOCs, benzene, toluene, ethyl benzene, (o,m,p) xylenes
- **Sulphur compounds**
  - Sulphur dioxide, Ethyl mercaptans, Dimethyl sulphide, Tetrahydrothiophene

Industrial emission and energy gases

Development of the following reference gas mixtures:

- **Stack emission gases**
  - Nitrogen monoxide, sulphur dioxide, carbon dioxide
- **Automotive exhaust gases**
  - Carbon monoxide, carbon dioxide, propane, oxygen
- **Natural gases**
  - Ethane, propane, n-butane, n-pentane, i-pentane
- **Refinery gases**
  - Carbon dioxide, carbon monoxide, methane, ethane, propane, 1,3-butadiene, oxygen nitrogen and helium (balance)
- **Development of the Biogas gas mixture**
  - Methane, carbon dioxide and hydrogen sulphide

News
Let's write in our African languages: Clean Air Journal's inclusive language policy

Mbalenhle Mpanza[1]
[1]University of Johannesburg, South Africa

https://doi.org/10.17159/caj/2023/33/2.17266

One way of taking action to improve air quality is to disseminate knowledge and educate people using languages that the majority of people understand. This is another form of citizen science, which allows individuals from diverse backgrounds to actively participate in scientific research, thereby generating new knowledge and understanding (Cardamone and Lobel, 2016).

The Clean Air Journal has introduced an inclusive language policy (https://cleanairjournal.org.za/InclusiveLanguage) which aims at improving the accessibility of research published in the journal. The policy allows for additional abstracts, written or narrated in any African language, to be published. An African language is considered to be any of the over 2000 languages spoken in Africa.

Several authors have contributed additional abstracts to the Clean Air Journal. Audio abstracts have been submitted by Bianca Wernecke for Wernecke et al. (2021) (https://cleanairjournal.org.za/article/view/9426) and by Rebecca Garland for Borduas-Dedekind et al. (2023) (https://cleanairjournal.org.za/article/view/15367). Dingemanse et al. (2022) have translated their abstract into Amharic, an official language of Ethiopia (https://cleanairjournal.org.za/article/view/13470/20567). Pieter Van Zyl submitted an Afrikaans abstract, for Swarz et al. 2022 (https://cleanairjournal.org.za/article/view/12866/20599).

Authors wishing to submit a translated or verbal abstract should express their intent to do so when their manuscript is accepted for publication. Accompanying the translated abstract should be a certification of translation (see the example at https://cleanairjournal.org.za/InclusiveLanguage). The certification of translation serves to confirm that the abstract (written or verbal) is an accurate translation of the content in the English abstract.

References

Borduas-Dedekind, N., Naidoo, M., Zhul, B., Geddes, J., and Garland, R. M. (2023) Tropospheric ozone (O_3) pollution in Johannesburg, South Africa: Exceedances, diurnal cycles, seasonality, O_x chemistry and O_3 production rates. *Clean Air Journal*, 33(1). Available at https://cleanairjournal.org.za/article/view/15367 (accessed 14 November 2023).

Cardamone, C N., and Lobel, L K. (2016). Using Citizen Science to Engage Introductory Students: From Streams to the Solar System. *J Microbiol Biol Educ*. 17(1): 117–119.

Dingemanse, J.D., Abiyu, M.A., Tesfaye, K.G., Roro F.F. (2022). Using student science to identify research priority areas for air pollution in a university environment: an Ethiopian case study. *Clean Air Journal*, 32 (2). Available at https://cleanairjournal.org.za/article/view/13470/20567 (accessed 14 November 2023).

Wernecke B., Burger R.P., Language, B., Wright, C.Y., and Piketh S.J. (2021). Quantifying potential particulate matter intake dose in a low-income community in South Africa. *Clean Air Journal*, 31 (2). Available at https://cleanairjournal.org.za/article/view/9426 (accessed 14 November 2023).

Swartz, J.-S., van Zyl, P. G., Beukes, J. P., Galy-Lacaux, C., Labuschagne, C., Brunke, E.-G., Mkololo, T., & Pienaar, J. J. (2022). Wet season chemical composition of atmospheric wet deposition at Cape Point . *Clean Air Journal*, 32(1). https://doi.org/10.17159/caj/2022/32/1.12866 (accessed 15 November 2023)

News
NACA conference 2023 – Collaborating for clean air: confronting climate change and air quality together

Danitza Klopper[1] and Roelof Burger[2]

[1]Chairperson of the NACA Scientific Review Committee, Department of Geography and Environmental Studies, University of Limpopo
[2]NACA President, Unit for Environmental Science and Management, North-West University

https://doi.org/10.17159/caj/2023/33/2.17004

The 2023 Annual NACA Conference, held in collaboration with the Department of Forestry, Fisheries, and the Environment's (DFFE) Air Quality Governance Lekgotla, proved to be a resounding success. This in-person gathering took place at the picturesque Protea Hotel: Ranch Resort, just south of Polokwane, drawing 141 attendees. All sessions were also streamed virtually, free of charge to all stakeholders who couldn't join the face-to-face event. The conference not only provided a unique opportunity for vibrant dialogues but also fostered valuable connections among participants. Under the theme *Collaborating for clean air: confronting climate change and air quality together*, the conference emphasised the indispensable role of collective efforts in addressing urgent environmental challenges around air quality. The proceedings commenced with a joint workshop by DFFE and NACA, laying the foundation for insightful technical sessions, expert presentations, and the sharing of the latest developments in the field of air quality management.

The first set of technical sessions was called *Phathu's Armchair*, where dynamic discussions were fostered with government officials tackling difficult yet pertinent questions across themes like *Stakeholder participation and cooperative governance, The state of ambient air quality in low-income urban settlements of South Africa, and Regulating the regulator*. A second set of sessions, hosted by Farina Lindeque, shed light on the need for assessing the value contribution of our research and work, and explored our efforts to optimise our research collaborations to generate value for all stakeholders affiliated with NACA. These sessions shone a spotlight on the complex and interconnected nature of air quality management and climate change mitigation, and emphasised the necessity for collective action and coordination among stakeholders to address our shared concerns. Current approaches often fall short due to a lack of effective mechanisms for knowledge sharing, learning, and collaboration. The significance of social learning, defined as the process of acquiring new knowledge, skills, and values through social interactions, was underscored as a means to enhance stakeholder capacity for more adaptive and integrated responses to air quality and climate challenges.

Three distinguished keynote speakers, namely Dr Millicent Kekana, Prof Hein Neomagus, and Mr Nicolas Zanetta, illuminated various facets of environmental research and challenges. Topics ranged from collaborative efforts in research to alternative carbon-based fuels and the local ecological implications of metal revolutions and energy transition.

NACA's unwavering commitment to fostering collaboration across academia, industry, government, and society was evident throughout the conference. A diverse array of stakeholders converged, exchanging ideas and cultivating a space for discussing innovative solutions. The carefully curated program acknowledged that addressing clean air challenges requires insights from all sectors of society, catering to the unique interests and concerns of each participant. Throughout the three days, the conference featured the presentation of 14 scientific papers, six engaging 3-minute talks, and 12 informative posters, showcasing the depth and diversity of contributions from the NACA community.

The NACA Golden Award was awarded to Prof Stuart J. Piketh and Grant Ravenscroft for their lifetime contribution to the

science and community. Awards were also presented for the following achievements (presenters are indicted with *):

Best 3-minute talk (3MT)

Characterizing the aerospora and ambient particulate matter over Potchefstroom during the period (13 March- 26 March 2023) (Keneilwe Podile*, Stuart J.Piketh, Frank H. Neumann and Dorra Gharbi)

Best poster

Diurnal variations of PM_{10} concentrations next to paved roads in South African low-income residential areas: a case in Northam (Ntokozo Rakitla*, Ncobile C. Nkosi and Nisa Ayob)

Best student paper (talk)

Estimating yield loss due to surface ozone using AOT40 for maize and soybean across South Africa's Highveld (Authors: Xander Oosthuysen*, Mogesh Naidoo and Rebecca M. Garland)

Best scientific paper winner

Monitoring of biological pollutants during the Comrades Marathon 2023 in South Africa (Authors: Dorra Gharbi, Nikiwe Ndlovu, Frank Harald Neumann, Henno Havenga, Keneilwe Podile*, Jemma Finch, Trevor Hill, Stuart Piketh, Roelof Petrus Burger)

We extend our sincere gratitude to the sponsors whose generous support played a pivotal role in making the 2023 Annual NACA Conference a resounding success. We would like to thank our sponsors, Air Resource Management, Airshed Planning Professionals, Ansyco, Argos Scientific, C&M Consulting Engineers, EnviroServ Waste Management, Eskom, Lakes Environmental Software, Mintek, Mine Dust Network, NMISA, North-West University, SACNASP, Shepstone & Wylie Attorneys, Skyside, SRK Consulting and Testo for their commitment to advancing environmental research and fostering collaboration. Whether participant roles ranged from researcher to policymaker, business leader, or dedicated member of society, the 2023 Annual NACA Conference offered an enriching platform for learning, contributing, and fostering positive change toward a cleaner and healthier environment which remains at the forefront of NACA's ongoing mission.

NWU®

Unit for
Environmental Sciences
and Management

Masters Programme in Environmental Management (M.Env.Man.)
with specialization in Air Quality and Climate Change

BACKGROUND

Climate change and air quality in environmental management faces significant current and future challenges, especially in South Africa (being one of the developing countries in the world). It is recognized that dealing with climate change and air pollution challenges will require a holistic approach from stakeholders from various disciplines.
The learning outcomes of this programme will allow graduates to successfully quip themselves with environmental management and air quality management skills especially those in the sector responsible for management, governance and strategic decision making.

MISSION STATEMENT

The programme aims to build capacity and skills within the air quality and climate change sectors. This is achieved through multi-disciplinary and international training collaboration.
Particular emphasis is placed on the development of analytical skills and critical thinking through high quality research outputs.
This will enable students to compete with confidence as environmental practitioners in the national and international labour market.

COURSE STRUCTURE

This course consists of 3 modules:

- **OMBO886:** Atmospheric Emissions and Impacts (40 Credits)
- **OMBO887:** Air Quality and Climate Change Law and Governance (40 Credits)
- **OMBO873:** Dissertation (100 Credits)

PROGRAMME ARRANGEMENTS

- Two year part-time Programme.
- Three five-day contact sessions per year in Potchefstroom.

ENTRY REQUIRMENTS

- Appropriate honours degree or equivalent.
- Ten students admitted annually based on qualifications and work experience.

MORE INFORMATION

- Module outcomes and programme videos: https://natural-sciences.nwu.ac.za/ccaqi
- Applications: http://studies.nwu.ac.za/postgraduatestudies/higher-degree-admissions
- Course fees: PC-studyfees@nwu.ac.za

CONTACT INFORMATION

- Senior Administrative Officer: Ms Coréne van der Merwe (13279815@nwu.ac.za / 018 285 2443)
- Programme Leaders: Prof Roelof Burger (Roelof.Burger@nwu.ac.za / 018 299 4269)
  Mr Phathu Mukwevho (Phathu.Mukwevho@nwu.ac.za)

In Memoriam
Remembering Gustav Marthinus Snyman
12 January 1930 –10 July 2023

Gerhard Held[1] and Kobus Pienaar[2]

[1]Formerly at Instituto de Pesquisas Meteorológicas, São Paulo State University, Bauru, Brazil (retired since March 2014) and Eskom Technology Services International, Cleveland, South Africa (until March 2000), gerhard@gheld.net.br
[2]Formerly at School of Physical and Chemical Sciences, North-West University, Potchefstroom (retired since October 2017)

https://doi.org/10.17159/caj/2023/33/2.17159

The National Association for Clean Air (NACA) mourns the passing of Gustav Snyman, an instrumental figure in atmospheric sciences in South Africa.

Born in Springs, Gustav started schooling there, but he attended numerous schools due to his father being a magistrate. He matriculated from Grey College in Bloemfontein in 1947. After considering various career possibilities during the next two years, he eventually took on a job in ISCOR's laboratory, while continuing to study analytical chemistry, receiving his Diploma in 1965.

One year later, he accepted a position at the National Building Research Institute of the CSIR. In the early 1980s, he moved to the Atmospheric Sciences Division (under Dr Ernest Carte), which had been merged with the then Air Pollution Research Group (initially under Dr Eric Halliday), under the umbrella of the National Physical Research Laboratory, which in 1989 was restructured into Earth, Marine and Atmospheric Science and Technology (Ematek) of the CSIR. Gustav was the responsible chemist for ion chromatographic analysis in the Air Quality Group, which maintained a network of some 30 particulate samplers from 1982, equipped with parallel sets of filters for PM_{10} and $PM_{2.5}$, across the Eastern Highveld region. He also played a key role in the acid rain studies, later consolidated into the Kiepersol Joint Venture (KJV), a collaborative project between Ematek/CSIR, Eskom/TRI and relevant industries, under the umbrella of the Department of Environmental Affairs and Tourism.

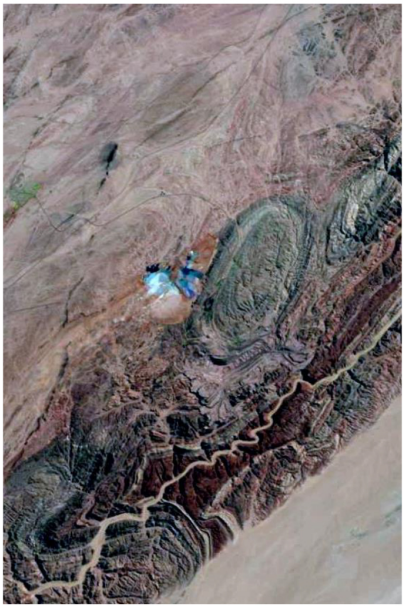
After his formal retirement in 1990, he continued as a consultant to the Air Quality Group of Ematek until 1991. In 1992 he accepted a consulting contract with the Environmental Sciences Group of Eskom's Technology Research and Investigations, where he became a dedicated member of the joint project with the Atmospheric Chemistry Research Group of the then Potchefstroom University for Christian Higher Education. Together with the late Clive Turner, Kobus Pienaar and John Osborne, he significantly contributed to the development of the sampling and analysis techniques of biogeochemically important solid and soluble anions and cations in airborne particulate matter. He also participated actively in several field campaigns in this joint venture between 1996 and 2000 and contributed significantly to the analytical chemistry successes of these studies.

Gustav was a loyal member and active participant in NACA activities and one of the driving forces in the atmospheric science community at the time. During his active years at the CSIR, he also participated in and contributed to conferences of the South African Society for Atmospheric Sciences (SASAS). He was also an active member of the Air Pollution Liaison Committee (APOLCOM) since its establishment in 1987, which united engineers and managers of heavy industries in the Eastern Transvaal Highveld (now Mpumalanga) to improve the effectiveness and efficiency of the air pollution abatement equipment at source.

During a full life of professional activities, he authored or co-authored many papers in scientific journals, conference contributions, as well as technical reports, and also significantly contributed to two chapters of "Air Pollution and its Impacts on the South African Highveld", published by the Environmental Scientific Association in 1996.

Gustav passed away on the morning of 10 July 2023 at the age of 93. We shall remember his jovial way when we were on extended field campaigns. Our sincere condolences are extended to his wife Elize, their four children, and numerous grand- and great-grandchildren.

MINE DUST NETWORK



The Mine Dust and Health Network was established in July 2019, through a research grant from the Global Challenges in Research Fund (GCRF), as part of the United Kingdom Research and Innovation

The network serves as a collaborative think tank and brings together stakeholders from different disciplines and interest groups. The interdisciplinary and collaborative approach which embraces a wide range of perspectives, seeks to shape research and inform policy in the complex and poorly understood space of mine dust and its impacts on the health of communities.

Our desired outcomes

Integrated solutions to mine dust tailored to resource limited environments and **co-created** by academics and all other relevant stakeholders

Government **policy** and **regulations** for managing mine dust health impacts are **meaningful** and **effective**

Capacity sharing and **knowledge exchange** enabling wide dissemination and application of research **results**, and increased potential for **future funding**

Thought leaders of the future to **influence** the social, regulatory and professional environments of their **resident countries**

Communities are better **informed** and **empowered** to take **ownership** of their own **well-being** and **future**

- The network provides a platform for conversations to share information and to collectively develop solutions for a healthy environment and healthy communities.
- Membership and participation in the network are entirely voluntary and is underpinned by a shared commitment to the well-being of affected communities.

Improved quality of life, health and environment of mining-impacted communities

www.linkedIn.com/company/mine-dust-network

www.minedust.org

GCRF Global Challenges Research Fund

minerals to metals

UNIVERSITY OF CAPE TOWN
IYUNIVESITHI YASEKAPA • UNIVERSITEIT VAN KAAPSTAD

Research article

Variability of ambient particulate matter loading at Henties Bay, Namibia

Monray D Belelie[1*], Henno Havenga[1], Danitza Klopper[2], Rebecca M Garland[3,4], Brigitte Language[1,5], Paola Formenti[5], Andreas Namwoonde[6], Roelof P Burger[1], and Stuart J Piketh[1]

[1]Unit for Environmental Sciences and Management, North-West University, Potchefstroom, South Africa
[2]Department of Geography and Environmental Sciences, School of Agricultural and Environmental Sciences, University of Limpopo, Mankweng
[3]Smart Places, Council for Scientific and Industrial Research, Pretoria, South Africa
[4]Department of Geography, Geo-informatics and Meteorology, University of Pretoria, Pretoria, South Africa
[5]Laboratoire Interuniversitaire des Systèmes Atmosphériques (LISA), UMR CNRS, Université Paris-Est Créteil, Université de Paris, Institut Pierre Simon Laplace, Créteil, France
[6]Sam Nujoma Marine and Coastal Resources Research Centre, Sam Nujoma Campus, University of Namibia, Henties Bay, Namibia
Corresponding author: monraybelelie@gmail.com

Received: 14 August 2023 - **Reviewed:** 3 October 2023 - **Accepted:** 5 October 2023
https://doi.org/10.17159/caj/2023/33/2.16670

Abstract

The Namibian coast is one of the areas of international interest for aerosol studies. This is due to the region's importance for the global radiation budget because of the presence of a semi-permanent stratocumulus cloud along the coast. Aerosol particles may scatter/absorb radiation and directly influence how long clouds last by modifying their properties. This is all dependent on the particles' chemical and physical properties influenced by the sources they were emitted from. In this study, we identified and investigated episodes of high (HAE) and low (LAE) PM concentrations and the meteorology that may favour their occurrence. Here, we investigated $PM_{2.5}$ (particles with an aerodynamic diameter of 2.5 µm or less) and PM_{10} (particles with an aerodynamic diameter of 10 µm or less) at Henties Bay, Namibia. Daily aerosol measurements were taken with E-samplers between 15 and 29 July 2019. The Hybrid Single-Particle Lagrangian Integrated Trajectory (HYSPLIT) model was used to investigate the long-range atmospheric transport of air masses that reached Henties Bay. The study found that during HAEs, the average $PM_{2.5}$ concentration was 28.40 ± 18.10 µg/m^3 and the average PM_{10} concentration was 68.20 ± 44.3 µg/m^3. In contrast, during LAEs, the average $PM_{2.5}$ concentration was 13.3 ± 9.52 µg/m^3 and the average PM_{10} concentration was 30.00 ± 23.00 µg/m^3. In both fractions, there was an observed dominant contribution from marine sources.

Keywords

PM, stratocumulus cloud, HYSPLIT, HAEs, LAEs

Introduction

Ambient particulate matter (PM) is used as an air quality indicator (Chen & Hoek, 2020) and has impacts on phenomena such as haze formation and climate change (Liu et al., 2018). PM is commonly categorised based on its size as $PM_{2.5}$ (aerosol with an aerodynamic diameter of 2.5 µm or less) and PM_{10} (aerosol with an aerodynamic diameter of 10 µm or less) (Kastury et al., 2017; Kastury et al., 2018). The distribution and physicochemical properties of PM particles in the atmosphere vary horizontally and vertically. This is due to their uneven source distribution and short lifetime, which is typically less than 10 days for particles <1 µm and even shorter for particles >1 µm (Klopper et al., 2020). A variety of primary sources, both natural and anthropogenic, may emit PM. It can also be formed as secondary particles through photochemical processes involving the primary precursors (Dai et al., 2019; Zalakeviciute et al., 2020; Khan et al., 2021). These sources include biomass burning, road traffic, sea salt, and dust, amongst others (Squizzato et al., 2017; Zalakeviciute et al., 2020; Psistaki et al., 2023). Given the various possible sources, the PM concentration and composition at a site depend on several factors such as the regional background and its meteorology (Yang et al., 2020; Zalakeviciute et al., 2020).

The Namibian coast is one of the areas of interest for PM studies and their impact on the climate. A recent study found that Namibian coastal areas have predominant PM inputs from sea

salt (75%) and mineral dust (16%) (Klopper et al., 2020). The production of primary marine aerosols is largely influenced by local wind stress, with pure sea salt being the major constituent. This production yields marine particles with a diameter of <20 µm ranging from 2000 to 10000 T/yr (de Leeuw et al., 2011; Fuzzi et al., 2015). Ultra-fine marine particles (<1 µm) may be transported over long atmospheric distances while the larger particles are deposited close to where they were produced (Fuzzi et al., 2015). The local meteorology in the arid region is influenced by the adjacent cold Benguela ocean current and semi-permanent stratocumulus cloud layer. The Benguela region is known for its high marine biogenic productivity which emits sulphur-containing compounds such as dimethyl sulphide and hydrogen sulphide into the surrounding atmosphere (Klopper et al., 2020). Once in the atmosphere, these compounds may be oxidised and produce secondary particles which then contribute to cloud droplet formation of the stratocumulus cloud (Andreae et al., 1995).

Anthropologically induced land surfaces, desert regions, and ephemeral dry lakes or riverbeds are the primary sources of dust globally (Mahowald et al., 2009). Specifically, most of its budget originates from fluvial dust sources (Poulton & Raiswell, 2002). Dry ephemeral lakes found in arid and semi-arid localities, such as Makgadikgadi and Etosha pans in Southern Africa, are the major global sources of aeolian dust to the ocean (Prospero et al., 2002). These sources have been hypothesised to be significant for fertilising the adjacent ocean (Piketh et al., 2000). Other important Southern African dust sources are the Kuiseb, Huab, Tsauchab, and Omaruru ephemeral riverbeds in Namibia. They may also play a noteworthy fertilisation role in phytoplankton in the adjacent Benguela (Jacobson et al., 2000; Jacobson & Jacobson, 2013; Dansie et al., 2017). Analyses using remote sensing have also revealed that these riverbeds are significant sources of dust plumes transported to the southern Atlantic (Eckardt & Kuring, 2005; Vickery et al. 2013).

The Henties Bay Aerosol Observatory (HBAO) continuously monitors aerosol measurements, including their chemical composition and concentrations (Klopper et al., 2020). This study expands on previous research at the HBAO by identifying coastal episodes of high and low PM concentrations, and the weather conditions that favour the occurrence of these episodes.

Methods

Sampling

Concentrations of particulate matter (PM) were sampled at the University of Namibia's Sam Nujoma campus (S22°5'43.944"; E14°15'9552") between 15 and 29 July 2019 (Figure 1) (Figure 2). The campus is located next to the Omaruru riverbed, southwest of the Etosha Pan and Huab riverbed and northwest of the Kuiseb- and Tsauchab riverbeds. The sampling was done by concurrently operating two E-samplers mounted on tripods (Met One Instrument, 2011). The E-samplers have a particle size range of 0.1–100 µm and a measurement range between 0 and 65 mg/m^3 with a precision of 2%. The measurements were converted from mg/m^3 to µg/m^3 by multiplying each value by 1000. One sampler continuously measured $PM_{2.5}$ and the other measured PM_{10} (Figure 2). Measurements were taken in 15-minute intervals at a flow rate of 2 L/min.

Unfortunately, the meteorology during the sampling period was not measured at the site. However, the Wlotzkasbaken weather station from the "Southern African Science Service Centre for

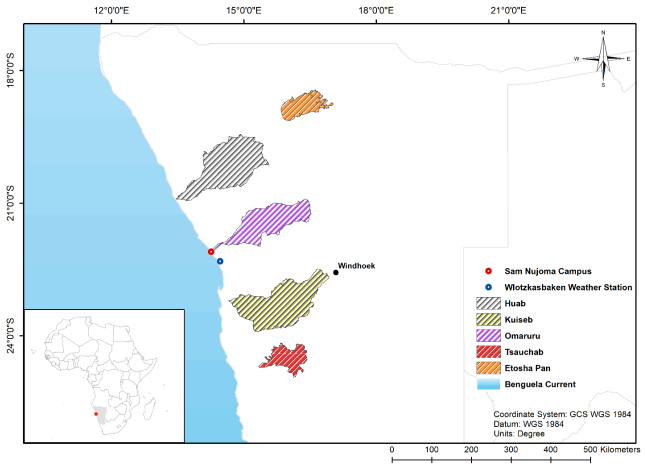


Figure 1: *Location of the Sam Nujoma campus, the Wlotzkasbaken weather station, and important Namibian dust sources.*

Figure 2: *E-samplers (mounted on tripod stands) that were used to concurrently measure $PM_{2.5}$ (left) and PM_{10} (right).*

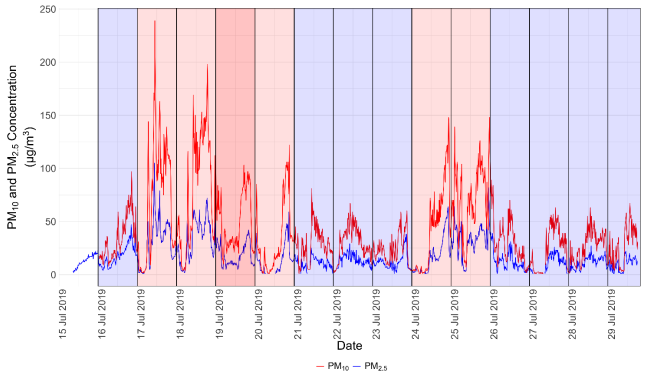


Figure 3: *Time series showing the observed $PM_{2.5}$ (blue) and PM_{10} (red) concentrations (µg/m^3) during the sampling period. The red boxes indicate the days on which HAEs were identified while the blue boxes indicate the LAEs.*

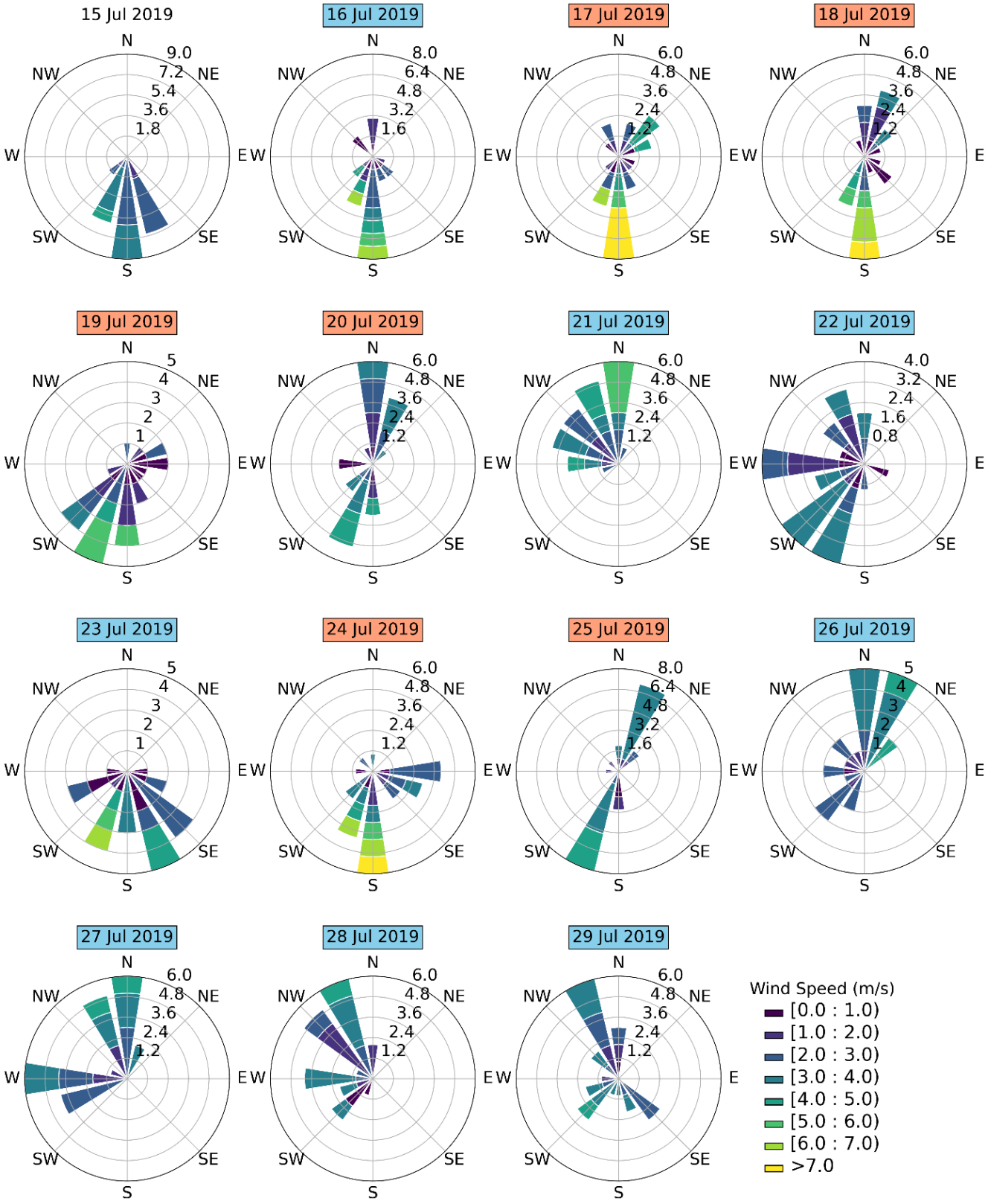


Figure 4: *The prevailing daily wind speed and direction at the Sam Nujoma campus during the sampling period. The dates highlighted in red represent the days on which there were HAE events while LAEs are highlighted in blue.*

Climate Change and Adaptive Land Management" (SASSCAL) is located about 32 km southeast of the monitoring site (Kaspar et al., 2015). An hourly wind speed and wind direction dataset for the weather station was obtained from the SASSCAL website, given its proximity to Henties Bay. This dataset was used to create both daily and hourly windroses for the sampling period.

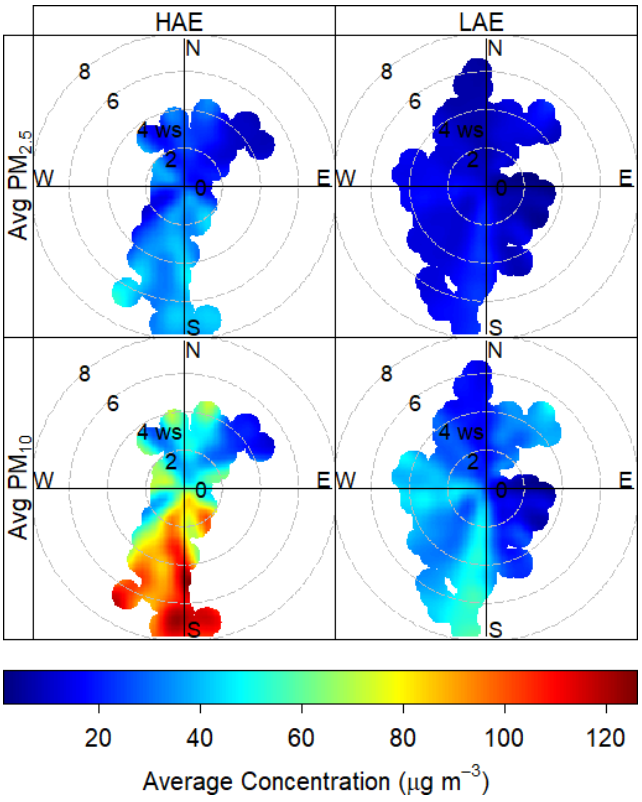


Figure 5: *Polar plots illustrating potential PM2$_{.5}$ and PM$_{10}$ emission sources affecting observed concentrations at Henties Bay.*

Dust event identification

Presently, there is no clear and agreed-upon method for defining dust events based on PM data (Wiggs et al., 2022). Therefore, in this study, we identified a dust event where the average hourly PM$_{10}$ measurements remained at or above 20.055 µg/m³ for 80% of the day. This concentration represents a "severe dust storm" according to the classification by Leys et al. (2011) and is a very conservative identifier of a dust event. We chose this as the most appropriate classification for this study as the hourly PM$_{10}$ concentrations never reached the thresholds of the other classes. Days which met these criteria were classified as high aerosol episodes (HAEs) while days which did not were classified as low aerosol episodes (LAEs).

HYSPLIT back trajectory analysis

For this study (Stein et al., 2015), the National Oceanic and Atmospheric Administration's (NOAA) Hybrid Single-Particle Lagrangian Integrated Trajectory (HYSPLIT) model ran 72-hour back-trajectories, which were initiated at a height of 250 m above ground level. The purpose of starting at this height was to model the transport of air masses into the marine boundary layer, which has a minimum height of approximately 500 m over the Namibian region (Klopper et al., 2020). The chosen height of 250 m corresponds to the first and second vertical levels in the model, which are situated at 1000 hPa (around 110 m above mean sea level (masl)) and 975 hPa (roughly 300 masl), respectively. The Global Data Assimilation System (GDAS) reanalysis dataset, which is provided by the National Centre for Environmental Prediction (NCEP) and has a resolution of

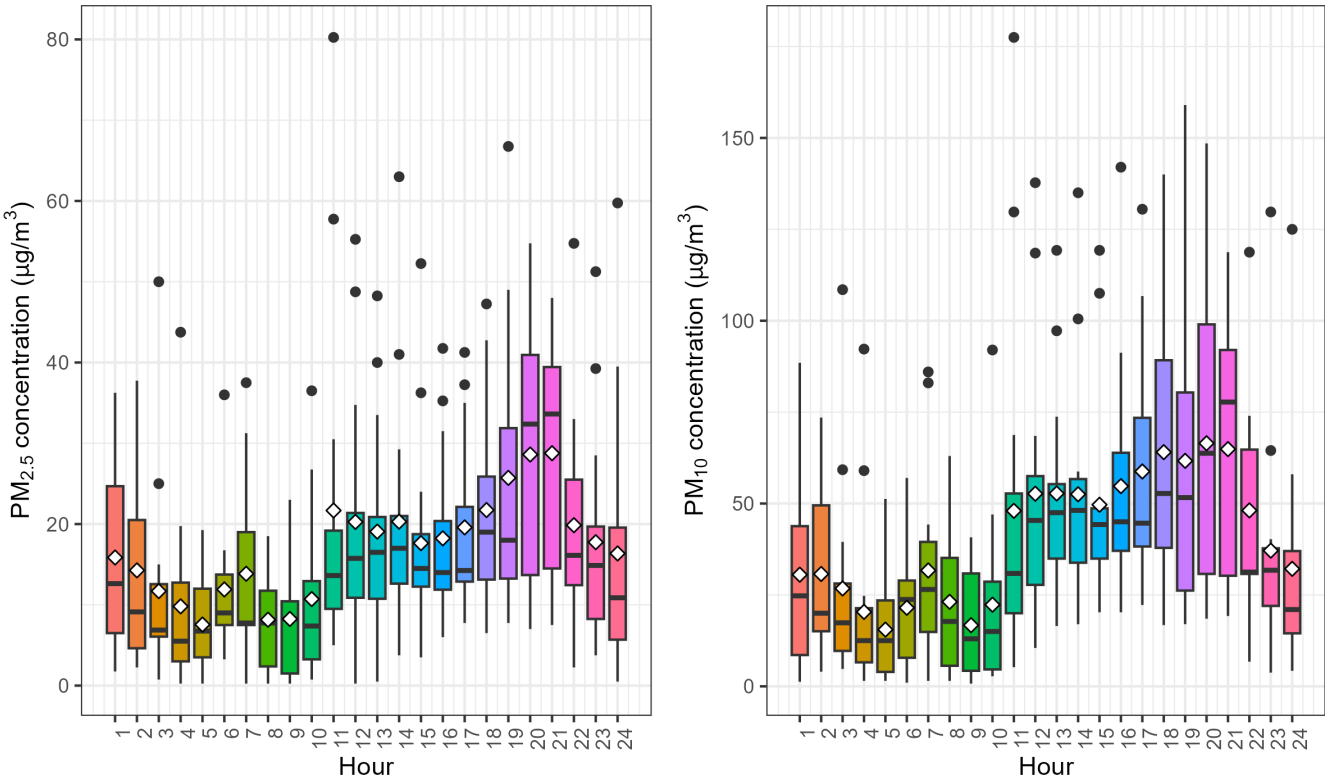


Figure 6: *Diurnal variation in PM$_{2.5}$ (left) and PM$_{10}$ (right) concentration (µg/m³) observed during the sampling period. The white dots indicate the mean while the black dots indicate outliers. Please note that the graphs are on different y-axis scales.*

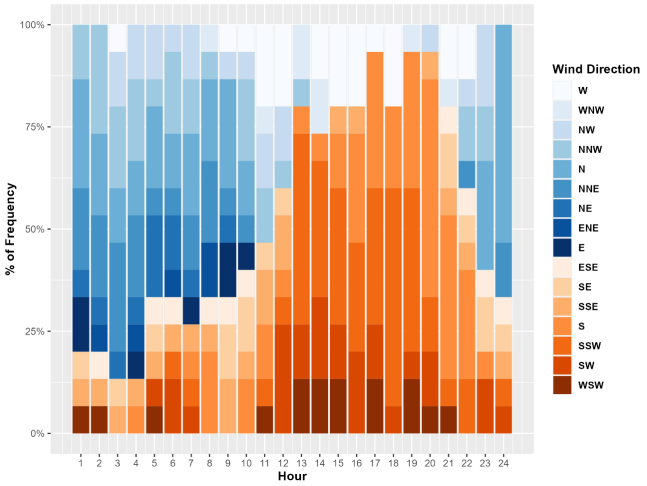


Figure 7: *Stacked bar graph illustrating the predominant wind directions per hour.*

1°x1°, was used to model these trajectories. The modelling was conducted using the Rstudio interface for Windows and relied on the rich_iannone/splitR and Openair packages from the open-source libraries (splitR is available from https://github.com/rich-iannone/SplitR; Carslaw & Ropkins, 2017).

Table 1: *Summary of the synoptic meteorology during the study period. The days on which there were HAEs are highlighted in yellow.*

Date	Episode	Synoptic Condition	Transport Origin
15-16 July	LAE	West coast trough, easterly transport	No long-range transport from further east
17-19 July	HAE	West coast trough, strong pressure gradient	Transport of air from central South Africa, Zimbabwe, Zambia
20-21 July	LAE	West coast trough, strong pressure gradient	Transport of air from central South Africa, Zimbabwe, Zambia
22 July	LAE	High pressure directly over the study site	Very little air transport from the subcontinent
23 July	LAE	Onshore flow along South-Atlantic High	Dominant circulation is driven onshore
24-25 July	HAE	West coast trough, strong pressure gradient	Transport of air from central South Africa, Zimbabwe, Zambia
26 July	LAE	West coast trough, strong pressure gradient	Transport of air from central South Africa, Zimbabwe, Zambia
27-28 July	LAE	Stable weather, no pressure gradient	No pressure gradient near Henties Bay, stable weather

Results and discussion

Daily PM concentrations

Figure 3 shows a time series of the 15-minute $PM_{2.5}$ and PM_{10} concentrations between 15 and 29 July 2019. The figure also highlights days when HAEs were observed. During the sampling period, the $PM_{2.5}$ concentrations ranged from 1.00–105.00 µg/m³ with a mean value of 17.70 ± 14.38 µg/m³. On the other hand, PM_{10} concentrations ranged from 1.00–239.00 µg/m³ with a mean value of 41.12 ± 35.31 µg/m³. According to the 'dust event' criteria, 17, 18, 19, 20, 24, and 25 July were classified as HAEs and the rest as LAEs. During the HAEs, the mean $PM_{2.5}$ concentration was 24.70 ± 18.20 µg/m³ and the mean PM_{10} concentration was 57.70 ± 44.20 µg/m³. In contrast, during LAEs, the mean $PM_{2.5}$ and PM_{10} concentrations were 12.7 ± 7.70 µg/m³ and 28.20 ± 17.80 µg/m³, respectively. The daily windrose plots in Figure 4 provide additional insight into the meteorological factors that may have affected the observed differences in $PM_{2.5}$ and PM_{10} concentrations during HAEs and LAEs. On days with HAEs, the wind rose plots show predominant southerly and southwesterly winds at speeds exceeding, on most days, 6 m/s. These speeds were the highest during the sampling period and agree with Klopper et al. (2020). Given the location of the study site, of course, the dominant PM source located upwind in those directions is the adjacent ocean. On 20 and 25 July, however, there were winds exceeding 4 m/s from the north and northeast suggesting a second potential emission source.

This is expected as the study by Klopper et al. (2020) showed that aerosol concentrations at the site are influenced mainly by sea salt (74.7%) followed by mineral dust (15.7%), ammonium (6.1%), fugitive dust sources (2.6%), and emissions from industry (0.9%). However, on 16 and 22 July, the predominant wind directions were also southwesterly and westerly, during an LAE. This suggests that other factors may also have influenced the observed PM concentrations and subsequent HAEs and LAEs.

For example, PM_{10} is mostly made up of particles that are directly emitted into the atmosphere (Wong et al., 2022). However, some of its constituents may also be formed through secondary processes. For example, when nitric acid, produced by the oxidation of nitrogen oxides reacts with pre-existing alkaline aerosols like sea salt and dust particles (Bian et al., 2014). According to Klopper et al. (2022), another factor might be present at the synoptic scale. In their study synoptic meteorology was shown to influence the land and sea breezes at the site (Klopper et al., 2020). We created polar plots to examine and display the potential sources of $PM_{2.5}$ and PM_{10} emissions during HAEs and LAEs (Figure 5). These plots illustrate the variations in PM concentrations based on the wind speeds and directions using polar coordinates, indicating emissions that may affect the receptor site. In addition to the expected dominant contribution of marine sources during HAEs, there appear to be potential continental sources located to the north-northwest and north-northeast of the site. As shown in Figure 1, the Huab riverbed is a potential major source in that direction.

There is also a possible source to the south-southeast and southeast of the site, which may be attributed to contributions from the Kuiseb riverbed. There's also a contribution around and to the east of the site which is most likely the adjacent Omaruru riverbed.

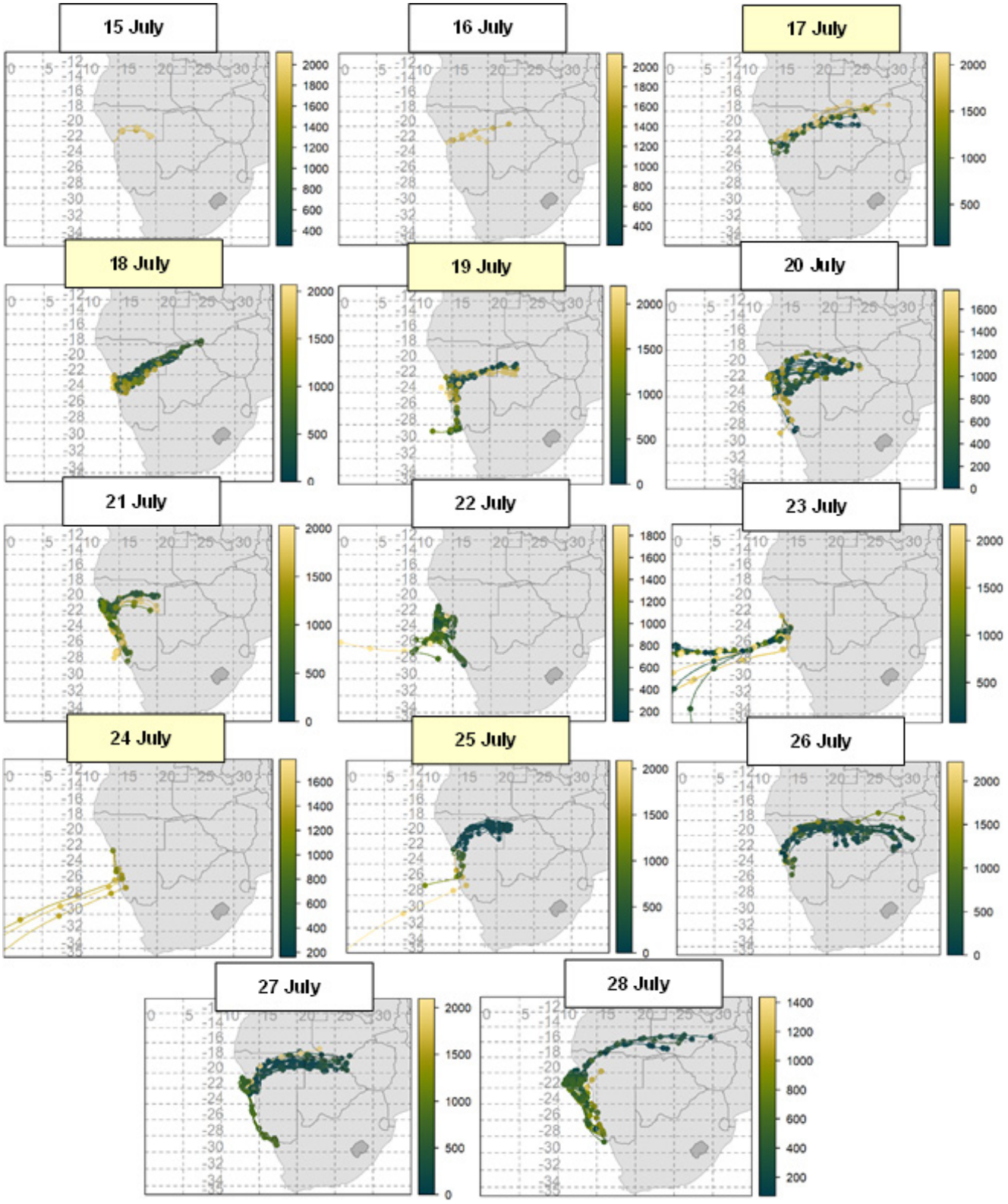


Figure 8: *Back-trajectories of 72-hours during 15-28 July 2019 (the different colours indicate the different computed trajectories). The scale to the right indicates the height above ground level (m). The dates on which there were HAEs are highlighted in yellow.*

Hourly PM concentrations

The boxplots in Figure 6 show the hourly variation in $PM_{2.5}$ and PM_{10} concentrations over the 2-week sampling period. The plots for both fractions show two clear periods in the diurnal concentrations. This indicates that the peaks observed in the morning and the beginning of the evening between the two studies are ascribed to different sources. The mean PM concentrations are generally lower during the early morning

hours (01:00–09:00) and then increase throughout the day, reaching their highest values during the late afternoon and evening hours (16:00–21:00). The maxima and minima also show a large range which indicates that there is a lot of variability in the data. Figure 7 shows the predominant hourly wind directions with a clearly defined and very strong day-night signal. During the day, the predominant wind directions were from the south and southwest while at night, northerly and northeasterly winds were predominant. This agrees with the observation by Klopper et al. (2020) in July 2016 at the site. They found that these day and night wind patterns are associated with sea breezes and land breezes, respectively. The high PM concentration from 09:00 to 21:00 may, therefore, be explained by the dominant sea breezes from the south and southwest during this time. This, in other words, signifies the importance of marine sources to HAE in the region.

Back-trajectories and synoptic meteorology

Figure 8 illustrates the 72-hour back-trajectories computed for the sampling period from 15 to 29 July. During most of the study period, air masses were predominantly transported from the interior. During HAEs, air masses were predominantly transported from the northeastern interior with some from the adjacent ocean. During LAEs, the transport was more varied – on some days air masses were transported from the ocean and on other days from the continent. The lowest PM concentrations were observed between 23-24 July (LAE) and during these days, the dominant transport pathway was from the adjacent ocean.

The synoptic conditions during the sampling period are summarised in Table 1. Generally, HAEs occurred under west coast trough conditions, requiring a strong pressure gradient, and enhanced easterly winds from the central regions. The sources of the aerosols might be as far as southern Zambia, the Caprivi, and dry western Namibian regions, as indicated by the back-trajectory. These conditions were also present on July 25 and 26, with transport from as far as central Zimbabwe. Transport during July may coincide with large veld fires over the region, which can explain why circulation with an inland origin is associated with high aerosol concentrations.

Conclusions

This study presents the results of a detailed analysis of daily $PM_{2.5}$ and PM_{10} concentrations between 15 and 29 July 2019 at Henties Bay, Namibia. The results show that during HAEs, the mean $PM_{2.5}$ concentration was 28.40 ± 18.10 µg/m^3 and the mean PM_{10} concentration was 68.20 ± 44.3 µg/m^3. During LAEs, the mean $PM_{2.5}$ and PM_{10} concentrations were 13.3 ± 9.52 µg/m^3 and 30.00 ± 23.00 µg/m^3, respectively. During HAEs, the predominant southerly and southwesterly winds are from the adjacent ocean. The results also show three other sources of PM emissions to the north and southeast of the site which are most likely the Namibian ephemeral river valleys. Hourly, the highest PM concentrations were observed during the late afternoon and evening hours (16:00–21:00). These concentrations may be explained by the dominant sea breezes from the south and southwest during this time.

During most of the study period, air masses were predominantly transported from the interior. During high aerosol episodes (HAEs), air masses were predominantly transported from the northeastern interior with some from the adjacent ocean while during low aerosol episodes (LAEs), the transport was more varied with air masses being transported from both the ocean and the continent. HAEs generally occurred under west coast trough conditions, requiring a strong pressure gradient and enhanced easterly winds from the central regions. The sources of aerosols might be as far as southern Zambia, the Caprivi, and dry western Namibian regions, as indicated by the back-trajectories. The occurrence of winter veld fires may explain why circulation with an inland origin is associated with HAEs.

Author contributions

MDB, RPB, and SJP: Study conceptualisation.
MDB: Data analysis and drafting of the paper.
DK, BL, and SJP: Methodology.
This manuscript was reviewed by each of the authors listed above.

Acknowledgements

This research was funded by the National Research Foundation of South Africa (Grant: 120764, 106431, 129320, and 114691).

Conflicts of interest

The authors declare that they have no conflicts of interest to disclose, except for RMG, who is an editor-in-chief of the Clean Air Journal. This relationship has been disclosed to the editorial office and all necessary steps have been taken to ensure that the review process was fair and unbiased.

References

Andreae, M.O., Elbert, W., & de Mora, S.J. 1995, 'Biogenic sulfur emissions and aerosols over the tropical South Atlantic: Atmospheric dimethylsulfide, aerosols and cloud condensation nuclei', *Journal of Geophysical Research* 100:11335-11356. https://doi:10.1029/94JD02828

Bian, Q., Huang, Huang, X.H.H. & Yu, J.Z. 2014, 'One-year observations of size distribution characteristics of major aerosol constituents at a coastal receptor site in Hong Kong – Part 1:

An earlier version of this paper was presented at the National Association of Clean Air (NACA) Conference in October 2022 and was published in its Proceedings.

Inorganic ions and oxalate', *Atmospheric Chemistry and Physics*, 14: 9013–9027. https://doi.org/10.5194/acp-14-9013-2014

Carslaw, D. and Ropkins, K. 2017, 'Package "openair": Tools for the Analysis of Air Pollution Data'. https://davidcarslaw.github.io/openair/ Accessed 20 July 2022.

Chen, J. and Hoek, G. 2020, 'Long-term exposure to PM and all-cause and cause-specific mortality: A systematic review and meta-analysis', *Environmental International* 143:105974. https://doi.org/10.1016/j.envint.2020.105974

Dai, Q., Bi, X., Liu, B., Li, L., Ding, J., Song, W. et al. 2019, 'A size-resolved chemical mass balance (SR-CMB) approach for source apportionment of ambient particulate matter by single element analysis', *Atmospheric Environment* 197:45-52. https://doi.org/10.1016/j.atmosenv.2018.10.026

Dansie, A.P., Wiggs., G.F.S & Thomas, D.S.G. 2017, 'Iron and nutrient content of wind-erodible sediment in the ephemeral river valleys of Namibia', *Geomorphology* 290:335–346. https://doi.org/10.1016/j.geomorph.2017.03.016

De Leeuw, G., Andreas, E.L., Anguelova, M.D., Fairall, C.W., Lewis, E.R., O'Dowd, C. et al. 2011, 'Production flux of sea spray aerosol', *Reviews of Geophysics* 49(2):RG2001. https://doi.org/10.1029/2010RG000349

Eckardt, F.D and Kuring, N. 2005, 'SeaWiFS identifies dust sources in the Namib Desert', *International Journal of Remote Sensing* 26:4159–4167. https://doi.org/10.1080/01431160500113112

Fuzzi, S., Baltensperger, U., Carslaw, K., Decesari, S., Denier van der Gon, H., Facchini, M.C. et al. 2015, 'Particulate matter, air quality and climate: lessons learned and future needs', *Atmospheric Chemistry and Physics* 15:8217-8299. https://doi:10.5194/acp-15-8217-2015

Jacobson, P.J and Jacobson, K.M. 2013, 'Hydrologic controls of physical and ecological processes in Namib Desert ephemeral rivers: Implications for conservation and management', *Journal of Arid Environments* 93:80–93. https://doi.org/10.1016/j.jaridenv.2012.01.010

Jacobson, P.J., Jacobson, K.M., Angermeier, P.L & Cherry, D.S. 2000, 'Hydrologic influences on soil properties along ephemeral rivers in the Namib Desert', *Journal of Arid Environments* 45:21–34. https://doi.org/10.1006/jare.1999.0619

Kaspar, F., Helmschrot, J., Mhanda, A., Butale, M., de Clercq, W., Kanyanga, J.K., Neto., F.O.S. et al. 2015, 'The SASSCAL contribution to climate observation 12:171–177. https://doi.org/10.5194/asr-12-171-2015

Kastury, F., Smith, E. & Juhasz, A.L. 2017, 'A critical review of approaches and limitations of inhalation bioavailability and bioaccessibility of metal(loid)s from ambient particulate matter or dust', *Science of The Total Environment* 574:1054-1074. http://dx.doi.org/10.1016/j.scitotenv.2016.09.056

Kastury, F., Smith, E., Karna, R.R., Scheckel, K.G. & Juhasz, A.L. 2018, 'An inhalation-ingestion bioaccessibility assay (IIBA) for the assessment of exposure to metal(loid)s in PM_{10}', *Science of The Total Environment* 631-632:92-104. https://doi.org/10.1016/j.scitotenv.2018.02.337

Klopper, D., Formenti, P., Namwoonde, A., Cazaunau, M., Chevaillier, S., Feron, A., Gaimoz, C. et al. 2020, 'Chemical composition and source apportionment of atmospheric aerosols on the Namibian coast', *Atmospheric Chemistry and Physics* 20:15811–15833. https://doi.org/10.5194/acp-20-15811-2020

Leys, J.F., Heidenreich, S.K., Strong, C.L, McTainsh., G.H & Quigley, S. 2011, 'PM_{10} concentrations and mass transport during "Red Dawn" - Sydney 23 September 2009', *AeoRe*, 3:327–342. https://doi.org/10.1016/j.aeolia.2011.06.003

Liu, Y., Zhang, W., Yang, W., Bai, Z. & Zhao, X. 2018, 'Chemical Compositions of $PM_{2.5}$ Emitted from Diesel Trucks and Construction Equipment', *Aerosol Science and Engineering* 2:51-60. https://doi.org/10.1007/s41810-017-0020-2

Mahowald, N.M., Engelstaedter, S., Luo., C, Sealy., A, Artaxo., P, Benitez-Nelson, C., Bonnet, S. et al. 2009, 'Atmospheric Iron Deposition: Global Distribution, Variability, and Human Perturbations', *Annual Review of Marine Science* 1:245–278. https://doi.org/10.1146/annurev.marine.010908.163727

Met One Instrument. 2011, 'E-Sampler-9800 Manual Rev M'. *E-Sampler Particulate Monitor Operation Manual*.

Piketh, S.J. 2000, 'Transport of Aerosols and Trace Gases over Southern Africa', Johannesburg: University of the Witwatersrand, (Thesis–PhD).

Poulton, S.W. 2002, 'The low-temperature geochemical cycle of iron: From continental fluxes to marine sediment deposition', *American Journal of Science* 302: 774–805. https://doi.org/10.2475/ajs.302.9.774

Prospero, J.M. 2002, 'Environmental characterisation of global sources of atmospheric soil dust identified with the NIMBUS 7 Total Ozone Mapping Spectrometer (TOMS) absorbing aerosol product', *Reviews of Geophysics* 40:1-31. https://doi.org/10.1029/2000rg000095

Psistaki, K., Achilleos, S., Middleton, N. & Paschalidou, A.K. 2023, 'Exploring the impact of particulate matter on mortality in coastal Mediterranean environments', *Science of The Total Environment* 865:161147. https://doi.org/10.1016/j.scitotenv.2022.161147

Squizzato, S., Cazzaro, M., Innocente, E., Visin, F., Hopke, P.K. & Rampazzo, G. 2017, 'Urban air quality in a mid-size city – $PM_{2.5}$

composition, sources and identification of impact areas: From local to long range contributions', *Atmospheric Research* 186:51-62. https://doi.org/10.1016/j.atmosres.2016.11.011

Stein, A.F., Draxler, R.R., Rolph, G.D., Stunder, B.J.B., Cohen, M.D & Ngan, F. 2015, 'NOAA's HYSPLIT Atmospheric Transport and Dispersion Modeling System', *Bulletin of the American Meteorological Society* 96:2059–2077. https://doi.org/10.1175/bams-d-14-00110.1

Vickery, K.J., Eckardt, F.D & Bryant, R.G. 2013, 'A sub-basin scale dust plume source frequency inventory for southern Africa, 2005–2008', *Geophysical Research Letters* 40:5274–5279. https://doi.org/10.1002/grl.50968

Wiggs, G.F.S., Baddock, M.C., Thomas, D.S.G., Washington, R., Nield, J.M., Engelstaedter, S. et al. 2022, 'Quantifying Mechanisms of Aeolian Dust Emission: Field Measurements at Etosha Pan, Namibia', *Journal of Geophysical Research: Earth Surface* 127(8):e2022JF006675. https://doi.org/10.1029/2022jf006675

Wong, Y.K., Liu, K.M., Yeung, C., Leung, K.K.M. & Yu, J.Z. 2022, 'Measurement report: Characterisation and source apportionment of coarse particulate matter in Hong Kong: insights into the constituents of unidentified mass and source origins in a coastal city in southern China', *Atmospheric Chemistry and Physics* 22: 5017–5031. https://doi.org/10.5194/acp-22-5017-2022

Yang, H., Peng, Q., Zhou, J., Song, G. & Gong, X. 2020, 'The unidirectional causality influence of factors on $PM_{2.5}$ in Shenyang city of China', *Scientific Reports* 10:8403. https://doi.org/10.1038/s41598-020-65391-5

Zalakeviciute, R., Rybarczyk, Y., Granda-Albuja, M.G., Suarez, M.V.D., & Alexandrino, K. 2020, 'Chemical characterisation of urban PM_{10} in the Tropical Andes', *Atmospheric Pollution Research* 11:343-356. https://doi.org/10.1016/j.apr.2019.11.007

AIR RESOURCE MANAGEMENT (PTY) LTD

www.arm-air.co.za

SUSTAINABLE AIR QUALITY IMPROVEMENT SOLUTIONS




Our Story

We are recognized experts at efficiently and cost-effectively solving complex air quality regulatory issues for our clients

We help organizations comply with applicable air quality regulatory requirements and optimize environmental performance for long-term sustainability.

Our Mission

We develop, manage, and deliver sustainable air quality solutions that are innovative & objective, to accomplish our clients' objectives.

Our Vision

To improve the world's air quality by providing sustainable air quality solutions to complex problems.

Our Services

Our team of engineers and scientists have a proven track record in delivering expert insight and tailored air quality solutions for industry, government, research, and the academic sectors.

Our suite of air quality services include:

- ✓ Air Quality Regulatory & Policy Services
- ✓ Air Quality Impact & Risk Assessments
- ✓ Air Quality Offsetting
- ✓ Cost Benefit Analysis for Air Quality
- ✓ Dispersion & Photochemical Modelling
- ✓ Ambient Air Quality Measurements
- ✓ NAEIS quantification & reporting
- ✓ Air Quality Training & Development
- ✓ Climate Risk & Assessment

Our Office

22 Magwa Crescent, Gateway West, Ground Floor, Waterfall City, Midrand, Johannesburg,, South Africa

Contact Us

Avishkar Ramandh
Phone: +27 84 690 7969
Or +27 10 009 5336
Email: avi@arm-air.co.za

Sustainable Development Goals

Research article

Decreasing trend in SO_2 concentrations over Durban: 2004 - 2014

Barbara Duigan[1*], **Sivakumar Kansdasami Sangeetha**[2] and **Venkataraman Sivakumar**[1,3]

[1]School of Chemistry and Physics, College of Agriculture Engineering and Science, University of KwaZulu-Natal, Westville Campus, Private Bag X5400, Westville, Durban 4000, South Africa
[2]School of Agricultural, Earth and Environmental Sciences, College of Agriculture, Engineering and Science, University of KwaZulu-Natal, Westville Campus, Private Bag X5400, Westville, Durban 4000, South Africa
[3]National institute for theoretical and computational sciences, University of KwaZulu Natal, Durban 4000, South Africa
* Corresponding author: blduigan02@gmail.com

Received: 24 January 2023 - **Reviewed:** 14 March 2023 - **Accepted:** 3 November 2023
https://doi.org/10.17159/caj/2023/33/2.15451

Abstract

The climatology of ambient SO_2 air pollution was investigated in Durban, South Africa using data collected by seven air quality monitoring stations (Ferndale, Grosvenor, Jacobs, Wentworth, Settlers School, Southern Works and Prospecton) operated by eThekwini municipality (2004–2014, 2018-2019). These sites constitute a mix of urban and industrial locations.

Yearly average trends indicated that no site exceeded the yearly average national guideline (19ppbv) 2004-2014, 2018-2019. Southern Works, Wentworth and Jacobs, recorded highest yearly averages, a consequence of situation within Durban South Industrial Basin (DSIB) while Ferndale recorded lowest yearly averages, a reflection of location in an urban environment. Results of linear fitting to yearly average data (2004-2014) indicated negative trends (all sites). The largest trend was recorded at Jacobs (-0.48ppbv yr-1) and smallest trends observed at Prospecton (-0.12ppbv yr^{-1}), Ferndale (-0.084ppbv yr^{-1}) and Grosvenor (-0.024ppbv yr^{-1}). Using these linear trends, projected SO_2 levels were calculated and compared to actual data where it existed (2019 - Wentworth, Settlers, Prospecton), (2018, 2019 Southern Works). Comparison of actual with projected data indicated that except for Prospecton, projected yearly averages are lower than actual yearly averages for these sites. The largest difference between projected and actual data occurred for Southern Works in 2019 (4.10 ppbv). Monthly averages displayed periodic behaviour with maxima recorded in winter and minima in summer. Highest monthly averages were consistently recorded at Wentworth, Jacobs or Southern Works. Maximum monthly average (2004-2019) was reported at Jacobs, June 2009 (21.71± 2.82ppbv) while minimum monthly average (2004–2019) was reported at Ferndale, December 2012 (0.32±0.04ppbv).

Jacobs recorded maximum seasonal values June (11.54±6.37ppbv) to November (8.32±3.59ppbv), February (7.18±5.33ppbv) and April (9.02±6.38ppbv) while Southern Works recorded maximum seasonal average January (7.37±5.09ppbv), March (9.03±5.81ppbv). During May, Southern Works, Jacobs, Wentworth reported closely matched values (9.04±4.50ppbv, 9.17±6.34ppbv, 9.49± 5.22ppbv). Minimum seasonal levels were recorded at Ferndale (1.06±0.58ppbv – 2.29±0.88ppbv). Ferndale, Grosvenor, Wentworth and Settlers School reported secondary maxima in September/October potentially indicating the influence of biomass burning at these locations.

Seasonal averages illustrated that cooler conditions favoured higher SO_2 levels and warmer conditions, lower concentrations. Comparison of average difference between seasonal maximum and seasonal minimum (average width of seasonal envelope) indicated that Jacobs had the largest seasonal envelope and Ferndale the smallest.

Previous Durban SO_2 studies are considerably older or of shorter duration than the analysis presented here and are limited as they principally focus on yearly average trends. The determination of SO_2 levels over additional averaging periods (monthly average, seasonal variation, seasonal average) coupled with a larger data set and the comparison across sites (industrial vs urban) provides a more detailed exposure profile as experienced by individuals living and working in the eThekwini municipality and thereby expands on previous investigations.

Keywords

Ambient SO_2 pollution, Durban, yearly average, monthly average, seasonal variation, seasonal average.

Introduction

Air pollution is a major problem of the new millennium, and it has become increasingly clear that human activities are playing an important role in the cycling of trace gases in the atmosphere (Carslaw and Carslaw 2001). Poor air quality can affect health and the wider environment, particularly in urban areas where the majority of people live and work. Due to its abundance and substantial health impacts, SO_2 was chosen as the target pollutant in this investigation.

Several factors influence air quality in urban areas. The levels of pollutants released into the atmosphere are directly related to the number of emission sources, distribution of emission sources and volume of pollutants released by each source. These sources may be in the form of stationary sources e.g. emissions from a combustion furnace flue stack or mobile sources such as exhaust emissions from cars / aeroplanes etc. and accompanying transport of pollutants (Masiol et al. 2014).

Air quality is also affected by the rate at which pollutants disperse. Dispersion is dependent on both wind direction and strength. Strong winds result in rapid dispersal of emissions whereas little or no wind results in the accumulation, and in some cases, high concentration of air pollutants. Local factors such as topography and proximity to coast, building height and time of year all affect local wind conditions and can play a role in increasing air pollution levels (Diab et al. 2002; Thambiran and Diab 2010).

As in the case of NO_2, SO_2 has multiple emission sources both natural and anthropogenic in origin (Masiol et al. 2014). Natural emission sources of SO_2 include volcanoes, grassland, and forest fires. Coal and petroleum often contain sulphur compounds and their combustion generates SO_2. Anthropogenic SO_2 emission sources therefore include combustion of fossil fuels / crude oil and coal transformation processes. It is also produced as a by-product of metal smelting (of sulphur containing ores) and other industrial processes (Masiol et al. 2014). Approximately 90% of sulphur present in fossil fuels enters the gas phase in the form of SO_2 during combustion unless it is deliberately removed from fuel gas (Hewitt 2001). As a result of these combined industrial activities, approximately 99% of the SO_2 present in ambient air is of anthropogenic origin (Hewitt 2001). The main sinks of SO_2 are the oxidation by OH radicals and wet deposition, namely the solution in cloud droplets where it is converted to sulphurous (H_2SO_3) and sulphuric acid (H_2SO_4) (Pham et al. 1995).

The main route of SO_2 exposure is inhalation and the target organs comprise the respiratory system. Once inhaled, SO_2 is absorbed through the respiratory tract and distributed to all parts of the body, including the brain and bone marrow. Exposure to SO_2 is linked to numerous adverse effects on the respiratory system including broncho-constriction, wheezing and increased asthmatic symptoms (Matooane and Diab 2003; Mentz et al. 2018). It has also been found that SO_2 is toxic to plants (Lee et al. 2017).

South Africa has been identified as a source of industrial pollution, significant on a global scale (Josipovic et al. 2010). Furthermore, the problem of air pollution in Durban has a long history, particularly in relation to the Durban South Industrial Basin (DSIB). The DSIB is an approximately 4 km wide area on the eastern seaboard of South Africa, extending south from the Durban Central Business District (CBD) for 24 km to Umbogintwini. This area includes the CBD and Port of Durban, which is the busiest port in Africa. Poor historical land-use planning has resulted in juxtaposition of residential and industrial areas in South Durban. This has led to a long-term ongoing conflict between local communities and industry, particularly regarding concerns about possible effects of ambient air pollution on the health of residents (South Durban basin multi-point plan case study report 2007; Matooane and Diab 2003).

Industrial development in the area started in the mid 1950's. At present, some 600 industries are reportedly located in South Durban Industrial Basin. Emission sources include Mondi (paper manufacturing), sugar refinery, sewage treatment works, a cluster of chemicals industries, major petrochemical and chemical storage facilities, textile manufacturing, metal smelting, breweries, factories relating to the paint and motor industries. During the study period (2004-2014, 2018-2019) the Engen refinery and SAPREF were also significant sources of air pollutants. The fact that several of the major facilities have relatively low stack height (50 – 100 m) further facilitates the increase in pollutants close to the ground (South Durban Basin multi-point plan case study report 2007). The DSIB is also the focal point of many of the city's major transport routes and this adds a further large contribution to emissions from vehicular traffic and shipping (South Durban Basin multi-point plan case study report 2007).

Other factors apart from emission sources affect air pollution levels in DSIB. The local climate has a direct influence on the fate of particulates and gases released into the atmosphere. Concentration of pollutants varies on a daily / hourly basis in response to changes in atmospheric stability, resultant mixing depth and atmospheric circulation patterns. High concentration of air pollutants is associated with poor dispersion conditions. Furthermore, South Durban has a basin-like topography that is conducive to the accumulation of pollution (South Durban Basin multi-point plan case study report 2007), (Air quality scoping report: Royal Haskoning 2014).

Winds in Durban basin blow predominantly from the south-south-west to south-west and north-north-east to north-east in approximately equal proportions. Winds from north-north-east / north-east are associated with high atmospheric pressure and fine weather however winds from south-south-west / south-west are associated with the passage of coastal low-pressure systems and cold fronts and hence accompany unfavourable weather. The direction of predominant winds parallel to the coast together with the DSIB topography results in the channelling of pollutants within the basin (South Durban Basin multi-point plan case study report 2007).

The effect of temperature inversion conditions on pollutant levels should also be considered. A temperature inversion occurs when the air temperature increases with height as opposed to the usual decrease in temperature with increasing altitude. Inversions are common overnight during periods of calm weather and are generally strongest in the early morning hours. This inversion phenomenon acts like the 'lid' on a containment vessel that traps pollutants close to the ground and prevents upward air movement. Temperature inversions coupled with low wind speed can result in high levels of pollutants near the surface. A brown haze is a common feature of air quality in Durban during winter and can be attributed to the photochemical action of NO_x, O_3, SO_2, PM and VOCs. Once air flow is restored, circulation of pollutants can occur. Finally, air quality may be negatively impacted through the transport of pollutants from inland areas down to the coast by north-westerly land breezes at night, particularly during winter.

The effect of rainfall on air pollution is also significant. Annual rainfall in Durban is 1009 mm with most of this rain falling in summer. This period of high rainfall is associated with periods of improved air quality as the overall effect of rainfall is to remove dust and pollutant gases from the atmosphere. During summer humidity is high (sometimes approaching 100%) while winter is characterised by low humidity (as low as 20%). This high humidity in summer implies that chemical reactions that require water vapour are accomplished more efficiently, hence airborne pollutants are removed more effectively and rapidly than during the dry winter conditions (South Durban Basin multi-point plan case study report 2007), (Naidoo et al. 2007). Overall, dispersion conditions in summer improve due to less stable air conditions, higher wind speed and the effect of rainfall. As a result of the contributing factors discussed above, DSIB one of the most heavily polluted industrial areas in South Africa.

Figure 1: *Location of seven air pollution monitoring sites employed in this investigation – Ferndale, Grosvenor, Jacobs, Wentworth, Settlers School, Southern Works and Prospecton.*

As SO_2 is a principal by-product of many industrial processes it is often used as an 'indicator pollutant'. The National Ambient Air Quality Standards (NAAQS) for SO_2 are 191ppbv (10-minute average), 134ppbv (hourly average), 48ppbv (daily average) and 19ppbv (yearly average) (Government Gazette RSA 2009). Air quality monitoring in Durban was initiated at Wentworth in 1958 (Diab and Motha 2007) and as a result SO_2 is therefore the pollutant with the longest record of near continuous monitoring. The present investigation aims to characterize the climatology and seasonal variation of SO_2 emissions in order to compare and expand on previous studies at this location (Diab et al. 2002; Guastella and Mjoli 2005; Gounden 2006; Diab and Motha 2007; Guastella 2008; Mdluli 2015; Khumalo 2020).

It should be noted that previous SO_2 trend studies are considerably older and for significantly shorter study periods than the present investigation, as in the case of Diab et al. 2002; Guastella and Mjoli 2005; Gounden 2006; Diab and Motha 2007; Guastella 2008. Furthermore, these studies largely focus on yearly trends. There are very limited studies on monthly averages, seasonal variation or seasonal averages. Analysis of monthly average trends are presented in Guastella and Mjoli (2005), however this work is based on data collected for the period 1996 to 2003. More recent studies (Mdluli 2015 and Khumalo 2020) solely focus on analysis of SO_2 yearly average trends. The motivation for this investigation was therefore to complete a detailed long term trend study of SO_2 climatology in Durban, including analysis of not only yearly averages but additionally monthly averages, seasonal variation and seasonal averages as well as an analysis of data recorded in both industrial and urban environments.

Data collection, processing and instrumentation

eThekwini Municipality commissioned the continuous monitoring network in December 2003 as one of the major elements of its Air Quality Management System. The network is composed of instrumentation owned and operated by the eThekwini municipality. The network consists of monitoring stations situated at a range of sites representing heavy industry, urban, residential and rural locations. The network instruments continuously measure priority pollutants (measurement techniques) CO (infrared spectrometry), NO_X (chemiluminescence), SO_2 (UV fluorescence), O_3 (UV photometry), PM_{10} (TEOM - tapered element oscillating microbalance) as well as other species at selected sites.

In this investigation, seven sites were chosen to examine the long-term trends on SO_2 ambient air pollution in South Africa, these are: Ferndale, Grosvenor, Jacobs, Wentworth, Settlers School, Southern Works and Prospection. Figure 1 indicates the locations of the seven monitoring stations.

The seven selected stations can be grouped in the following way: One northern station – Ferndale, which is located at the Ferndale Primary School, is situated some distance from roads and industry. The closest industries are found approximately 2 km to the SE in River Horse Valley Industrial Park and Briardene Industrial Park. This site is representative of an urban environment, and it is the only monitoring station situated in the north of Durban. The remaining six stations - Jacobs, Grosvenor, Wentworth, Settlers School, Southern Works, Prospection are all located in the heart of the DSIB with the exception of Prospecton. It is the most southerly site of the selected locations and serves as an industrial background station measuring pollutant levels entering the DSIB from the Prospecton/Umbogintwini area.

The quality of data collected at the above monitoring stations is assured in a number of ways as a robust server-based data acquisition sytem called Envista Air Rosources Manager is employed. This system is globally used and has built-in data flags which are in accordance with ISO/IEC 17025:2017. Data validation is accomplished by adhering to the National Norms & Standards for Ambient Air Quality Monitoring developed by the Department of Forestry, Fisheries and Environment (DFFE)

Table 1: *Monitoring station (and GPS coordinates), SO_2 monitoring instrumentation, location characteristics and years of available data.*

Station	GPS Coordinates	SO_2 monitoring instrumentation	Location characteristics	Years of available data
Ferndale	-29.77806 30.97805	Monitor labs ML, 9850B	Urban	2004 - 2013
Grosvenor	-29.92089 31.00436	Thermo Scientific, Model 43iQ series	Industrial	2004 - 2012
Jacobs	-29.92831 30.97937	Thermo Scientific, Model 43iQ series	Industrial	2004 - 2014
Wentworth	-29.93306 30.98774	Teledyne API, Model T100	Industrial	2004 – 2014, 2019
Settlers School	-29.95875 30.97905	Teledyne API, Model T100	Industrial	2004 – 2014, 2019
Southern Works	-29.95984 30.97395	Thermo Scientific Model 43iQ series	Industrial	2004 – 2014, 2018, 2019
Prospecton	-30.00311 30.92960	Environment SA, Model AF22M	Industrial background	2004 – 2014, 2019

(National). Data outliers are flagged such as duplications, and erroneous data such as spikes and dips often due to abrupt shutdowns (loadshedding). Data validation checks generally take place every 15 days.

In terms of quality control, the practice of ISO 17025, SANAS TR0703 and the National Department (DFFE) Draft Norms and Standards for Ambient Air Quality Monitoring are followed. Station checks are done daily. In addition to data collection, the Envista system assists with data management and the operation of the monitoring stations. Each station has a datalogger installed with Envida software. Envidas is a package/component of Envista. All instruments are configured to transmit data from instrument to the datalogger, Envidas captures and saves this data on the logger PC. Envidas is configured to the remote server running the Envista DMS, data is transmitted from station level to server level via network connection and these applications. Data for each station and instrument is remotely checked each day on Envista by the data team and the technicians and technicians respond to stations that display no online status for stations or instruments, and/or any data that appears spurious.

Bi-weekly calibration is done in-house and is a one-point check on the gas instruments, using low concentration (ppb) certified and accredited gas. Multi-point calibration is done quarterly, in-house - three times per year. Here a dilution calibrator, zero air generator and a high concentration (ppm) certified and accredited gas is employed. Calibration at zero and high span is undertaken, and gas is diluted to multiple points in order to achieve a linear relationship. Finally, one external calibration is done by a SANAS accredited lab yearly.

Instrument maintenance is done in accordance with manufacturer specifications and standards including preventative maintenance such as planned services. In addition, to bi-weekly calibrations instrument checks are conducted and instrument check sheets completed. In these checks, monitoring of the instrument response not only to gas but also to electronic signals that have a defined operating range/threshold value e.g., instrument flows, temperature, pressure, voltage of certain components: lamps, PMT, etc. takes place (eThekwini air quality monitoring network – Annual Report 2009).

For each monitoring station, Table 1 gives details of GPS coordinates, SO_2 monitoring instrumentation and total years of available data during the period 2004 – 2019. All instrumentation is based on the UV fluorescence technique for SO_2 monitoring.

All original data was obtained in hourly intervals in excel format and processed with the use of MATLAB. In order to remove any spurious values from that dataset (such data gaps, zero and negative values), a 70% data completeness criterion was applied thereby removing any average data point that was calculated from less than 70% available data in that particular dataset. This rule was applied throughout the analysis presented in this work.

All raw hourly data sets were manipulated in order to calculate SO_2 yearly averages, monthly averages, seasonal variation and seasonal averages. It should be noted that not all sites have complete data sets 2004 -2019. For sites where data was available 2018-2019, this was included in order to optimize the data set and consequent findings. For yearly averages, monthly averages and seasonal averages, graphs are plotted for the entire time period in order to facilitate comparison between sites.

Results and discussion

Annual average trends

As a first step in investigating trends in the Durban air pollution SO_2 dataset, yearly averages were calculated for each site and the results presented below. Figure 2 shows a composite of seven plots, each illustrating the yearly average 2004 – 2019 for the monitoring sites Ferndale, Grosvenor, Jacobs, Wentworth, Settlers School, Southern Works and Prospecton.

Figure 2 yields a number of important observations. Of the chosen monitoring sites, Southern Works, Wentworth and Jacobs recorded the highest yearly averages over the study period. In 2004, the maximum values recorded were reported

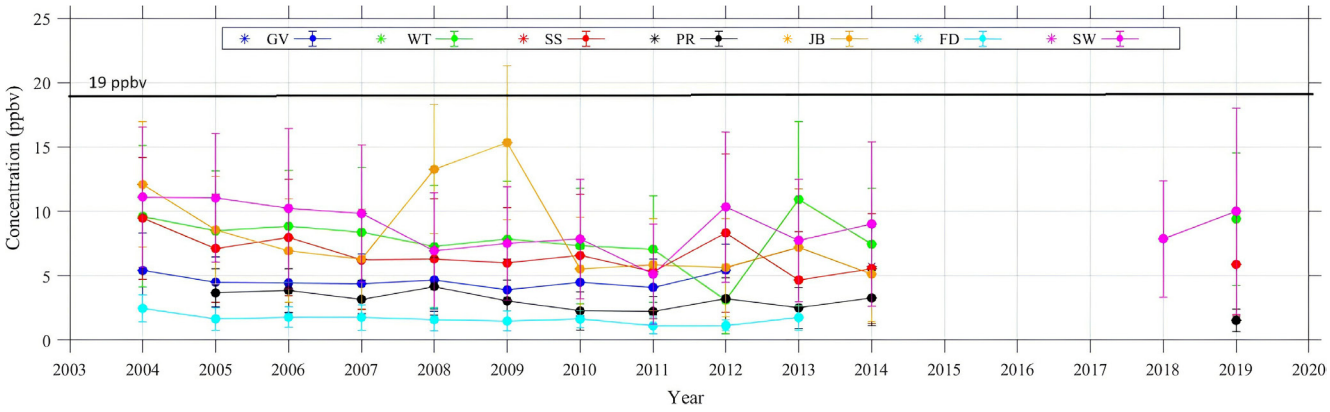


Figure 2: *Comparison of yearly average SO_2 concentration (ppbv) for the study period 2004 – 2019 at monitoring sites Ferndale (FD), Grosvenor (GV), Jacobs (JB), Wentworth (WT), Settlers School (SS), Southern Works (SW), Prospecton (PR). The permissible NAAQS early average exposure limit of 19ppbv is indicated by the solid black line for reference.*

at Jacobs (12.09 ± 4.87ppbv) and Southern Works (11.12 ± 5.46ppbv). Between 2005 – 2007 Southern Works recorded maximum yearly averages (for the dataset) in the range 11.04 ±4.99ppbv – 9.83 ±5.36ppbv, respectively. During 2008 and 2009, Jacobs reported maximum values of 13.27 ± 5.02ppbv and 15.34 ± 5.97ppbv. These levels significantly exceed those recorded at the remaining 6 sites. Yearly average SO_2 concentration recorded at Jacobs in 2009 is the highest value recorded for all seven sites (2004 – 2019). In 2011 and 2013, Wentworth recorded maximum values for the dataset of 7.06 ± 4.14ppbv and 10.92 ± 6.03ppbv respectively. During the remaining years 2010, 2012, 2014, 2018, 2019 Southern Works reported maximum values for the dataset, and these were in the range 7.85 ± 4.62ppbv to 10.34 ± 5.83ppbv.

From 2011 to 2014, Southern Works, Settlers school and Prospecton show a correlation in SO_2 levels. It is possible that Settlers School and Southern Works may have been influenced by the same emission sources which resulted in this behaviour, however, Prospecton is the most southerly site of the 7 locations, and it is therefore uncertain why this would show a similar pattern of SO_2 observations during this period (2011 – 2014).

The Prospecton and Ferndale monitoring sites are characterized by consistently low SO_2 concentrations with Ferndale recording the lowest yearly average for the study period, during 2011 (1.11 ± 0.64ppbv). Since the Ferndale monitoring site is situated north of Durban and is also some distance from main roads and heavy industry, these factors must therefore be the main contributing factor to the low SO_2 levels observed at this site. Finally, the guideline for permissible yearly average SO_2 exposure is 19ppbv and for the period of study, all yearly average values for the seven monitoring stations dataset are below this limit.

For each location, maximum and minimum yearly average (2004 – 2019) were determined together with difference between maximum and minimum values. In each case, a linear trend was fitted to the data and the gradient and norm of residuals of the trend calculated. Because of the substantial data gap, 2015 – 2018 (Southern Works), 2015 - 2019 (remaining sites), the linear trend was fitted for the period 2004 - 2014 and projected data for 2018, 2019 (Southern Works), 2019 (all remaining sites), compared with actual data. It should be noted that actual data only exists for 2018, 2019 (Southern Works), 2019 (Wentworth, Settlers School, Prospecton). For comparison, Table 2 shows maximum value and year, minimum value and year, difference between maximum and minimum values, linear trend fitted to yearly average data (2004-2014) and projected yearly average data (2019) for seven monitoring sites. Actual data recorded at Wentworth, Settlers School, Prospecton (2019), Southern Works (2018, 2019) is also presented. Maximum, minimum yearly average values and actual data (2018, 2019) are expressed with standard deviation in parenthesis.

Consideration of Table 2 indicates that Jacobs shows the largest maximum value for the dataset, namely 15.34 ± 5.97ppbv recorded in 2009. Southern Works, Wentworth and Settlers School all record maximum values in the range 9.47 – 11.12ppbv. The locations Grosvenor, Prospecton and Ferndale all record significantly lower maximum values for their datasets with these values in the range 2.47 - 5.45ppbv. Furthermore, three sites record dataset maximum values in 2004 and these are Ferndale, Settlers School and Southern Works. There is a second grouping of dataset maximum values in 2008-2009 and these are recorded at Prospecton and Jacobs respectively. There is a third grouping of dataset maximum values in 2012-2013 recorded at Grosvenor and Wentworth respectively. The observation that maximum values appear to fall into three groupings seems to indicate that the three groupings reflect similar causal factors at specific locations at similar times. Both Settlers School and Southern Works are located within close proximity to each

Table 2: *Maximum value and year, minimum value and year, difference between maximum and minimum values, linear trend fitted to yearly average data (2004-2014) and projected yearly average data (2018, 2019) for seven monitoring sites. For comparison, actual data recorded at Wentworth, Settlers School, Prospecton (2019), Southern Works (2018, 2019) is also presented. Maximum and minimum yearly average values are expressed with standard deviation in parenthesis.*

Monitoring site	Maximum of data set (ppbv) and year	Minimum of data set (ppbv) and year	ΔSO_2 (ppbv)	Gradient of linear trend (ppbv yr^{-1})	Norm of residuals	Projected yearly average 2019 (ppbv)	Actual yearly average 2019 (ppbv)
Ferndale	2.47 (1.03) 2004	1.11 (0.64) 2011	1.36	-0.084	0.84	0.75	No data
Grosvenor	5.45 (2.01) 2012	3.91 (2.37) 2009	1.54	-0.024	1.47	4.34	No data
Jacobs	15.34 (5.97) 2009	5.13 (3.65) 2014	10.21	-0.48	10.07	3.55	No data
Wentworth	10.92 (6.03) 2013	3.13 (2.63) 2012	7.80	-0.19	5.83	5.97	9.40 (5.14)
Settlers School	9.47 (4.73) 2004	4.68 (3.76) 2013	4.79	-0.27	3.51	3.94	5.88 (4.01)
Southern Works	11.12 (5.46) 2004	5.12 (3.88) 2011	6.00	-0.29	5.22	2018: 6.19 2019: 5.92	2018: 7.89 (4.53) 2019: 10.02 (8.00)
Prospecton	4.17 (2.24) 2008	1.53 (0.86) 2019	2.64	-0.12	1.66	2.02	1.53 (0.86)

Table 3: *Results obtained from Gustella and Mjoli (2005) for SO_2 five-year mean (1998-2002) and SO_2 mean for 2003 with data from this investigation (2004).*

Location	Five-year mean (1998 – 2002) (ppbv) from Gustella and Mjoli (2005)	Mean for 2003 (ppbv) from Gustella and Mjoli (2005)	Mean for 2004 (ppbv) from present study
Wentworth	18	11	9.61
Southern Works	26	21	11.12
Settlers School	15	14	9.46

other, and Wentworth and Grosvenor form a further pair of locations presumably similarly impacted by the same factors at approximately the same time. Minimum values recorded for the dataset seem to follow a less well-defined pattern. For all sites excluding Prospecton (minimum recorded in 2019), minima for the remaining six sites are recorded in the range 2009 – 2014.

Table 2 also illustrates that the greatest difference in maximum and minimum values is seen in the data recorded at the Jacobs site (10.21ppbv). This is followed by Wentworth (7.80ppbv), Southern Works (6.00ppbv) and Settlers School (4.79ppbv). These sites are also located closest to the Durban South Industrial Basin (DSIB) and are presumably more directly impacted by variability in the emissions from the heavy industry in this area. Sites such as Ferndale and Prospecton show smaller differences between the maximum and minimum values for the period of study. These sites are located further from the heart of the DSIB – taking Jacobs as a central location, Ferndale and Prospecton are located 16.72 km and 9.59 km respectively from the Jacobs monitoring site. These smaller differences between maximum and minimum values are presumably related to the fact that emissions have had the chance to disperse substantially before impacting at these sites.

Inspection of Table 2 shows that fitted trends for all sites are negative (2004-2014). The site with the largest negative trend is Jacobs (-0.48ppbv yr^{-1}). This is followed by Southern Works with a value of -0.29 ppbv yr^{-1}. The remaining sites fall within the range -0.024 to -0.27ppbv yr^{-1} in decreasing order: Settlers School (-0.27ppbv yr^{-1}), Wentworth (-0.19ppbv yr^{-1}), Prospecton (-0.12ppbv yr^{-1}), Ferndale (-0.084ppbv yr^{-1}) and Grosvenor (-0.024ppbv yr^{-1}). It is important to note that although the gradients of the linear fits in Table 2 are small, they do indicate that SO_2 emissions at all monitoring sites have decreased over the study period, which is a significant observation.

Comparison of actual data 2018, 2019 (Southern Works), 2019 (Wentworth, Settlers School, Prospecton) with projected data obtained from the fitted linear trend, indicates that except for Prospecton, projected yearly averages are lower than actual yearly averages for these sites. The largest difference between projected and actual data occurs at Southern Works (2019) with a difference of 4.10pppbv. For the remaining sites this difference in descending order is Wentworth (3.43ppbv), Settlers School (1.94ppbv) and Southern Works for 2018 (1.70ppbv). These sites are all located in the DSIB and it is concerning to note that significant differences between actual and projected data are reported for these locations. The case of Prospecton is interesting in that it records a smaller than projected value for 2019, indicating that SO_2 remediation measures in this location are resulting in a positive impact on ambient SO_2 levels. It should also be noted that this site is located in the extreme south of the chosen study area and is not characterized by the high degree of heavy industry as seen in the DSIB. Considering the above, it is important to note, that although three sites report increases in actual data relative to projected data (which is a concern), the projected data falls withing the standard deviation associated with the actual data in all cases (including Prospecton). This illustrates that although the presence of the data gap within the dataset is not optimal, its effect is not as significant as might be initially expected and the comparison of actual and projected data (2018, 2019) is a useful tool for validating the dataset.

As an aside, a detailed study of the major SO_2 emitters in close proximity to these sites would be useful in determining what has changed 2004-2014 relative to 2018, 2019 to produce the observed increases. This may be related to changes in the dominant types of industry in these areas, but also to possible modification in manufacturing processes that may result in increased SO_2 production. A thorough investigation in this regard could form the basis of a future investigation.

Using the linear fit determined from yearly average data for the seven sites (2004-2014) and projected data (2019), reduction in SO_2 levels (2004-2019) corresponded to the following sequence: Jacobs (67%), Ferndale (63%), Settlers School (50%), Southern Works (47%), Prospecton (47%), Wentworth (32%) and Grosvenor (8%). It is interesting to note that the two largest reductions are seen at an industrial site followed by an urban one. However as discussed previously, this finding should be placed in the contexts of the increases in actual data compared to projected data seen for Wentworth, Settlers School, Southern Works and Prospecton.

In Diab and Motha (2007), data were continuously monitored at Wentworth (1958 – 2005) and results from a long term SO_2 trend study presented. Data (µg m^{-3}) were derived from SO_2 bubbler records based on the hydrogen peroxide method of collection. Significant changes in ambient air quality were noted, namely extended periods of increasing trends in SO_2 interspersed with shorter periods of decreasing trends. During the 1980's and early 1990's an increasing trend in SO_2 levels was reported with levels exceeding 80 µg m^{-3} (30ppbv) in 1989 and 1991 and approximating maximum levels recorded in 1962. After 1991, SO_2 concentrations declined, and levels stabilized between 50 – 60 µg m^{-3} (19 – 23ppbv) over the following few years. Since

1998 they reported a further decline to a mean annual value of approximately 40 µg m^{-3} (15 ppbv) by the end of the dataset (2005). This should be compared with the values obtained in this work, the maximum value recorded at Wentworth is 10.92ppbv (2013) with maximum yearly average and minimum yearly average bound in the range 7.80ppbv. Furthermore, the continuation of the decline in yearly average SO_2 since 1998, as reported by Diab and Motha (2007), is confirmed by this work through the negative linear trend fitted to Wentworth yearly average data presented here. It is unfortunate that Diab and Motha (2007) only provided a descriptive analysis of their dataset, and a linear trend was not calculated (1958 – 2005), hence no direct comparison of a linear trend is possible with the present investigation. Comparison is further limited as Diab and Motha (2007) only contained data recorded at Wentworth.

In Guastella and Mjoli (2005), data from permanent SO_2 monitoring stations at Wentworth, Southern Works, AECI and Settlers School were presented. Measurements of SO_2 were undertaken by means of continuous ultraviolet fluorescence analysers with data logged at five-minute intervals. Data were recorded for the period 1996 – 2003 at Wentworth, Southern Works and AECI while data acquisition only started at Settlers School from 2000 resulting in a comparably shorter dataset (2000 – 2002). The authors reported that SO_2 concentrations were highest at Southern Works where the National Guideline of 19ppbv was exceeded for all years, except 2002. Data recorded at Wentworth showed that this guideline was also exceeded for the years 1996 – 1999. They emphasized that for the early part of their dataset, concentrations at Southern Works and Wentworth were comparable however in the later part of their dataset, levels at Wentworth showed a sustained decrease. Although yearly average data were presented for the above-mentioned sites, no linear trends were fitted to it, other than to comment on the obvious average decline in levels as seen in each of the relevant plots. They did however present mean SO_2 values (1998 – 2002) in comparison with levels recorded in 2003 for Southern Works, Wentworth and Settlers School (the mean for Settlers School was only applicable for the period June 2000 – December 2002). Their findings included the following: At Southern Works, there was an increase in annual average for 2003 relative to 2002, however levels in 2003 were still below those of the 5-year mean, thereby indicating an improvement in air quality. At Wentworth they noted a substantial reduction in annual average for 2003 relative to the 5-year mean while the annual average 2003 at Settlers School was consistent with the mean of the previous two years. A comparison of results obtained by Guastellla and Mjoli (2005), with those presented in this investigation is given in Table 3.

Consideration of Table 3 indicates that historically (prior to the dataset presented here) SO_2 levels at Southern Works were substantially higher than those recorded at other monitoring stations and also exceeded in both instances (five year mean and mean for 2003) the yearly average exceedance of 19 ppbv but that by 2004, concentrations have decreased to below this recommended level. For all three sites, it is also seen that the 2004 yearly average value recorded in the present study is lower than the mean (1996 – 2002) and 2003 level as presented in Guastella and Mjoli (2005). This further supports the evidence that, on average, air quality has improved at these three locations for the period 1996 – 2004.

Monthly average trends and seasonal variation

In order to investigate month to month variation in SO_2 levels at each monitoring site, monthly averages were calculated for the entire dataset and for each location. Figure 3 shows a composite

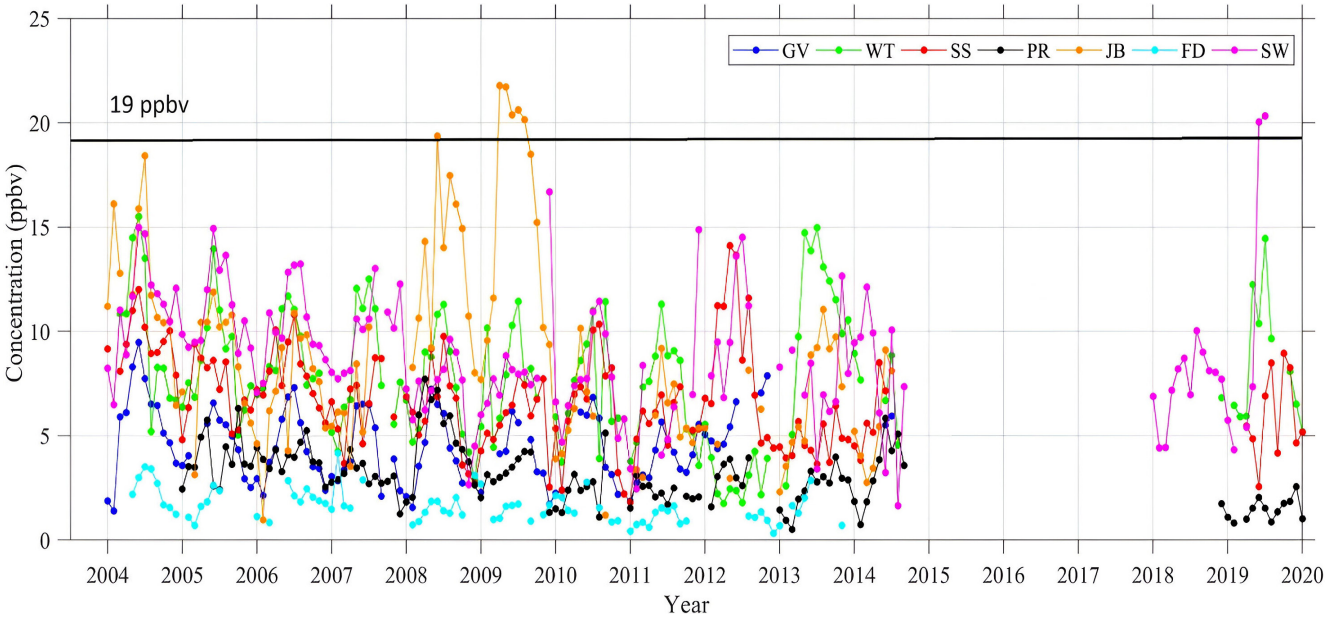


Figure 3: *Comparison of monthly average SO_2 concentrations (2004 – 2019) for the monitoring stations Grovenor (GR), Wentorth (WT), Settlers School (SS), Prospecton (PR) Jacobs (JB), Ferndale (FD) and Southern Works (SW). The permissible NAAQS yearly average exposure limit of 19ppbv is indicated by the solid black line for reference.*

of seven plots, each illustrating the monthly average 2004 – 2019 for the monitoring sites Ferndale, Grosvenor, Jacobs, Wentworth, Settlers School, Southern Works and Prospecton. For clarity, error bars have been removed in this figure. This determination of SO_2 monthly averages, allows the dataset to be investigated on a shorter time scale in order to determine more accurately, individual exposures experienced by those living in close proximity to the study sites.

In order to investigate month to month variation in SO_2 levels at each monitoring site, monthly averages were calculated for the entire dataset and for each location. Figure 3 shows a composite of seven plots, each illustrating the monthly average 2004 – 2019 for the monitoring sites Ferndale, Grosvenor, Jacobs, Wentworth, Settlers School, Southern Works and Prospecton. For clarity, error bars have been removed in this figure. This determination of SO_2 monthly averages, allows the dataset to be investigated on a shorter time scale in order to determine more accurately, individual exposures experienced by those living in close proximity to the study sites.

Figure 3 indicates that on average, all data sets exhibit the well documented pattern as observed by Diab et al. (2002) and Guastella and Mjoli (2005), namely high SO_2 concentrations in winter and low concentrations in summer. When datasets for multiple years are displayed together, a characteristic periodic behaviour is seen. This observation is a direct consequence of poor dispersion conditions experienced in Durban during winter. Temperature inversions trap a layer of air close to the ground causing the build-up of pollutants at this low level. This effect is further amplified by the fact that the DSIB is located in a basin type structure which favours the formation of temperature inversions. These factors all contribute to produce elevated SO_2 levels in the cooler months. The observations of these previous studies are therefore reaffirmed by the present investigation.

Comparison of the monthly data sets reveals that the highest monthly averages (2004 - 2019) are consistently observed at one of the three sites: Wentworth, Jacobs or Southern Works. The exception to this trend is Settlers School which records values comparable to those at Southern Works during Winter 2010 and Winter 2012. The highest monthly averages for the whole dataset were recorded at Jacobs during 2008 and 2009 with a maximum (21.71 ±2.82ppbv) measured during June 2009. June 2008 also recorded elevated SO_2 levels (19.36 ± 3.23ppbv) at this site relative to the rest of the datasets. It is interesting to note that the years 2008 – 2010 saw unusually high concentrations of SO_2 at this location. This is presumably attributable to locally increased emissions from industry in the area such as change in operational procedures with resulting impact on emission profiles.

The lowest monthly averages for the whole dataset are consistently recorded at Prospecton and Ferndale and these fall within the approximate range 1 – 7ppbv. Monthly average data for all monitoring stations appears to follow an approximately downward trend as would be expected given the negative linear trend as determined from yearly average SO_2 data.

Maximum and minimum monthly averages for each monitoring site (2004 - 2019) were determined and are presented in Table 4. Anomalous readings that do not conform to the well documented pattern of maximum SO_2 levels in winter and minimum SO_2 levels in Summer are indicated in Table 4 by (*).

It is noted from Table 4 that in most instances all sites conform to the expected periodic pattern of maximum and minimum levels. However, there are several instances that do not adhere to this pattern namely: Ferndale – maximum monthly average recorded in February 2007 and minimum monthly average measured in May 2012 at Wentworth. For the anomalous maximum reading, it possible that this corresponds to non-standard emissions from industry during this time and in close proximity to this site such that this effect dominated over the increased dispersion conditions typically experienced in summer. This would be unusual given that the Ferndale site is situated some distance from industrial sources.

For the anomalous minimum value recorded at Wentworth, observed high wind-speed from a south westerly direction may have also favoured the dispersion of pollutants. The case of Prospecton is puzzling as both minimum and maximum monthly average do not adhere to the established seasonal trend. The reason for this behaviour is unknown. The monitoring sites,

Table 4: *Maximum and minimum monthly averages for each monitoring site (2004 - 2019). Anomalous readings that do not conform to the well documented pattern of maximum SO_2 levels in Winter and minimum SO_2 levels in Summer are indicated by (*). Maximum and minimum values expressed with standard deviation in parenthesis.*

Monitoring site	Maximum monthly average (2004 - 2019) (ppbv) and date	Minimum monthly average (2004 - 2019) (ppbv) and date
Ferndale	4.16 (0.35) February 2007 (*)	0.32 (0.04) December 2012
Grosvenor	9.47 (2.50) June 2004	1.56 (1.14) February 2008
Jacobs	21.71 (2.82) June 2009	0.96 (0.61) February 2006
Wentworth	15.50 (3.80) June 2004	1.75 (1.35) May 2012 (*)
Settlers School	14.40 (6.68) June 2012	1.81 (0.54) January 2011
Southern Works	20.32 (7.26) July 2019	2.46 (1.37) February 2011
Prospecton	7.71 (1.91) October 2008 (*)	0.51 (0.19) March 2013 (*)

Southern Works, Settlers School, Jacobs and Grosvenor do not show any anomalous behavior in terms of the typical seasonal trend.

In Guastella and Mjoli (2005), the following long-term trends in monthly averages were noted: For Wentworth, distinct seasonal fluctuations in SO_2 levels were recorded for the period 1996 – 2003 with monthly average concentrations measured in winter higher than those recorded in the summer months. This dataset indicated a general decrease in SO_2 concentrations from 1996 and the authors report that this trend is related to a decrease in SO_2 in winter as averages in summer were approximately constant. The authors propose that this reduction was related to reduction in emissions from refineries and coal-burning by smaller industries. At Southern Works, data collection for this location began in May 1998. Monthly averages (1998 - 2003) were seen to be fairly variable and the authors stated that this was clearly related to emission scenarios. This dataset showed a general decreasing trend over the study period with a slight increase in 2003. At Settlers School, data collection began from June 2000. Although this dataset is variable, the long-term trend in SO_2 showed an approximate decrease in levels. Monthly average SO_2 levels dropped to a minimum for the study period in December 2002. The results presented in the present investigation clearly confirm these previous observations in particular, maximum monthly averages recorded in cooler conditions with the converse also confirmed. The continued decline in SO_2 monthly levels reported in this work is clearly supported and expands upon this previous study.

In order to investigate seasonal variation in SO_2 levels at each monitoring site, average months were calculated for the entire dataset (2004 – 2019) and for each location. Figure 4 shows a composite of seven plots, illustrating the seasonal variation for the monitoring sites Ferndale, Grosvenor, Jacobs, Wentworth, Settlers School, Southern Works and Prospecton.

Figure 4 indicates that all monitoring stations record a maximum in the cooler months and minimum in warmer months, which is in agreement with the previous finding of Diab et al. (2002); Guastella and Mjoli (2005). In Durban, wind speeds are higher in spring (September – November) while low wind speeds are typically recorded in autumn / winter (April – June). High levels of pollutants are generally connected to low wind speed conditions in winter and at night / early morning. This is because of poor vertical mixing and low horizontal transport out of the source area (South Durban Basin multi-point plan case study report 2007), (Air quality scoping report, Royal Haskoning DHV, 2014). The observations presented in this investigation are clearly confirmed by the above findings.

From Figure 4 it is seen that Jacobs records the highest average seasonal values for the months June to November as well as February and April while Southern Works records the highest seasonal average for the months January and March. For the month of May, three sites are approximately equal, namely, Wentworth, Jacobs and Southern Works. The locations Ferndale and Prospecton consistently record the lowest seasonal SO_2 levels.

For all sites, the seasonal maximum is recorded in Winter (June – August) except for Prospecton which records a maximum in September. All sites record a minimum in Summer (December, January, February) except for Ferndale which records a minimum in early autumn (March). A possible explanation for this observation may be due to increased dispersion conditions at this site prevailing later in the year than at other locations.

The influence of biomass burning on ambient air pollution is well documented (Sinha et al. 2003; Behera and Balasubramanian 2014; Agbo et al. 2021). Savanna fires in Africa, constitute approximately two thirds of the savanna burned world-wide and chemical species released as emissions from these fires, include CO, NO_x, SO_2, hydrocarbons, halocarbons together with particulates. It has been reported in numerous sources (Crutzen and Andreae 1990; Blake et al. 1996; Yokelson et al. 1996 and 2003; Andreae and Merlet 2001, Sinha et al. 2003a) that the number of Southern African savanna fires peaks in the dry season (April – October). The effects of these fires on ambient air pollution levels are further amplified by meteorological

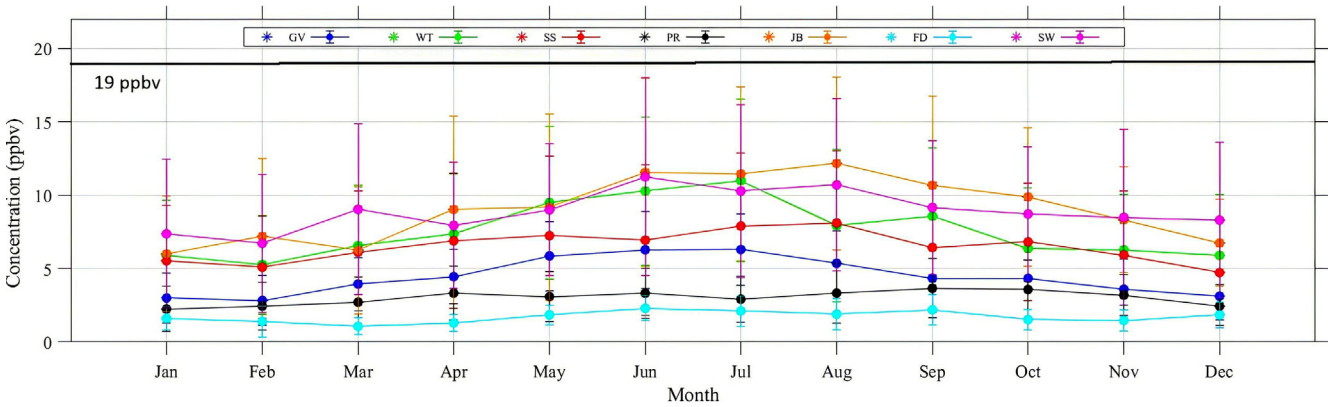


Figure 4: *Comparison of SO_2 seasonal variation (2004 – 2019) for the monitoring stations Grovenor (GV), Wentorth (WT), Settlers School (SS), Prospecton (PR) Jacobs (JB), Ferndale (FD) and Southern Works (SW). The permissible NAAQS yearly average exposure limit of 19ppbv is indicated by the solid black line for reference.*

conditions during this dry season such as stable air masses, southeasterly trade winds and subtropical high pressure over South Africa (McGregor and Nieuwolt 1998). Furthermore, the presence of stable layers reduces vertical mixing resulting in pollution build-up close to the surface (Cosijn and Tyson 1996; Hobbs 2002 and 2003).

Four sites, Ferndale, Grosvenor, Wentworth and Settlers School show a secondary maximum in September - October which (given the previous work mentioned above) may be attributed to the influence of biomass burning at these locations. Prospection is characterized by a single maximum in September, and it is possible that for this site, the impact of biomass burning dominates all other factors influencing the behaviour of SO_2 levels at this site.

Table 5 shows a comparison of time and level of maximum, minimum and percentage change in seasonal SO_2 levels over the year (between maximum and minimum), for the seven monitoring sites. The seasonal percentage change in SO_2 levels appears to fall into two separate groups. Ferndale, Grosvenor, Jacobs and Wentworth are characterized by seasonal percentage change in the range 122 – 103% while Settlers School, Southern Works and Prospecton are characterized by seasonal percentage change in the range 62 - 72%. Assuming that rainfall is approximately constant for all sites (and that emission profiles from industry are approximately the same throughout the year), it is possible that this observation is more closely related to other dispersion factors such as wind speed. High percentage changes indicate a greater variability in SO_2 levels over the year which must be related to conditions where more efficient stagnation mechanisms result in pollution build up in winter. The implication is that winds speeds are therefore lower in Winter at the sites Ferndale, Grosvenor, Jacobs and Wentworth, thereby resulting in higher levels of SO_2 and hence larger percentage changes between winter and summer levels. The converse is therefore that Prospecton, Settlers School and Southern Works experience higher wind speeds in winter, resulting in less build up pollutants near the surface and hence a smaller percentage change between winter and summer values at these locations. Prospecton is an interesting case in that the seasonal profile at this location is different from the remaining sites and this is probably the result of other factors impacting on the seasonal behaviour at this site.

Seasonal averages per year

In order to investigate the variation from season to season for each year of the dataset (7 sites), a seasonal average was calculated in the following way: For each site and each year of the dataset, an average for Spring (September, October, November), Summer (December, January, February), Autumn (March, April, May) and Winter (June, July, August) was calculated. The results are presented in Figure 5 which is a composite of seven plots, each illustrating the seasonal averages per year for the monitoring sites Ferndale, Grosvenor, Jacobs, Wentworth, Settlers School, Southern Works and Prospecton. The results are discussed individually for each site below. For clarity, error bars have been removed from the data points in the individual graphs.

For each monitoring site (and each year), difference between maximum and minimum seasonal value (or width of seasonal envelope) $\Delta_{Max,\ Min}$, was calculated together with associated percentage change. This difference between maximum and minimum values is clearly a combination of many factors such as changes in emission profile at the location but is possibly more representative of differences in meteorological factors (such as rainfall, wind speed) this is because the effect due to seasonal changes is generally expected to dominate over the effect of changes in emission profile over the time period of 3 months (1 season). For comparison purposes, the difference between minimum and maximum seasonal values is only significant for years where the average for all four seasons is available. Furthermore, for each monitoring site, the number of anomalous observations was calculated as a percentage of the total number of observations (where this is taken as the reduced data set using only years where data for all seasons is available). An anomalous observation is defined as not conforming to the well documented pattern of maximum in winter and minimum in summer. For each monitoring site, Table 6 shows $\Delta_{Max,\ Min}$ and

Table 5: *Maximum and minimum SO_2 seasonal values and corresponding percentage change over the year for seven monitoring sites. Sites with a secondary maximum are indicated by (*). Maximum and minimum values expressed with standard deviation in parenthesis.*

Monitoring site	Maximum of seasonal variation (ppbv) and month		Minimum of seasonal variation (ppbv) and month		Percentage change between maximum and minimum over year
Ferndale	2.29 (0.88) 2.19 (*) (1.04)	June September	1.06 (0.58)	March	116
Grosvenor	6.28 (2.62) 4.33 (*) (2.13)	June - July October	2.83 (1.70)	February	122
Jacobs	12.15 (5.91)	August	6.00 (3.93)	January	103
Wentworth	11.00 (5.55) 8.57 (*) (4.64)	July September	5.27 (3.34)	February	109
Settlers School	8.11 (4.92) 6.81 (*) (4.02)	August October	4.72 (3.57)	December	72
Southern Works	11.26 (6.73)	June	6.71 (4.70)	February	68
Prospecton	3.65 (2.02)	September	2.25 (1.53)	January	62

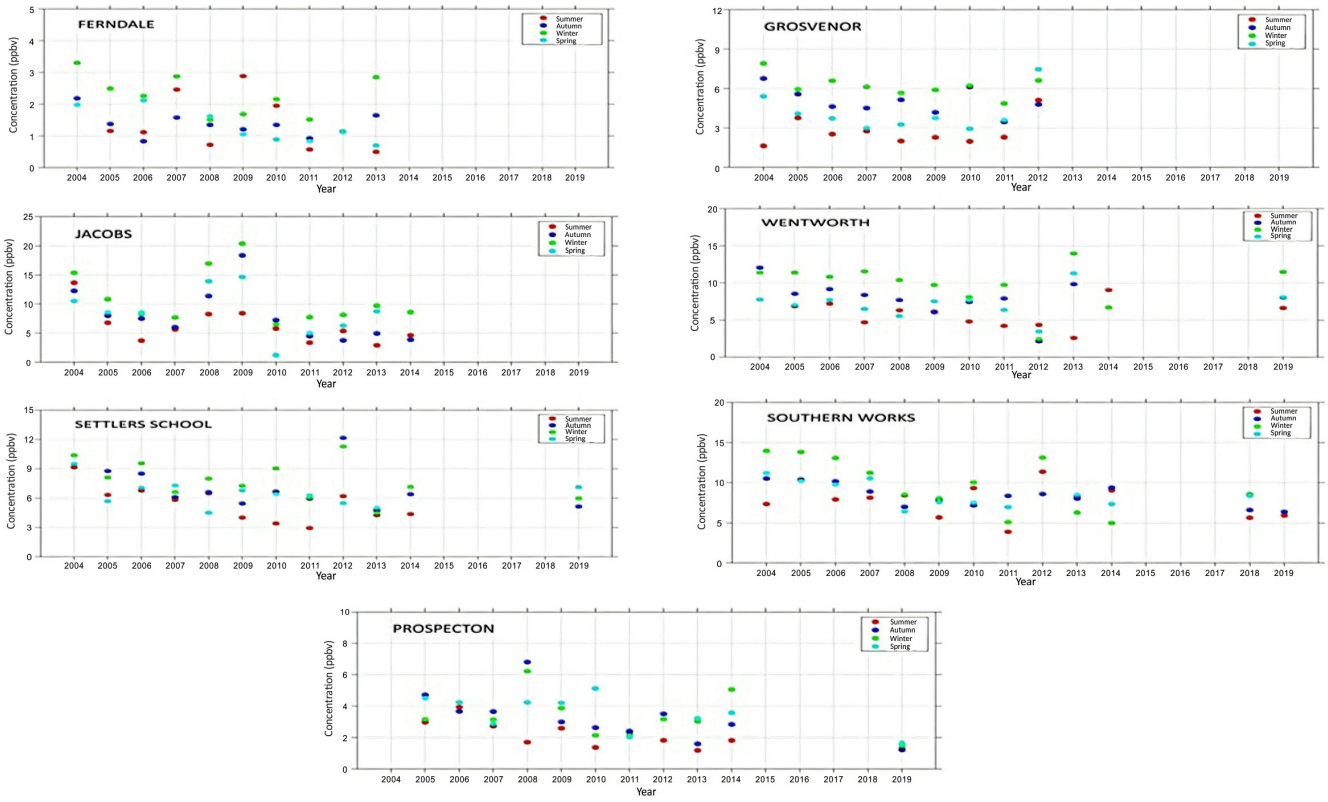


Figure 5: *Seasonal averages per year (ppbv) for the study period 2004 – 2019 at monitoring sites Ferndale, Grosvenor, Jacobs, Wentworth, Settlers School, Southern Works, Prospecton.*

corresponding percentage, average $\Delta_{Max, Min}$ (or average width of seasonal envelope) and percentage of anomalous observations in each data set. Values in red indicate years where data for all four seasons are available. Maximum and minimum $\Delta_{Max, Min}$ are indicated for each monitoring site.

Table 6 indicates that the largest $\Delta_{Max, Min}$ is recorded at Jacobs (11.95ppbv) in 2009 while the smallest $\Delta_{Max, Min}$ is recorded at Prospecton (0.38ppbv) in 2011. Furthermore, during 2004 Grosvenor and Southern Works both record maximum $\Delta_{Max, Min}$ while in 2013, Ferndale and Wentworth record maximum $\Delta_{Max, Min}$. In 2005, Grosvenor and Jacobs record minimum $\Delta_{Max, Min}$. In the case of 2004 and 2005, it is possible that this reflects real meteorological impacts on SO_2 levels as the sites Grosvenor and Southern Works (2004) and Grosvenor and Jacobs (2005) are in close proximity however the link between Ferndale and Wentworth (2013) may be more coincidental as these locations are further apart allowing for more significant differences in meteorological factors.

Comparison of average difference between seasonal maximum and seasonal minimum (or average width of seasonal envelope), for years with all seasonal data available, and for each location, is in descending order: Jacobs - 6.21ppbv, Wentworth - 5.05ppbv, Southern Works - 3.64ppbv, Grosvenor - 3.62ppbv, Settlers School - 3.35ppbv, Prospecton - 2.35ppbv and Ferndale - 1.43ppbv.

From the above it appears that Jacobs and Wentworth (5.05 – 6.21ppbv) form a separate grouping from Southern Works, Grosvenor and Settlers School (3.35 – 3.64ppbv). Prospecton and Ferndale have the lowest average difference between seasonal maximum and minimum 1.43 – 2.35ppbv). The average difference or width of envelope for seasonal average is clearly a combination of factors and it seems that at a minimum, the groupings outlined above constitute locations where the same sets of factors dominate in terms of SO_2 fluctuations. This is evidenced by consideration of the map of monitoring sites in that Jacobs and Wentworth are closely associated while (with the exception of Grosvenor) the same is true for the proximity of Southern Works and Settlers School. Finally, it is reasonable to assume that since Ferndale and Prospection are the furthest from the central DSIB, that they would be impacted by lower industrial emissions and a different set of meterological factors than those experienced in the DSIB.

In order to understand the above findings, an investigation of the location of emission sources in close proximity to each monitoring site and their individual contributions would prove difficult due to the very varied nature of industry at these sites, the complicated issue of quantifying the contribution from individual sources (e.g. factories/manufacturing plants, etc.) but also the temporal complexity of obtaining such data (if it existed at all) for the historical period 2004-2014. For these reasons, the authors have not pursued this line of enquiry in this work. There is however, limited meteorological information in the form wind direction and wind speed data for three of the sites, namely Grosvenor, Wentworth and Southern Works and this was used to generate wind roses for these locations. This analysis showed that during summer, the dominant wind direction for Grosvenor, Wentworth, Southern Works was NNE, NNE and NNW respectively while in winter, dominant wind direction was

WSW/SW, NNE and SW respectively. On average, wind speed at Wentworth was higher for all seasons compared to that at Grosvenor and Southern Works. Furthermore, assuming that 4ms^{-1} is a wind speed capable of efficient pollutant dispersal, the frequency of occurrence (%) of winds in excess of 4m^{-1}, over all directions, was calculated for the three sites, for summer and winter. This analysis showed that for Grosvenor, frequency of occurrence (%) of winds with speed > 4 ms^{-1} was 12.3 (summer), 15.5 (winter), for Wentworth the corresponding values were 49.0 (summer), 48.5 (winter) and for Southern Works 7.7 (summer), 12.2 (winter). From these results, a counter-intuitive pattern of behaviour is seen. It would be expected that since summer is characterized by lower levels of SO_2 relative to winter, it could be assumed that the frequency of occurrence for winds in excess of 4ms^{-1} should be higher during this season, however the opposite pattern is seen for Grosvenor and Southern Works, with both seasons of approximately equal frequency of occurrence for Wentworth. It is possible that this anomaly is related to wind direction. The phenomenon of higher wind speed increasing surface SO_2 levels has been highlighted in Diab et al. 2002. They reported that high SO_2 values of up to 40ppbv at Southern Works were recorded under low wind speed conditions. This relationship was observed until a critical speed of 3.5 ms^{-1} was reached, at this point a sustained rise in SO_2 corresponded to an increase in wind speed. An analogous observation was reported at Wentworth, where the critical wind speed was slightly higher, namely 4.5 ms^{-1}. They proposed that such behaviour was related to the location of monitoring stations in relation to principal SO_2 sources and could result from stack down-drafting in strong winds, resulting in high SO_2 concentrations close to the ground. They further suggested that this was indicative of SO_2 emissions originating from elevated sources such as industrial stacks. The above results from the present work appear to broadly support the findings of Diab et al. (2002) and show that similar sets of

Table 6: *For each monitoring site, difference between seasonal maximum and minimum $\Delta_{Max, Min}$ (ppbv) and corresponding percentage change, average $\Delta_{Max, Min}$ and percentage of anomalous observations in each data set. Values in red indicate years where data for all four seasons are available. Maximum and minimum $\Delta_{Max, Min}$ are indicated for each monitoring site.*

Year	Ferndale $\Delta_{Max, Min}$ %	Grosvenor $\Delta_{Max, Min}$ %	Jacobs $\Delta_{Max, Min}$ %	Wentworth $\Delta_{Max, Min}$ %	Settlers School $\Delta_{Max, Min}$ %	Southern Works $\Delta_{Max, Min}$ %	Prospecton $\Delta_{Max, Min}$
2004	1.32 66.4	6.27 MAX 383.8	4.81 45.7	4.28 55.2	1.22 13.3	6.60 MAX 89.7	_
2005	1.33 115.1	2.18 MIN 57.9	4.06 MIN 59.9	4.37 62.4	3.09 54.4	3.59 35.1	1.74 58.4
2006	1.29 154.0	4.06 160.7	4.81 129.1	3.62 50.2	2.51 35.6	5.15 65.1	0.56 15.1
2007	1.29 81.8	3.36 122.0	2.04 36.1	6.85 145.3	1.45 24.8	3.09 37.9	0.92 33.9
2008	0.90 MIN 123.9	3.65 182.2	8.68 105.0	4.84 87.7	3.48 77.1	2.05 31.9	5.11 MAX 299.1
2009	1.82 173.3	3.60 157.62	11.95 MAX 141.9	3.65 60.2	3.25 80.7	1.91 MIN 33.7	1.61 61.9
2010	1.26 115.9	4.23 213.8	6.04 507.8	3.28 68.3	5.63 164.5	2.86 39.8	3.75 273.1
2011	0.95 163.8	2.57 111.8	4.36 129.2	5.51 130.5	3.34 351.6	4.47 114.9	0.38 MIN 18.4
2012	-	2.66 55.6	4.36 115.7	1.92 MIN 79.7	6.68 MAX 121.7	4.52 52.5	1.68 92.2
2013	2.35 MAX 467.4	-	6.79 233.1	11.37 MAX 437.5	0.74 MIN 17.4	2.20 35.0	2.03 170.6
2014	-	-	4.72 121.9	2.36 35.3	2.76 63.1	4.40 88.4	3.24 178.1
2018	-	-	-	-	-	2.73 48.4	-
2019	-	-	-	4.86 73.2	1.96 38.1	0.45 45.4	0.38 30.7
Average $\Delta_{Max, Min}$ (ppbv)	1.43	3.62	6.21	5.05	3.35	3.64	2.35
Anomalous observations %	33.3	11.1	22.2	16.7	38.9	33.3	50.0

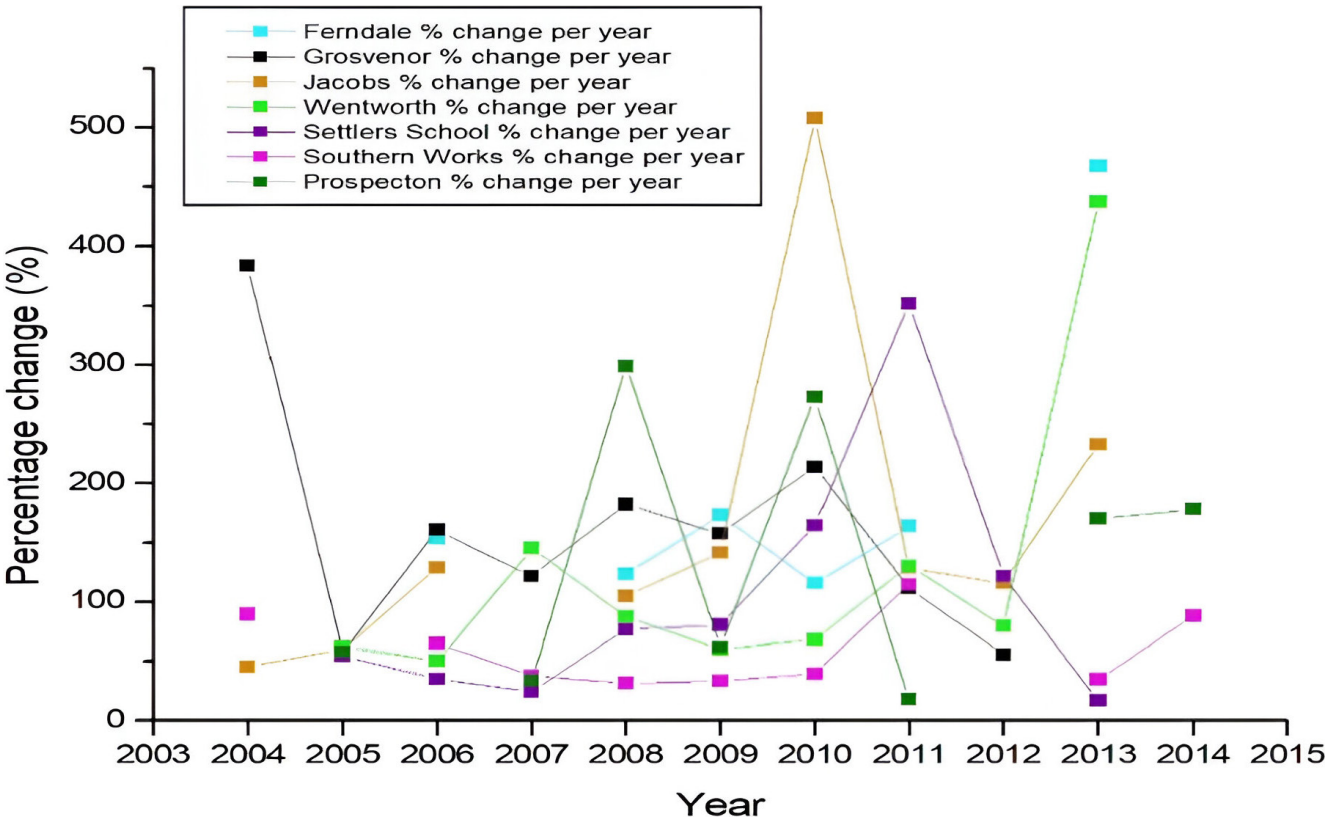


Figure 6: *Percentage change between SO_2 maximum seasonal value and SO_2 minimum seasonal value, for each location and years that have all seasonal data points.*

conditions are operative at Grosvenor and Southern works (as evidenced in this study by similar seasonal envelopes). The case of Wentworth clearly requires a more detailed study to fully determine relevant contributing factors.

Grouping the sites in terms of total number of anomalous seasons expressed as a percentage of the total (for the entire dataset) in descending order: Prospecton - 50.0%, Settlers School - 38.9%, Southern Works - 33.3%, Ferndale - 33.3%, Jacobs - 22.2%, Wentworth - 16.7%, Grosvenor - 11.1%. Excluding Prospecton, the percentage of anomalous seasons falls in a more closely bound range 22.2 – 38.9% for sites Jacobs, Ferndale, Settlers School and Southern Works. It is possible that this percentage change is related to the impact of dispersion factors at these sites, increased emissions or both. Wentworth and Grosvenor appear to form a further grouping. Finally, it is difficult to draw any further definite conclusions due to the small sizes of the data sets.

In order to compare seasonal variations between sites, percentage changes between SO_2 maximum seasonal value and minimum seasonal value were plotted for each year and for each location and are shown in Figure 6. This is only done for years for which four seasonal data points are available. It is difficult to draw any firm conclusions from this figure. From 2009-2011, there appears to be a correlation between Wentworth and Southern Works together with a correlation between Prospecton and Jacobs for the same time interval. A number of very large percentage changes occur in 2004 (Grosvenor), 2008 and 2010 (Prospecton), 2010 (Jacobs), 2011 (Settlers School) and 2013 (Wentworth). The cause of this behaviour is unknown.

Summary and conclusion

Trends in yearly average SO_2 levels at the seven sites indicated that no location exceeded the yearly average national guideline of 19ppbv for the study period (2004 – 2019).

Southern Works, Wentworth or Jacobs consistently recorded maximum yearly averages (2004-2019) and this is clearly a direct result of the location of these sites in the DSIB. In 2004, maximum was reported at Jacobs (12.09±4.87ppbv), and 2005–2007, Southern Works recorded maximum yearly averages ranging 11.04±4.99ppbv to 9.83±5.36ppbv, respectively. During 2008, 2009, maxima of 13.27±5.02ppbv and 15.34±5.97ppbv were observed at Jacobs. In 2011, 2013, Wentworth recorded maxima of 7.06±4.14ppbv and 10.92±6.03ppbv respectively. During 2010, 2012, 2014, 2018, 2019 Southern Works reported maximum values ranging 7.85±4.62ppbv to 10.34± 5.83ppbv. Ferndale consistently reported low SO_2 concentrations with minimum for the entire dataset measured in 2011 (1.11±0.64ppbv). These observations are clearly a consequence of its location in an urban environment.

Results of linear fitting to yearly average SO_2 data (2004-2014) indicated that SO_2 emissions show a negative downward trend

over the study period for all monitoring sites. The largest negative linear trend was recorded at Jacobs (-0.48ppbv yr^{-1}) while the smallest trends were observed at Ferndale (-0.084ppbv yr^{-1}) and Grosvenor (-0.024ppbv yr^{-1}). It is important to note that although the gradients of the linear fits are small, they do indicate that SO_2 emissions at all monitoring sites have decreased over the study period, which is a significant observation.

Comparison of actual data for Southern Works (2018, 2019) and Wentworth, Settlers School, Prospection (2019) with projected values determined from the fitted trends indicate that although there are increases in actual data relative to projected data (Southern Works, Wentworth, Settlers School) and a decrease (Prospection), the projected data values for these years still fall within the standard deviation of the actual data recorded. This illustrates that the effect of the data gap (2015-2018/2019) is not as significant as might have been expected and this comparison provides a useful tool for data validation.

Using the linear fit determined from yearly average data for the seven sites (2004-2014) and projected data (2019), reduction in SO_2 levels (2004-2019) corresponded to the following sequence: Jacobs (67%), Ferndale (63%), Settlers School (50%), Southern Works (47%), Prospecton (47%), Wentworth (32%) and Grosvenor (8%). It is interesting to note that the two largest reductions are seen at an industrial site followed by an urban one.

Monthly average SO_2 levels at all sites, displayed the well-documented periodic behaviour of higher concentrations recorded in winter compared to lower concentrations in summer. This is in agreement with previous studies (Diab et al. 2002, who reported trends for Wentworth and Southern Works). The highest monthly average values were consistently recorded at one of the three sites: Wentworth, Jacobs or Southern Works. The highest value of the dataset was 21.71 ± 2.82 ppbv recorded at Jacobs in June 2009. The lowest monthly averages are consistently recorded at Prospecton and Ferndale with the lowest value of this data set (0.32 ± 0.04 ppbv) recorded at Ferndale in December 2012.

Seasonal variation in SO_2 concentration were investigated and determined that Jacobs recorded the highest average seasonal values for the months June to November as well as February and April while Southern Works recorded the highest seasonal average for the months January and March. For the month of May, three sites were approximately equal, namely, Wentworth, Jacobs and Southern Works. The locations Ferndale and Prospecton consistently reported the lowest seasonal SO_2 levels.

For all sites, the seasonal maximum was recorded in Winter (June – August) except for Prospecton which recorded a maximum in September. All sites recorded a minimum in Summer (December, January, February) except for Ferndale which recorded a minimum in early autumn (March). Four sites, Ferndale, Grosvenor, Wentworth and Settlers School reported a secondary maximum in September – October. This may indicate the impact of biomass burning at these locations as the effect of biomass burning on surface SO_2 levels is well documented (Agbo et al. 2021). Prospection is characterized by a single maximum in September, and it is possible that for this site, the influence of biomass burning dominates all other factors influencing the behaviour of SO_2 levels at this site.

It should be noted that few investigations of seasonal averages in Durban exist which emphasizes the importance of the present work. Two short studies (both less than two years in duration) reported seasonal behaviour at monitoring sites including some of those employed in this study. In Naidoo et al. (2007), SO_2 was monitored at 16 sites using ultraviolet fluorescence spectrometry methods (January 2004 – October 2005). In a more recent investigation, Tularam et al., 2020 employed Ogawa passive samplers to measure SO_2 levels at 40 monitoring sites, including 23 sites located in the south of Durban and 17 sites in the north (July 2015 to June 2016). Both of these studies reported strong seasonality with on average, high levels were recorded in winter (June – August) and the lowest levels in summer (December – February).

For the first time, this investigation presented analysis of seasonal averages for each site using a data set longer than 2 years. These seasonal averages illustrated that on average, cooler conditions generally favoured higher SO_2 levels and warmer conditions were characterized by lower SO_2 concentrations – which is in alignment with previous studies. However, there are exceptions to this rule with Prospection showing the greatest number of anomalous incidences to this general rule and Grosvenor the least number of exceptions.

Comparison of average difference between seasonal maximum and seasonal minimum (or average width of seasonal envelope), for years with all seasonal data available, and for each location, indicated that Jacobs and Wentworth formed a separate grouping from Southern Works, Grosvenor and Settlers School. Prospecton and Ferndale had the lowest average difference between seasonal maximum and minimum. The average difference or width of envelope for seasonal average is clearly a combination of factors and it seems that at a minimum, the groupings outlined above constitute locations where the same sets of factors dominate in terms of SO_2 fluctuations. The finding of this grouping is significant as it highlights the range of seasonal variation for industrial sites vs urban locations, with associated implications for personal exposure. It should be noted that the data sets for the calculated seasonal averages at each site are relatively small and that a larger, more complete data set would allow for a more accurate determination of the above grouping. This is clearly an area where further research is required.

Finally, through the use of a substantially larger dataset coupled with a range of averaging periods (yearly average, monthly average, seasonal variation, seasonal average) differences in exposure scenarios between industrial and urban locations have been determined together with the characterization of a more

detailed exposure profile for individuals living and working in the greater Ethekwini municipality (2004-2019).

Acknowledgements

The authors would like to thank South African Air Quality Information System (SAAQIS) and service providers for sharing data from the ground-based monitoring stations that made this research possible.

Author contributions

B.L. Duigan: Conceptualization and drafting of manuscript, S.K. Sangeeetha: Contribution towards method and data analysis and V. Sivakumar: Supervision and advice on research. All authors contributed towards results and discussion and finalizing the manuscript for journal submission.

References

Agbo E., Walgraeve C., Ikechukwu Eze J., Ugwoke P.E., Ukoha P.O and van Langenhove H., 2021, 'Review on ambient and indoor air pollution status in Africa', *Atmospheric Pollution Research*, 12: 243-260, https://doi.org/10.1016/j.apr.2020.11.006

Air quality scoping report: prepared for the southern wastewater treatment works, Royal Haskoning DHV, 09 March 2014

Andreae M.O. and Merlet P., 2001, 'Emission of trace gases and aerosols from biomass burning', *Global Biogeochem. Cycles*, 15, no 3: 955-966, https://doi.org/10.1029/2000gb001382

Behera S.N., and Balasubramanian R., Influence of biomass burning on temporal and diurnal variations of acidic gases, particulate nitrate, and sulphate in a tropical urban atmosphere, 2014, *Advances in Meteorology*, 2014: 1-13, https://doi.org/10.1155/2014/828491

Blake N.J., Blake D.R, Sive B.C., Chen T-Y, Sherwood Roland F., Collins J.E., Sachse G.W. and Anderson B.E., 1996, 'Biomass burning emissions and vertical distribution of atmospheric methyl halides and other reduced carbon gases in the South Atlantic region', *J. Geophys. Res.*, 101, no. D19: 24151-24164, https://doi.org/10.1029/96jd00561

Carslaw N. and Carslaw D., 2001, 'The gas-phase chemistry of urban environments', *Surveys in Geophysics*, 22, no. 1: 31-53, https://doi.org/10.1023/a:1010601507383

Cosijn C. and Tyson P.D., 1996, 'Stable discontinuities in the atmosphere over South Africa', *S. Afr. J. Sci.*, 92: 381-385, https://journals.co.za > doi › pdf › AJA00382353_7756

Crutzen P.J. and Andreae M.O., 1990, 'Biomass burning in the tropics: Impact on atmospheric chemistry and biochemical cycles', *Science*, 250, no. 4988: 1669-1678, https://doi.org/10.1126/science.250.4988.1669

Diab R., Prause A. and Bencherif H., 2002, 'Analysis of SO_2 pollution in the south Durban industrial basin', *S. Afr. J. Sci.*, 98: 543-546, https://journals.co.za › doi › pdf › EJC97430

Diab R.D. and Motha A., 2007, 'An analysis of key institutional factors influencing air quality in south Durban using the DPSIR framework', *South African Geographical Journal*, 89, no. 1: 22-33, https://doi.org/10.1080/03736245.2007.9713869

Ethekwini air quality monitoring network – Annual Report 2009, Pollution control section, saaqis.environment.gov.za

Guastella L and Mjoli D., 2005, 'Sulphur dioxide measurements in South Durban: The culmination of 8 years of monitoring', *Clean Air Journal*, 14, no. 1, https://doi.org/10.17159/caj/2005/14/1.7158

Guastella L., Back of port – Situation assessment: air quality, Ethekwini municipality, 2008

Gounden Y., 2006, 'Ambient sulphur dioxide (SO_2) and particulate matter (PM_{10}) concentrations measured in selected communities of North and South Durban', M Med Sci Thesis, UKZN

Government Gazette – Republic of South Africa, Vol 534, No. 32816, Pretoria 24 December 2009

Hewitt C.N., 2002, 'The atmospheric chemistry of sulphur and nitrogen in power station plumes', *Atmospheric Environment*, 35, no. 7: 1155-1170, https://doi.org/10.1016/s1352-2310(00)00463-5

Hobbs P.V., 2002, 'Clean air slots amid atmospheric pollution', *Nature*, 415, no. 6874: 861, https://doi.org/10.1038/415861a

Hobbs P.V., 2003, 'Clean air slots amid dense atmospheric pollution in southern Africa', *J. Geophys. Res.*, 108, no. D13: 8490, https://doi.org/10.1029/2002jd002156

Josipovic M., Annegarn H., Kneen M.A., Pienaar J.J. and Piketh S.J., 2010, 'Concentrations, distributions and critical level exceedance assessment of SO_2, NO_2 and O_3 in South Africa', *Environ Monit Assess.*, 171, no. 1-4: 181-196, https://doi.org/10.1007/s10661-009-1270-5

Khumalo T., 2020, State of air report and national air quality indicator, Powerpoint presentation, 4 – 6 October 2020, 15th Air Quality Governance Lekgotla, Kempton Park – Gauteng, Department of environmental affairs RSA.

Lee H.K., Khaine I., Kwak M.J., Jang J.H., Lee T.Y., Lee J.K., Kim I.R., Kim W.I., Kyeong S.O. and Woo S.Y., 2017, 'The relationship between SO_2 exposure and plant physiology: A mini review', *Hortic. Environ. Biotechnol.*, 58, no. 6: 523-529, https://doi.org/10.1007/s13580-017-0053-0

Masiol M., Agostinelli C., Formenton G., Tarabotti E. and Pavoni B., 2014, 'Thirteen years of air pollution hourly monitoring in a

large city: Potential sources, trends and effects of car-free days', *Science of The Total Environment*, 494-495: 84-96, https://doi.org/10.1016/j.scitotenv.2014.06.122

Matooane M. and Diab R., 2003, 'Health risk assessment for sulphur dioxide pollution in south Durban, South Africa', *Archives of Environmental Health*, 58, no. 12: 763-770, https://doi.org/10.3200/aeoh.58.12.763-770

McGregor G.R. and Nieuwolt S., 1998, Tropical climatology: an introduction to the climates of the low latitudes (Second edition), John Wiley & Sons (Chichester), https://doi.org/10.1002/(sici)1097-0088(199909)19:11<1279::aid-joc444>3.0.co;2-u

Mdluli T., 2015, State of air report and national air quality indicator, Powerpoint presentation, 28 September 2015, 10th Air Quality Governance Lekgotla, Bloemfontein, Department of Environmental Affairs, RSA.

Mentz G., Robins T.G., Batterman S. and Naidoo R.N., 2018, 'Acute respiratory symptoms associated with short term fluctuations in ambient pollutants among schoolchildren in Durban, South Africa', *Environmental Pollution*, 233: 529-539, https://doi.org/10.1016/j.envpol.2017.10.108

Naidoo R., N. Gqaleni, S. Batterman and T. Robins, Multipoint Plan: Project 4: Health Study and Health Risk assessment (Final Project Report) – South Durban Health Study, Research principals, February 2007

Pham M., Muller J.F., Brasseur G.P., Granier C. and Megie G., 1995, 'A three-dimensional study of the tropospheric sulphur cycle', *Journal of Geophysical Research*, 100, no D12, 26061-26092, https://doi.org/10.1029/95JD02095

South Durban basin multi-point plan case study report, Air quality act implementation: Air quality management planning. Series C, book 12, Zanokuhle Environmental Service (ZES): lisa@zes.co.za, 2007

Sinha P., Hobbs P.V., Yokelson R.J., Blake D.R., Gao S. and Kirchstetter T.W., 2003, 'Distributions of trace gases and aerosols during the dry biomass burning season in southern Africa', *Journal of Geophysical Research*, 108, no D17: 4536-4559, https://doi.org/10.1029/2003jd003691

Sinha P., Hobbs P.V., and Yokelson R.J., Bertschi I.T., Blake D.R., Simpson I.J., Gao S., Kirchstetter T.W. and Novakov T., Emissions of trace gases and particles from savanna fires in southern Africa, *J. Geophys. Res.*, 108, no. D13: 8487, https://doi.org/10.1029/2002jd002325

Thambiran T. and Diab R., 2010, 'A review of scientific linkages and interactions between climate change and air quality, with implications for air quality management in South Africa', *S.Afr.J.Sci*, 106, no. 3/4: 1-8, https://doi.org/10.4102/sajs.v106i3/4.56

Tularam H., L.F. Ramsay, S. Muttoo, R.N. Naidoo, B. Brunekreef, K. Meliefste and K. de Hoogh, Harbour and intra-city drivers of air pollution: Findings from a land use regression model, Durban, South Africa, *International Journal of Environmental Research and Public Health*, 17, 5406-5422, 2020, https://doi.org/10.3390/ijerph17155406

Yokelson R.J., Griffith D.W.T. and Ward D.W., 1996, 'Open-path Fourier transform infrared studies of large-scale laboratory biomass fires', *J. Geophys. Res.*, 101, no. D15: 21067-21080, https://doi.org/10.1029/96jd01800

Yokelson R.J., Bertschi I.T, Christian T.J., Hobbs P.V., Ward D.E., and Hao W.M., 2003, 'Trace gas measurements in nascent, aged and cloud-processed smoke from African savanna fires by air-bourne Fourier transform infrared (AFTIR) spectroscopy', *J. Geophys. Res.*, 108, no. D13: 8478, https://doi.org/10.1029/2002jd002322

Research article

The impact of vehicle parameters on non-exhaust traffic road PM_{10} emissions: A case in South African low-income settlement

Ncobile C Nkosi[*1], **Roelof P Burger**[2], **Christiaan Pauw**[3], **Nisa Ayob**[1], **and Stuart J Piketh**[2]

[1]Unit for Environmental Sciences and Management, North-West University, Mafikeng, South Africa
[2]Unit for Environmental Sciences and Management, North-West University, Potchefstroom, South Africa
[3]NOVA Institute, Pretoria, South Africa
*Corresponding author: ncobile.nkosi@nwu.ac.za

Received: 4 April 2023 - **Reviewed:** 11 July 2023 - **Accepted:** 9 November 2023
https://doi.org/10.17159/caj/2023/33/2.15497

Abstract

About 70% of the roads in low-income settlements are unpaved and close (≤15m) to the residences, thus a major source of ambient and indoor PM_{10} concentrations. International studies have suggested that decreasing vehicle speed and managing vehicle type on paved or unpaved roads can reduce vehicle dust emissions. These mitigation strategies should be examined before being adopted and gazetted into the South African air quality management plan. This study aimed to characterise roads and traffic, emphasising determining the impact of vehicle type on PM_{10} emissions. GIS was used to determine the proportion of paved and unpaved roads. A traffic counter was used to monitor vehicles and to determine traffic composition, diurnal cycles, and average speed per road type. Field campaigns (summer; 6 days; 15 hrs per day) were carried out in Bokamoso to monitor on-road vehicle PM_{10} concentration using a TSI DustTrak DRX® real-time optical aerosol counter. The Box Model method quantifies vehicle PM_{10} emission factors for heavy-duty, medium-duty, and motor vehicles. About 0.88 km of road within Bokamoso is paved with a daily traffic volume of >2000, and 3.6 km is unpaved with >250 daily traffic volume. Paved road heavy-duty vehicle non-exhaust PM_{10} emissions reported a positive medium to strong coefficient of determination to vehicle speed increase with a coefficient of determination of 0.59. Even though there is a positive coefficient of determination between heavy-duty vehicle non-exhaust PM_{10} emission factors and speed on unpaved roads, it is weak (0.2) due to low and less variable vehicle speed. Motor vehicle paved and unpaved road non-exhaust emission factors showed no significant coefficient of determination with increased vehicle speed. Paved road non-exhaust emission factors were not significantly variable with vehicle type. They ranged between an average of 0.15-0.18 g/km/h, with motor vehicles reporting an average of 0.18 g/km/h while heavy-duty vehicles reported an average of 0.15 g/km/h. On the contrary, unpaved road non-exhaust traffic PM_{10} emissions ranged between 0.16-0.3 g/km/h. The emission factors presented may be used to model vehicle traffic emissions, improve on-road vehicle dust emissions impact assessment and as a guide when deciding on local applicable road dust mitigation strategies.

Keywords

Vehicle dust emissions, mitigation strategies, PM_{10}, paved-unpaved roads

Introduction

Traffic re-suspended road dust comprises loose soil particles released by moving vehicle friction on paved or unpaved roads. Road emissions can be line sources when a single vehicle travels over a distance or continuous line sources when many closely packed vehicles are moving on the road (Alshetty & Shiva Nagendra, 2021). Jones (2000) estimated vehicle dust (PM_{10}) emission per vehicle from the unpaved road to be 0.534 kg/vehicle km, with varying chemical compositions depending on the road soil type. Peaks in traffic activity are between 05:00-09:00, 13:00-14:00 and 17:00 to 21:00 (Nkomo,2018). Traffic peaks result in hourly road dust emission episodes within low-income residential areas. A study characterising respirable indoor particulate matter in South African low-income settlements reported traces of road dust inside residents' homes (Language, 2020). Crustal soil was a major contributor to the total indoor respirable PM, accounting for 34-45% in Agincourt, 13-16% in Giyani, 32-55% in KwaZamokuhle, 9% in kwaDela, and about 25% in Joubertin (Language, 2020). PM from local sources such as construction work, vehicles re-suspended from unpaved roads and regionally transported aerosols were reported to contribute between 19 to 65% to the total ambient fine and coarse particulate matter concentrations (Muyemeki et al., 2021).

Transport has been identified as a fast-growing source of ambient pollutants in African countries, with an annual increase of 4% in on-road vehicle count. (OICA,2014). Due to increased transport demand, South Africa leads the continent with an annual vehicle count increase of 6k to 12k. A number of studies based on African literature report high ambient PM_{10} concentrations (80-360 $\mu g.m^{-3}$) attributed to traffic emissions (Terrouche et al., 2016; Zghaid et al., 2009; Lowenthal et al., 2014). Most traffic emissions published studies investigate exhaust emissions (Kirago et al. 2022; Ayetor et al. 2021). This is also true for South Africa; there is a vast body of literature reporting on traffic exhaust emissions (Tongwane et al., 2015; 2021; Mahlangu et al., 2020). However, there is limited literature on non-exhaust traffic emissions. This introduces uncertainties when quantifying the contribution of this source towards degraded ambient air quality and limits the development of non-exhaust traffic emission control strategies, standards and regulations.

In developed countries, non-exhaust vehicle emissions account for 60-90% of measured PM_{10} ambient concentrations (Gilliess et al., 2003). Jones (2000) estimated South African non-exhaust traffic emissions (PM_{10}) per vehicle from the unpaved road to be 0.534 $kg.km^{-1}$ with varying chemical compositions depending on the soil type of the road. However, due to international-based emission factors, these estimations do not entirely consider local traffic, road and meteorological characteristics. Worawat Songkitti et al., (2022) reported an increase of 25% in PM_{10} and $PM_{2.5}$ non-exhaust vehicle emissions when an electric vehicle payload was increased by 60–70 kg. Developing nations depend heavily on public transportation (buses and taxis) for long-distance travel and heavy-duty trucks for transporting goods; this finding is particularly significant.

Land use features, local meteorology (temperature, relative humidity, wind), road surface morphology characteristics (size, shape, pavement, speed humps), traffic flow, composition and vehicle speed have been reported to have an impact on vehicle road dust emissions hence the reported high spatial and temporal variability (Pachon et al., 2021). Vehicle road dust loading can range between 50mg/m^2 and 300 mg/m^2 on the same street or in different lanes of the same street (Gustaffson et al., 2019). This emphasises that several micro-scale factors govern non-exhaust PM traffic emissions and are highly variable over space and time. Considering this high variability, measuring non-exhaust emissions using local road, meteorological, and traffic characteristics is necessary to accurately estimate their contribution.

Therefore, this study aims to investigate the impact of vehicle weight and speed impact on non-exhaust traffic emissions and quantify PM_{10} emission factors per vehicle class using empirical field measurements. The emission factors presented in this article will reduce uncertainties associated with vehicle traffic emissions' contribution towards ambient air quality.

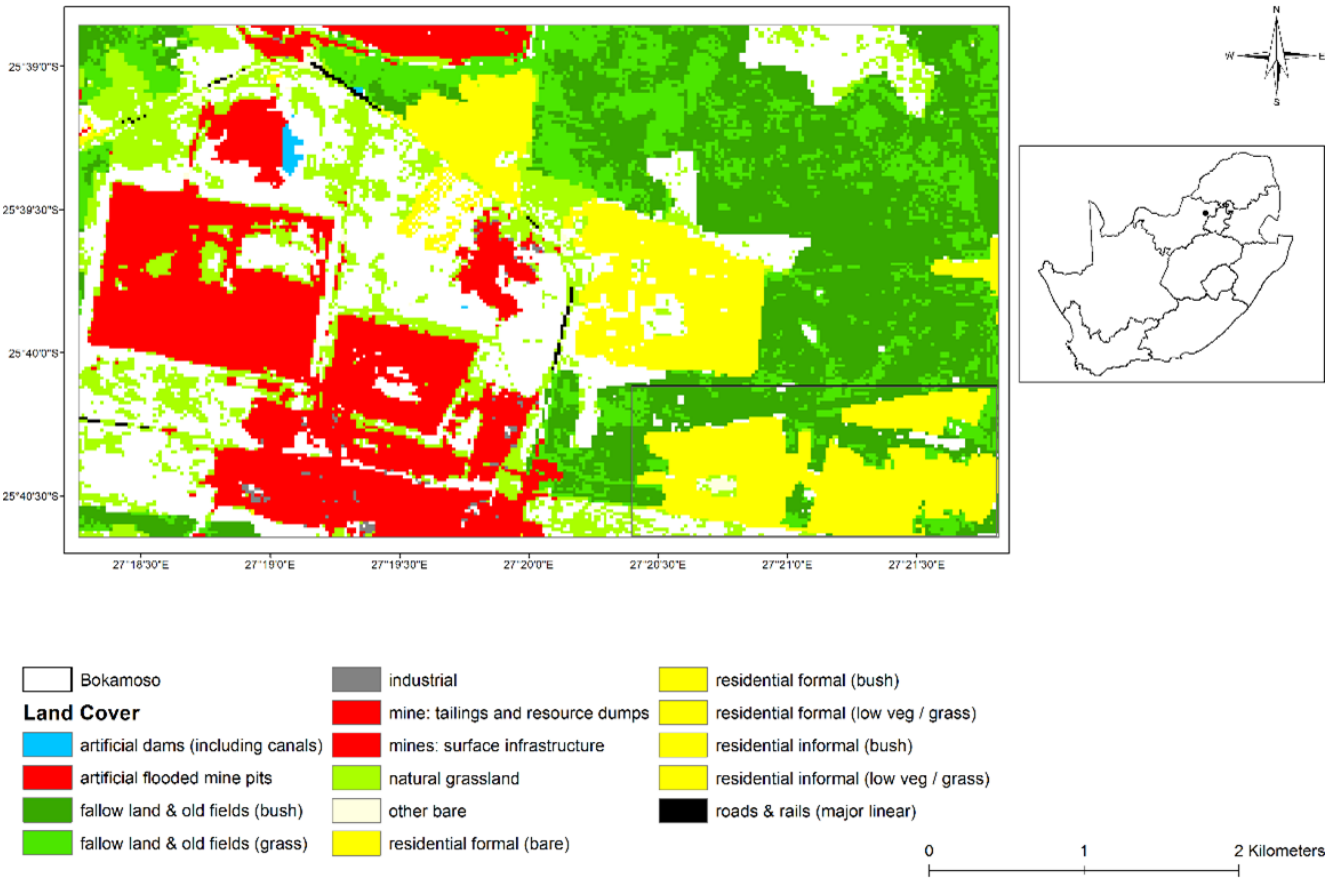


Figure 1: *Bokamoso low-income residential area location and land use characteristics.*

The reported emission factors add to the dust source emission inventory in South Africa and will reduce uncertainties associated with mobile dust sources and improve road dust particulate matter exposure estimates.

Methods

Study area description and methodology

The North West province is located in a semi-arid area and ranked third in South Africa regarding people living in poverty and lacking infrastructure (Stats SA, 2016). It has a paved road network of 26% and 74% unpaved roads (Department of Public Works and Roads, 2019). Bokamoso (25.6728 S, 27.3455 E) is a low-income residential area within the Rustenburg Municipality selected for the vehicle road dust monitoring campaign due to the presence of unpaved roads observed to be carrying low traffic volumes as well as paved roads observed to have higher traffic volumes. Bokamoso is one of the many low-income residential areas in South Africa; thus, it is important to understand the traffic patterns and the emissions associated with such a setting. The residential area has a population density of 2724 persons/km^2 (Stats SA General House Survey, 2022). It has been reported to have high levels of unemployment, with 18.7% of the residents having no income (Stats SA Census, 2022). About 30% of the people living in Bokamoso earn between R3 200 - R 6 366 monthly (Stats SA General House Survey, 2022). The residential area is characterised by unpaved roads, unpaved household yards, open soccer fields, and close-by mining and industrial activities. Bokamoso's land use characteristics and high population density (2724km^{-2}) drive the traffic activity within the residential area. The field road dust monitoring campaign took place during the summer.

The road dust monitoring campaign on-site meteorological station reported an average wind speed of 1.5 m.s^{-1} (Figure 2) with an ambient temperature of 25°C and humidity of 42% during the campaign period (Table 2).

Experimental setup and data collection methods

The field monitoring campaign aimed to characterise traffic and

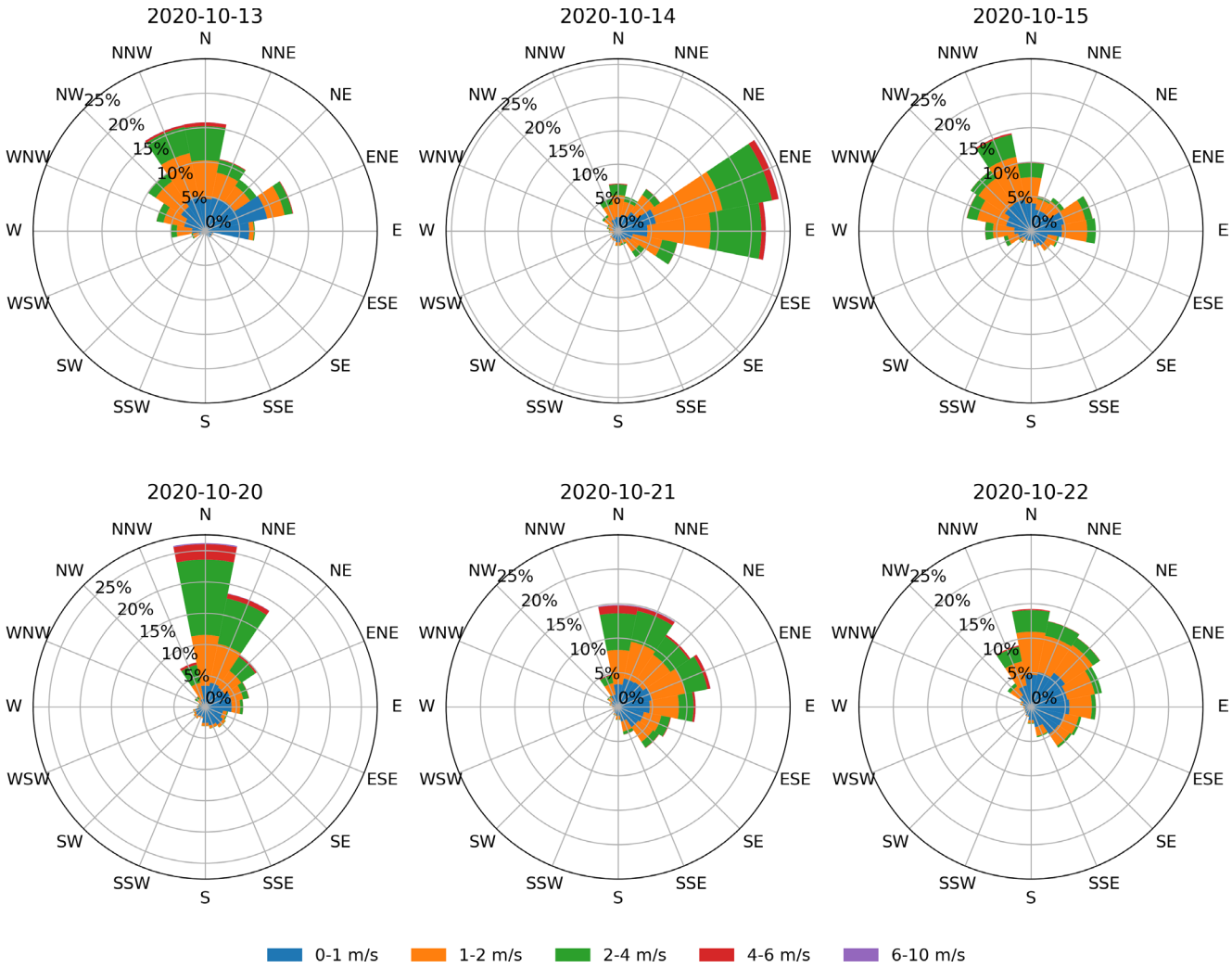


Figure 2: *Wind speed and direction for monitoring Bokamoso paved and unpaved road dust.*

Table 1: *Averaged daily meteorological conditions during the road dust monitoring campaign (On-site meteorological station)*

Road dust Monitoring campaign dates for October 2020	Paved road non-exhaust PM_{10} traffic emissions monitoring campaign days			Unpaved road non-exhaust PM_{10} traffic emissions monitoring campaign days		
	13	14	15	20	14	15
Wind Speed (m/s)	122	1.2	NO data	1.56	1.51	0.96
Temperature (°C)	21.51	25.7		31	26.9	26.96
Relative Humidity (%)	55.51	41.6		28	38	42
Rainfall (mm)	0	0		0	0	Drizzling

quantify non-exhaust PM_{10} emissions from paved and unpaved roads in Bokamoso took place from 13 October 2020 until 22 October 2020 from 05h:00-21h:00. Three days paved road site (Tuesday until Thursday) and three days unpaved (Tuesday-Thursday) road site. Particulate matter concentrations PM_{10} concentrations were monitored using a TSI DustTrak DRX® real-time optical aerosol counter. A meteorological station mounted with a rain gauge, temperature, humidity, and wind sensor was set up next to the aerosol counter to monitor meteorological conditions. A traffic counter (Sierzega SR4) captured vehicle data (type, speed, direction, and count). The experimental setup was positioned 2 m from the road edge.

A traffic counter was placed next to the weather station in order to monitor traffic activity and vehicle characteristics such as speed (km/h), vehicle length (decimeter), and direction (+-). The instrument uses the measured length to class the vehicles into categories. Vehicles with (dm) 0 - 20 are classified as motorcycles or bicycles, vehicles with lengths between 20 – 60 dm are classified as passenger cars, or (Motor Vehicles), vehicles with a length between 65 – 95 dm are classified as Medium Duty Vehicle (buses and taxis). Heavy-duty, multi-trailer trucks were assigned to vehicles with 95 – 255 dm lengths (Palo et al., 2019). This data determined the hourly composition on paved and unpaved roads. Also, determine the emission factor variation per vehicle class (Motorcycles, Motor vehicles, Medium duty and Heavy duty vehicles) as they differ in weight.

Data analysis methods

Bokamoso open streets, https://www.openstreetmap.org road shape file, was uploaded into GIS ArcMap and digitised to determine the proportion of paved and unpaved roads within the residential area. Descriptive statistics and clustering analysis was used to classify the vehicles, record their speed, and determine hourly traffic activity, traffic composition and sum on paved and unpaved road.

A simple box mathematical model was used to determine on-road PM_{10} emission factors as a function of vehicle type, weight, and speed. The model is based on a mass-balanced principle, as in Equation 1 (Font et al., 2014). The source, in this case, the road "box", has a (width), L (length) and H (height).

$$q = \frac{\mathrm{uh}(c-b)}{L} \quad (1)$$

Where q= emission factor ($mg.m^{-2}s^{-1}$); u=wind speed ($m.s^{-1}$); h= height from the ground where the measurements are taken (m); c= PM concentration during peak periods ($mg.m^{-3}$) b=PM concentration during non-peak periods ($mg.m^{-3}$) L= length of the road segment for the sampling point (m).

The model assumes that (i) the particulate matter emission rate from the source and airflow out the "box" is constant and in equilibrium, (ii) the concentration of the pollutant in the atmosphere is well mixed, and the airflow within the box is uniform (ii) other atmospheric processes that lead to secondary formation of particles and deposition by gravitation are suspended and insignificant (Font et al., 2014). Specific case studies from the vehicle counter and DustTrak PM concentration and meteorological datasets were selected to determine the impact of vehicle mass on the PM emission factors.

Results

The road network in Bokamoso

Bokamoso has 81% (3.7km) of the unpaved road network, and 19 (0.9km) is paved. The paved roads are covered with asphalt with an even width of about 6 meters, starting at the residential area entrance with a double-line road network that runs through the residential area towards the exit. The residential areas' unpaved roads are bare soil rough surfaces with multiple road networks varying widths of 4-8 meters, highly connected and ultimately connected to the paved road.

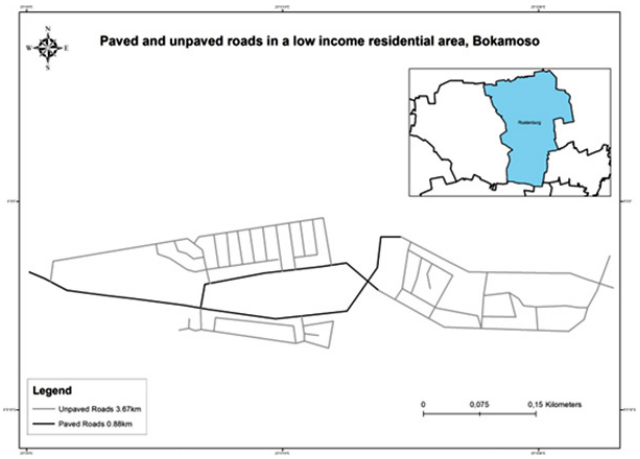


Figure 3: *Bokamoso paved and unpaved road network*

Traffic characteristics within Bokamoso

Bokamoso paved roads reported an average of 2 119 vehicles per day and a total of 6 357 in three days. In contrast, unpaved

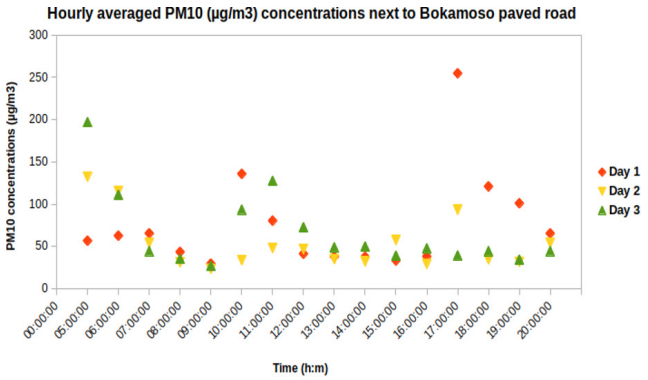


Figure 4A: *Paved roadside PM_{10} concentration measurementsnetwork*

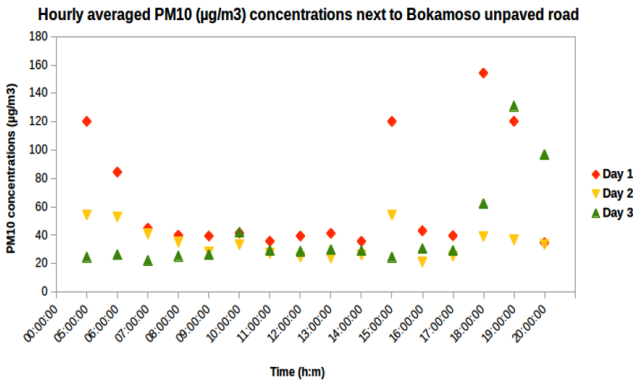


Figure 5A: *Unpaved roadside PM_{10} concentration measurements*

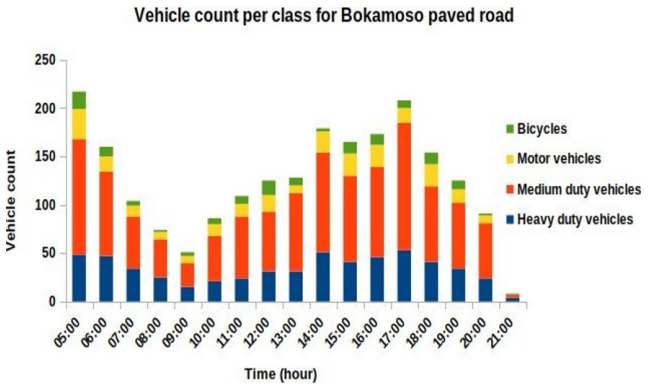


Figure 4B: *Paved road day 1 (Tuesday) diurnal traffic composition*

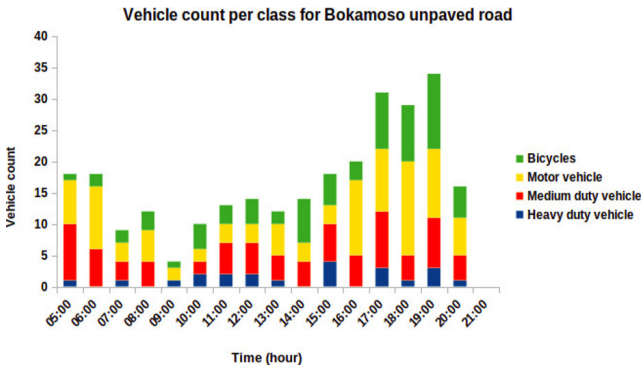


Figure 5B: *Unpaved road day 1(Tuesday) diurnal traffic composition*

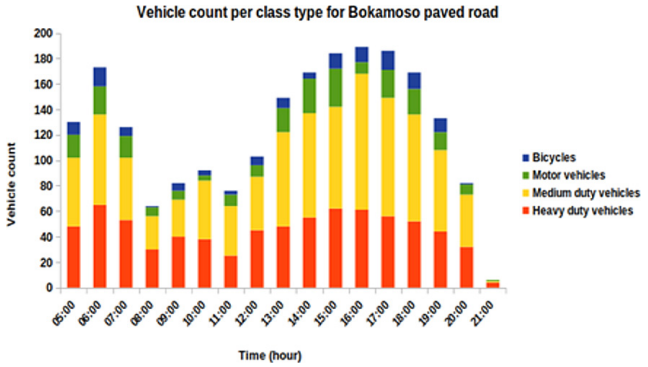


Figure 4C: *Paved road day 2 (Wednesday) diurnal traffic composition*

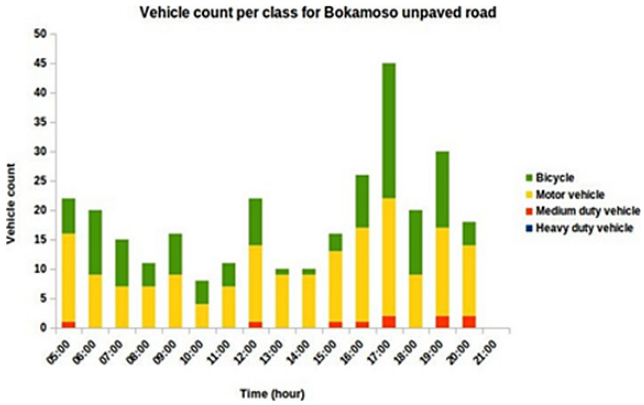


Figure 5C: *Unpaved road day 2 (Wednesday) diurnal traffic composition*

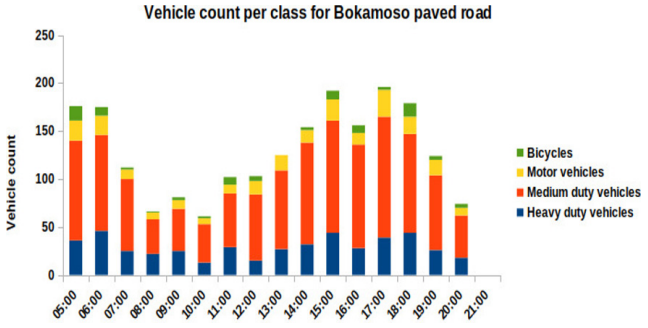


Figure 4D: *Paved road day 3 (Thursday) diurnal traffic composition*

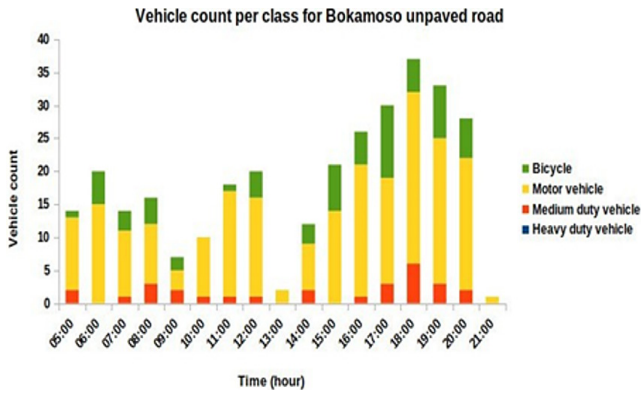


Figure 5D: *Unpaved road day 3 (Thursday) diurnal traffic composition*

roads have an average traffic volume of 298 per day and a total of 896 in 3 days. Medium-duty commercial and passenger vehicles dominate unpaved roads, while medium and heavy-duty vehicles dominate paved roads. The PM_{10} and traffic count graphs (Figures 4 and 5) show a strong coefficient of determination between traffic density per hour and concentration measured. The paved road monitoring point shows higher concentrations up to averaged 200 $\mu g.m^{-3}$ with peak concentrations between 05h00-07h00: 09h00-11h00 and 13h00-20h00 (Figure 4A-D). Unpaved road monitoring concentration peaks are up to averaged 160 $\mu g.m^{-3}$ with peak hours from 06h00-08h00, 10h00-13h00 and 15h30-19h30 (Figure 5A-D). Unpaved roads show lower concentrations compared with paved as multiple unpaved roads (3.7 km) networks feed traffic to the double-line paved roads (0.9 km).

Impact of vehicle speed on the paved and unpaved road per vehicle type

Paved and unpaved heavy-duty vehicle emission factors correlate positively with vehicle speed, Figure 6. The speed for unpaved roads ranged between 10-30 $km.h^{-1}$, mainly due to the national set vehicle speed for heavy-duty vehicles on unpaved roads within residential areas. Therefore, although there is a positive coefficient of determination between heavy-duty vehicle non-exhaust PM_{10} emission factors and speed on unpaved roads, it is weak (0.2) due to low and less variable vehicle speed. Unpaved road emission factors ranged from 0.05-0.2 g/km/h. Paved road heavy-duty vehicles' speed had a wider range of 30-70 $km.h^{-1}$ compared to unpaved road heavy-duty vehicles. However, they reported similar emission factors (0.05-0.3 g/km/h). Paved road heavy-duty vehicle non-exhaust PM_{10} emissions reported a positive medium to strong coefficient of determination to vehicle speed increase with a coefficient of determination of 0.59.

Medium-duty vehicle PM_{10} emissions for paved roads showed a weak linear coefficient of determination (0.2) with vehicle speed, while unpaved road emissions showed a positive, strong linear coefficient of determination (0.89) with vehicle speed (Figure 7A). Overall, paved road medium-duty vehicles reported higher PM_{10} emission factors (0.05-0.325 g/km/h) than unpaved road medium-duty vehicles. This can be explained by the overall high-speed range for paved (24-61 $km.h^{-1}$) road medium-duty vehicles compared to unpaved road medium-duty vehicles (8-20 $km.h^{-1}$).

Motor vehicle paved road non-exhaust emission factors showed no significant coefficient of determination with increased vehicle speed (Figure 8). Most of the emission factors were below 0.4 g/km/h with speeds ranging between 10-60 $km.h^{-1}$. Unpaved road motor vehicle emission also showed no significant coefficient of determination (0.2) with vehicle speed, with most emission factors below 0.25 g/km/h with speeds ranging between 8-22 $km.h^{-1}$. This might mean further regulating or reducing motor vehicle speed for unpaved and paved roads may not significantly reduce non-exhaust motor vehicle PM_{10} emissions.

Variability of vehicle PM_{10} emission factors per vehicle type on paved and unpaved roads

PM_{10} emission factors for five vehicles per vehicle class were calculated using the box model to investigate paved and unpaved road PM_{10} emissions as a function of vehicle weight for the different vehicle categories. Figure 9 shows a variation of non-exhaust traffic PM_{10} emissions per vehicle type on paved and unpaved road surfaces. Paved road non-exhaust emission factors were not significantly variable. They ranged between an average of 0.15-0.18 g km/h, with motor vehicles reporting an average of 0.18 g/km/h while heavy-duty vehicles reported an average of 0.15 g/km/h. On the contrary, unpaved road non-exhaust traffic PM_{10} emissions ranged between 0.16-0.3 g/km/h. On unpaved roads, medium-duty vehicles reported the lowest average (0.16 g/km/h), followed by motor vehicles (0.18 g/km/h), while heavy-duty vehicles reported the highest (0.3 g/km/h) average non-exhaust PM_{10} emission factor.

The emission factors have a limitation of being a small sample size. Only seven case studies per vehicle type were used to derive the averaged emission factors as this was a real-world field monitoring campaign, and other variables that significantly impacted the emissions, like ambient wind speed and vehicle speed, were difficult to control. Unpaved road monitoring sites had higher emission factors but lower traffic density as there are multiple road networks within the residential area. Paved roads had lower emission factors but higher traffic density as all unpaved road networks within the residential area feed traffic to paved roads. Also, the box model method tends to underestimate emissions calculated when the wind speed is extremely low or high (Haustein et al., 2015).

Discussion

About 80% of Low-income residential area's road network is unpaved and highly connected, thus containing many cross-section braking zones (T junctions, four-way stops) (Gwilliam et al., 2008). This is true for the Bokamoso residential area's road network, as seen in Figure 3. Low-income residential unpaved roads have no flood control infrastructure, poor drainage systems and lack traffic management infrastructures (Nkomo, 2016). They also have a high deterioration rate with a lifespan of less than one year (Saha and Ksaibati, 2017). This decreases overall air quality in surrounding areas as unpaved roads and sidewalks are an infinite source of loose soil particles, especially during non-rainy seasons.

It has been reported that traffic counts have a low coefficient of determination (0.55-0.65) with non-exhaust PM_{10} emissions factors since traffic-heavy paved roads are not effective dust reservoirs (Schaap et al., 2009; Amato et al., 2013; Abu-Allaban et al., 2003; Etyeme-zian et al., 2003). In addition, Padoann et al., (2017) reported higher PM_{10} emissions from secondary paved roads with fewer than 1000 vehicles per day compared with primary roads with over 2000 vehicles per day. These results

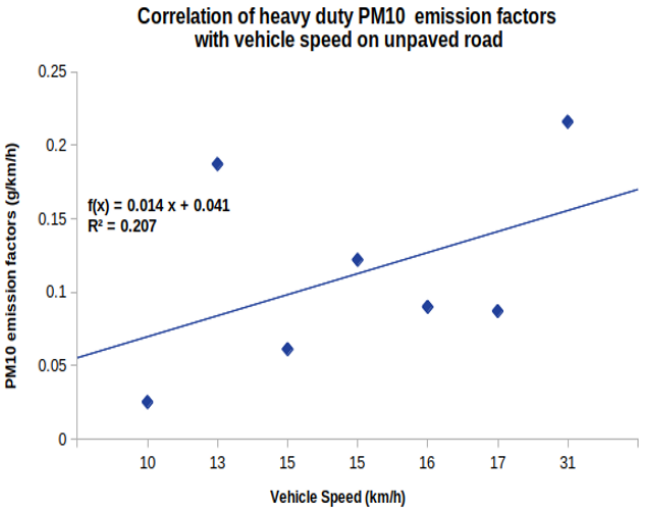


Figure 6A: *Coefficient of determination between vehicle speed and PM_{10} emissions for heavy-duty vehicles on unpaved roads*

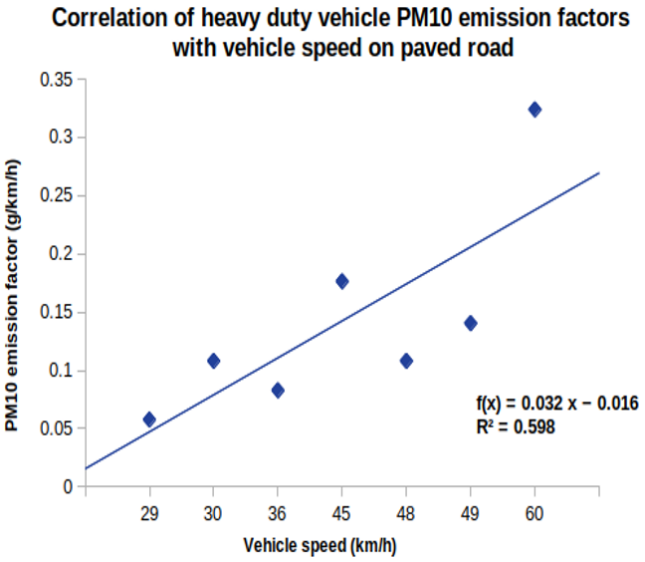


Figure 6B: *Coefficient of determination between vehicle speed and PM_{10} emissions for heavy-duty vehicles on paved roads*

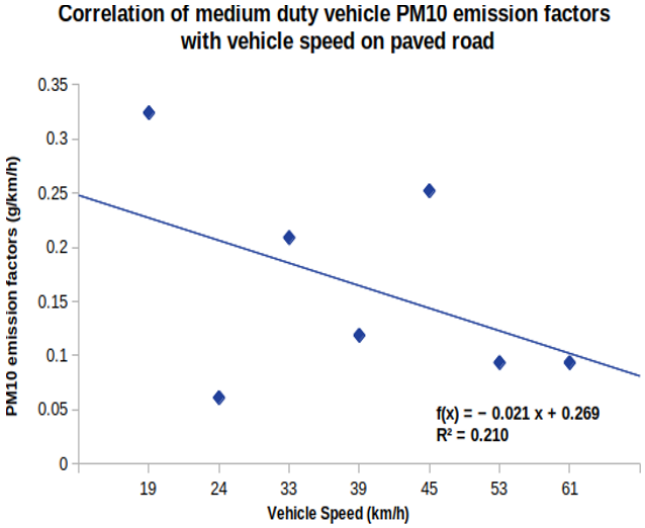


Figure 7A: *Coefficient of determination between vehicle speed and PM_{10} emissions for medium-duty vehicles on paved roads*

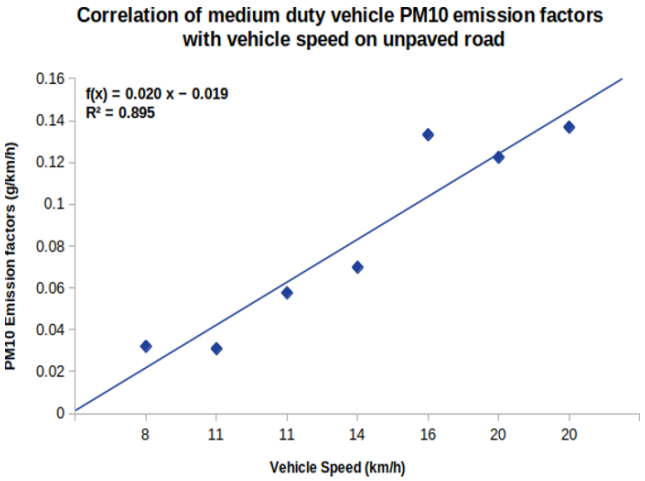


Figure 7B: *Coefficient of determination between vehicle speed and PM_{10} emissions for medium-duty vehicles on unpaved roads*

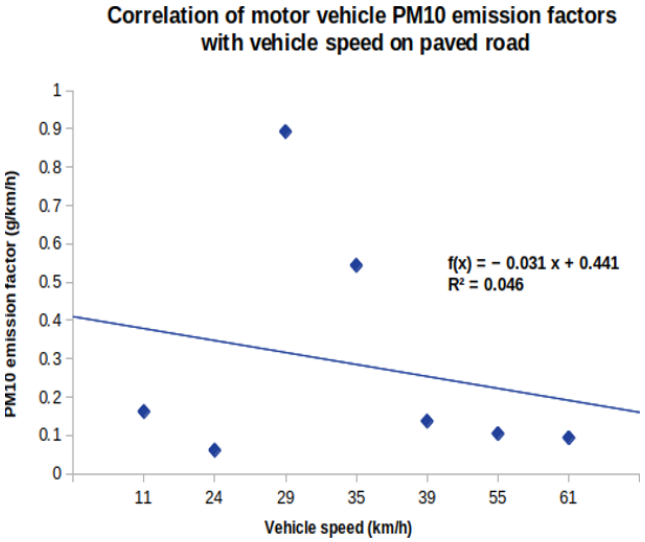


Figure 8A: *Coefficient of determination between vehicle speed and PM_{10} emissions for Motor vehicles on paved roads*

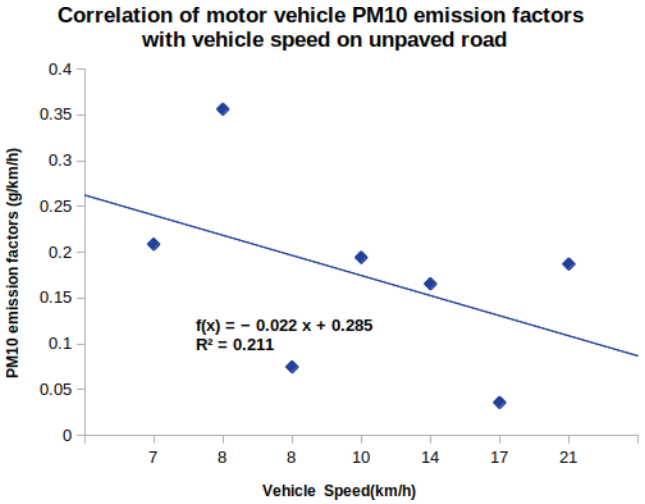


Figure 8B: *Coefficient of determination between vehicle speed and PM_{10} emissions for Motor vehicles on unpaved roads*

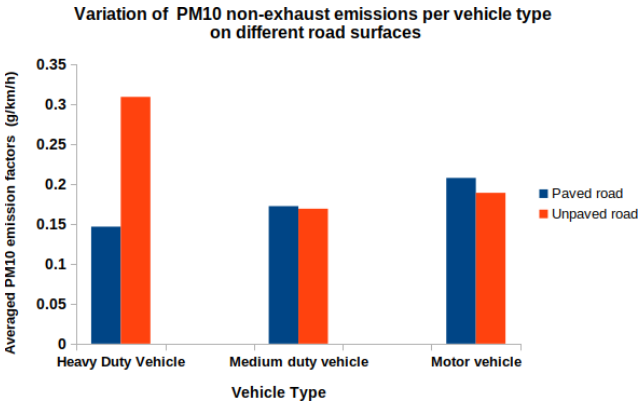


Figure 9: *The impact of vehicle weight on paved and unpaved road PM_{10} emission factors*

may not apply to South African low-income paved residential roads as unpaved sidewalks, household yards, and recreation areas continuously supply loose soil particles to adjacent paved roads. This explains the higher PM_{10} concentrations (0.5-1.5 $mg.m^{-3}$) on the site of the paved road when compared to the unpaved road (0.2-0.6 $mg.m^{-3}$) in Bokomo's site.

Local meteorological factors such as ambient wind speed and relative humidity affect non-exhaust traffic emissions. Panchon et al., (2021) investigated seasonal impact on road dust loading for 41 sites within Bagota. The results showed doubled dust loading measurements in the dry season (1.8-45 $mg.m^{-2}$) compared to the wet season (1-22 $mg.m^{-2}$). This is because soil moisture decreases road surface dust emission strength and loose soil particle availability (Munkhtsetseg et al., 2016). The results presented in this paper are from summer road dust monitoring field campaigns; hence, they may be applicable only in summer, and non-exhaust traffic emission estimates for paved and unpaved roads in winter are expected to be higher.

About 63% of South African low-income residents use buses and taxis (Nkomo et al. 2016;2018). The North West province's public transport composition in 2003 was mainly taxis (64%), buses (30) and trains (6%); these statistics increased in 2013 to 67% (taxis) and 21% for buses (National Household Transport Survey, 2003 and 2013). Residents working close to the residential areas use bicycles or private staff public transport (motor vehicles) to work or nearby destinations. According to the National Household Transport Survey, 2013, about 42.4% of low-income South African residents walk to workplaces, 38% use public transport (buses, taxis and trains), while 18.8% use motor vehicles. This correlates well with Bokomo's unpaved road traffic composition as there is a substantial amount of bicycles, especially on unpaved roads. The medium and heavy-duty vehicles on the unpaved road are mostly taxis and buses (public transport); on the paved roads, they are mostly trucks and heavy-duty trucks transporting goods to nearby areas.

There is a portion of low-income residents that prefer private staff transport (motor vehicles) due to public (taxis) transport, long departure waiting periods, and poor road infrastructure that decreases riding quality (Feikie et al. 2018). This could explain the significant count in motor vehicle class compositions as seen in Bokamoso's residential area paved and unpaved road traffic diurnal cycles. The recent (2021) South African state of traffic report showed a vehicle increase count of 2.28 % (13,712) for the North West province from the year 2019-2021. However, at the national scale, low-income residents (quintile 1) only own 6% of the total registered motor vehicles in South Africa (National Household Transport Survey, 2019).

Kuhns et al. (2003) stated that an increased vehicle weight and speed increases the amount of non-exhaust traffic PM_{10} emitted. This is because an increase in vehicle speed increases ambient wind turbulence between vehicle tires and, therefore, may increase non-exhaust traffic emissions on unpaved roads or paved roads with loose soil particles. Hussein et al. (2008) supported this and reported high dust (10 times greater) concentration measurements for a vehicle travelling at 100 $km.h^{-1}$ compared to a vehicle travelling at 20 $km.h^{-1}$. (Amato et al., 2017) identified that when speed had increased from 80 to 110 $km.h^{-1}$, the mean PM concentration was higher with increased vehicle speed. Similar findings are reported for Bokamoso's unpaved roads. A positive coefficient of determination of 0.86 and 0.47 between vehicle speed and PM_{10} emission factors was reported for medium and heavy-duty vehicles, respectively. Both these vehicles showed a positive coefficient of determination on unpaved roads, except for motor vehicles. This difference in the coefficient of determination relationship could be because of the average speed limits and weight differences. Motor and medium-duty vehicle speed limit on unpaved roads ranged between 8-21 km/h. The speed limit for heavy-duty vehicles ranged from 10-32km/h. Motor and medium-duty vehicles showed a low coefficient of determination (0.01;0.19) between vehicle speed and calculated PM_{10} emissions for paved roads. However, there was a positive coefficient of determination for heavy-duty vehicles on paved roads (0.63).

Vehicle type, weight and length mostly determine the vehicle wheel size, count and type. Vehicle wheel type varies per vehicle class, while vehicle wheel count and size tend to increase from lowest vehicle (Motorcycle) class to highest (heavy duty vehicle) (Wenzel,2010). Gillies et al., (2005), found a variation in non-exhaust PM emission factors per vehicle type, with heavy-duty vehicles having higher (48 $g\ km^{-1}h^{-1}$) PM_{10} emission factors when compared with light-duty vehicles (0.8 $g\ km^{-1}h^{-1}$). Other studies (Schaap et al., 2008 : Amato, 2012) found that heavy-duty vehicles' non-exhaust PM_{10} emissions were up to 9 times higher when compared with motor vehicles, which showed a low coefficient of determination with an almost negative coefficient of determination line.

The study has limitations, several other sources of PM in residential areas, such as domestic burning, and yard sweeping, may lead to an overestimation of the traffic PM_{10} concentration recorded by the aerosol counter. The study was conducted during the COVID-19 lockdown period; therefore, the traffic flow or counts at 05h00 am and 21h00 may be overestimated due to

the lockdown curfew times, which were set to be 04h00- 22h00. Other vehicles use a different road or decrease speed when they see the instrument set up. Therefore, this may result in over and under estimation of the calculated emission factors and may not be applicable under normal traffic conditions. The aerosol counter (DustTrak) has high precision; however, it has been reported to underestimate ambient PM_{10} coarse concentration by 20% and overestimate PM fine fraction by a factor of 2 (Javed and Guo., 2021). In the case of traffic, the traffic counter (Sierzega SR4) could experience vehicle counting and classification errors (Kijewska and Iwan, 2019).

Conclusion

More than 80 % of the roads in Bokamoso low-income residential areas are unpaved and are a daily source of particulate matter, with peak emissions reported during traffic rush hours. The residential area has a high traffic activity driven by the area's land use characteristics. Bokamoso paved roads reported an average of 2 119 vehicles per day and a total of 6 357 in three days. In contrast, unpaved roads have an average traffic volume of 298 per day and a total of 896 in three days. Paved and unpaved heavy-duty vehicle emission factors correlate positively with vehicle speed. Paved road heavy-duty vehicle non-exhaust PM_{10} emissions reported a positive medium to strong coefficient of determination to vehicle speed increase with a coefficient of determination of 0.59. Even though there is a positive coefficient of determination between heavy-duty vehicle non-exhaust PM_{10} emission factors and speed on unpaved roads, it is weak (0.2) due to low and less variable vehicle speed. Motor vehicle paved and unpaved road non-exhaust emission factors showed no significant coefficient of determination with increased vehicle speed. Paved road non-exhaust emission factors were not significantly variable with vehicle type. They ranged between an average of 0.15-0.18 g/km/h, with motor vehicles reporting an average of 0.18 g/km/h while heavy-duty vehicles reported an average of 0.15 g/km/h. On the contrary, unpaved road non-exhaust traffic PM_{10} emissions ranged between 0.16-0.3 g/km/h. The study results will inform decisions regarding road dust emissions interventions in South Africa. The authors recommend a laboratory-based study to further investigate the impact of vehicle characteristics on road dust emissions.

Acknowledgements

The authors would like to thank the Climatology Research Group and NOVA-Institute for assisting with data collection. We are also grateful to Bokamoso councillors and community leaders who supported the campaign.

Note

An earlier version of this paper was presented at the National Association of Clean Air (NACA) Conference in October 2022 and was published in its Proceedings.

References

Abu-Allaban M, Gillies J A , Gertler A W, Clayton R, Proffitt D (2003) Tailpipe, re-suspended road dust, and brake-wear emission factors from on-road vehicles. *Atmospheric Environment* (Oxford, England: 1994), 37(37): 5283–5293. https://doi.org/10.1016/j.atmosenv.2003.05.005

Alshetty D, Shiva Nagendra M S, (2021) Urban characteristics and its influence on resuspension of road dust, air quality and exposure. Air Quality, Atmosphere & Health 15(2): 273-287.

Ayetor G K, Mbonigaba I, Ampofo J, Sunnu A (2021) Investigating the state of road vehicle emissions in Africa: A case study of Ghana and Rwanda. *Transportation Research Interdisciplinary Perspectives*, 11(100409): 100409. https://doi.org/10.1016/j.trip.2021.100409

Department of transport, state of traffic report, (2021) https://www.gov.za/sites/default/files/gcis_document/202201/transport-annual-report-2020/21.pdf, accessed 24-08-2023.

Feikie X E , Das D K, Mostafa Hassan M, (2018) Perceptions of the factors causing traffic congestion and plausible measures to alleviate the challenge in Bloemfontein, South Africa. 37th Annual Southern African Transport Conference (SATC 2018) Proceedings, Pretoria, South Africa. 9-12 July 2018.

Font A, Baker T, Mudway I, Purdie E, Dunster C, Fuller, G, (2014) Degradation in urban air quality from construction activity and increased traffic arising from a road-widening scheme. Science of the Total Environment, 497(498):123–132.

Gillies J A, Etyemezian V, Kuhns H, Nikolic D, Gillette D A, (2005) Effect of vehicle characteristics on unpaved road dust emissions. Atmospheric Environment, 39(13): 2341–2347. https://doi.org/10.1016/j.atmosenv

Gustafsson, M., Blomqvist, G., Järlskog, I., Lundberg, J., Janhäll, S., Elmgren, M., Johansson, C., Norman, M. and Silvergren, S., (2019). Road dust load dynamics and influencing factors for six winter seasons in Stockholm, Sweden. Atmospheric Environment: X, 2, p.100014.

Gwilliam K., Foster V., Archondo-Callao R., Briceño-Garmendia C., Nogales A., Sethi K., (2008) The Burden of Maintenance: Roads in Sub-Saharan Africa, Background Paper 14 (Phase I) 1-58, https://roadsforwater.org/wp-content/uploads/2013/10/the-burden-of-maintenance_roads-in-SSA.pdf, accessed 20-08-2023.

Haustein K, Washington R, King J, Wiggs G, Thomas D, S G, Eckardt, F D, Bryant R G, Menut L, (2015) Testing the performance of state-of-the-art dust emission schemes using DO4 Models field data. Geoscientific Model Development, 8(2):341–362. https://doi.org/10.5194/gmd-8-341-2015.

Jones D, (2000) Road dust: just a nuisance or a significant road management issue. Pretoria, South Africa Transport Conference, Pretoria, 17-20 July:1-13.

Kijewska K, Małecki K, Iwan S (2016) Analysis of data needs and having for the integrated urban freight transport management system. In *Communications in Computer and Information Science* (pp. 135–148). Springer International Publishing.

Kirago L, Gatari M J, Gustafsson Ö, Andersson A, (2022) Black carbon emissions from traffic contribute substantially to air pollution in Nairobi, Kenya. Communications Earth & Environment, 3(1). https://doi.org/10.1038/s43247-022-00400-1

Kuhns H, Gillies J, Watson J, (nd.) Vehicle-Based Road Dust Emissions Measurements. http://www3.epa.gov/ttn/chief/conference/ei12/fugdust/huhns.pdf accessed 06-06-2021.

Language B P (2020) Characterisation of Respirable Indoor Particulate Matter in South African Low-Income Settlements. North-West University, PhD dissertation.

Lowenthal D H, Gertler A W, Labib M W, (2014) Particulate matter source apportionment in Cairo: recent measurements and comparison with previous studies. *International Journal of Environmental Science and Technology: IJEST*, 11(3): 657–670. https://doi.org/10.1007/s13762-013-0272-6

Mahlangu A, Schaberg P, Wattrus M, Forbes P, (2020) Characterisation of semi-volatile hydrocarbon emissions from diesel engines. *Clean Air Journal*, 30(1). https://doi.org/10.17159/caj/2020/30/1.7672

Muyemeki L, Burger R, Piketh S J, Language B, Beukes J P, Van Zyl PG, (2021) Source apportionment of ambient $PM_{10-2.5}$ and $PM_{2.5}$ for the Vaal Triangle, South Africa. *South African Journal of Science*, 117 (5/6): 1–1.

National Household Travel Survey, 2003. file:///home/lapss46/Downloads/ddi-documentation-english-569.pdf accessed 23-08-2023

National Household Travel Survey, 2013. https://www.statssa.gov.za/publications/P0320/P03202013.pdf accessed 23-08- 2023.

National Household Travel Survey, 2020. https://www.statssa.gov.za/publications/P0320/P03202020.pdf accessed 23-08- 2023.

Ncube K, Shackleton C M, Swallow B M, Dassanayake W, (2016) Impacts of HIV/AIDS on food consumption and wild food use in rural South Africa. Food Security, 8: 1135-1151.

Nicholson K W, Branson J R, (1990) Factors affecting resuspension by road traffic. Science of the Total Environment, 93:349–358.

Nkomo S. L, Desai S, Peerbhay K, (2016) Assessing the Conditions of Rural Road Networks in South Africa Using Visual Observations and Field-Based Manual Measurements. Review of Social Sciences, 1(2): 42-55.

Nkomo SL (2018) Assessing the condition of unpaved rural road networks and the associated impacts on the livelihoods of rural communities: A case study of four rural communities in KwaZulu, South Africa. University of KwaZulu-Natal (Thesis-PhD).

Pachon, J., Vanegas, S., Saavedra, C., Amato, F., Silva, L., Blanco, K., Chaparro, R. and Casas, O., (2021). Evaluation of factors influencing road dust loadings in a Latin American urban center. Journal of the Air & Waste Management Association, 71(2), pp.268-280.

Paľo J, Caban J, Kiktová M, Černický Ľ, (2019) The comparison of automatic traffic counting and manual traffic counting. *IOP Conference Series. Materials Science and Engineering*, 710(1): 012041. https://doi.org/10.1088/1757-899x/710/1/012041

Saha P, Ksaibati K, (2017) Developing an optimisation model to manage unpaved roads. J*ournal of Advanced Transportation*, 2017, 1–11. https://doi.org/10.1155/2017/9474838

Schachtebeck C, Mbuya J M, (2016) Assessing the potential benefits of road infrastructure development for poverty alleviation: Lessons Learnt from developing Economies. Southern African Transport Conference, Pretoria, .88-98.

Sehmel G.A. (1973) Particle resuspension from an asphalt road caused by car and truck traffic. Atmospheric Environment, 7:291–309.

Songkitti W, Sa-ard-iam S, Plengsa-Ard C, Wirojaskunchai E, (2022) Effects of payloads on non-exhaust PM emissions from A hybrid electric vehicle during A braking sequence. *Aerosol and Air Quality Research*, 22(7): 220150. https://doi.org/10.4209/aaqr.220150

Statistics South Africa, (2016) General Household water and sanitation: an in-depth analysis of the general Household Survey 2002-2015. Pretoria, South Africa.

Statistics South Africa, (2022) General Household water and sanitation: an in-depth analysis of the general Household Survey 2020-2022. Pretoria, South Africa.

Terrouche A , Ali-Khodja H , Kemmouche A , Bouziane M , Derradji A , Charron A, (2016) Identification of sources of atmospheric particulate matter and trace metals in Constantine, Algeria. *Air Quality, Atmosphere, & Health*, 9(1): 69–82. https://doi.org/10.1007/s11869-014-0308-1

Tongwane M, Piketh S, Stevens L, Ramotubei T, (2015) Greenhouse gas emissions from road transport in South Africa and Lesotho between 2000 and 2009. *Transportation Research. Part D, Transport and Environment*, (37) 1–13. https://doi.org/10.1016/j.trd.2015.02.017

Zghaid M, Noack Y, Bounakla M, Benyaich F. (2009) Pollution atmosphérique particulaire dans la ville de Kenitra (Maroc). *Pollution Atmospherique*, N°203. https://doi.org/10.4267/pollution-atmospherique.1184

ABOUT US

skyside
Creating a brighter tomorrow

At Skyside, we're creating a brighter tomorrow by scientifically establishing the safety of our air, water and soil.

We track environmental improvement.
Our laboratories and services are internationally accredited to ISO17025 and have been recognised by the National Association for Clean Air (NACA) as the leaders in air quality.

Let's work together to improve our world.

OUR PRIMARY SERVICES

AMBIENT AIR QUALITY TESTING

STACK EMISSIONS TESTING

ODOUR MONITORING eNOSE

NOISE MONITORING

COMPRESSED AIR TESTING

WATER TESTING

EQUIPMENT & SOFTWARE

EMERGENCY RESPONSE

VALUE ADDED SERVICES

Skyside South Africa is continuously developing partnerships with cutting-edge product developers from around the world to allow you to gather air monitoring data empowering you to make better decisions daily.

OHIO LUMEX
Skyside South Africa is a distributor of Ohio Lumex Products. Ohio Lumex is a worldwide leader in environmental monitoring and consulting.

RESPIRER URBAN SERVICES
Skyside supplies the ATMOS Particulate Monitor; a low-cost air quality monitoring device that provides real-time air quality data.

COMON INVENT eNOSE
Skyside supplies e-Nose odour monitoring devices. These are compact instruments that detect odour changes in the air quality that may be hazardous.

POWER BI
Skyside offer a data management and reporting system for dust fallout monitoring which helps you read data, discover insights, and forecast trends across a wide range of datasets.

ACCREDITATION

Skyside is proud of our distinction of being the first South African emissions and dustfall testing company to achieve ISO 17025 accreditation for our most commonly applied methods.

National Association For Clean Air

A member and sponsor of the National Association of Clean Air and recognised in 2016 as leaders in air quality, Skyside is at the forefront of stack emissions testing and air quality data.

Research article

Microstructure and chemical composition of deposited particulate matter from gasoline and diesel vehicle exhaust emissions

Abdelkader Lemou[1,2,*], **Nabila Ait Ouakli**[1], **Amina Djadi**[1], **Lyes Rabhi**[1,2], **Nabila Cherifi**[1], **Riad Ladji**[1] and **Noureddine Yassaa**[2,3]

[1]Unité de Recherche en Analyses Physico-Chimiques des Matrices Fluides et Sols-Centre de Recherche Scientifiques et Techniques en Analyses Physico-chimiques URAPC-MFS - CRAPC, BP 384, Zone Industrielle Bou-Ismail RP 42004 Tipaza, Algérie
[2]Laboratoire d'Analyse Organique Fonctionnelle, Faculté de Chimie, Université des Sciences et de la Technologie Houari Boumediene, BP 32, El-Alia, Bab-Ezzouar, 16111 Alger, Algérie
[3]Commissariat aux Energies Renouvelables et à l'Efficacité Energétique, CEREFE, 23 Rue Docteur Slimane Asselah, Telemly, Algiers, Algeria
*Corresponding author: madjid.des@hotmail.com

Received: 15 December 2022 - **Reviewed:** 6 July 2023 - **Accepted:** 2 November 2023
https://doi.org/10.17159/caj/2023/33/2.15341

Abstract

A comprehensive physicochemical characterization of transport-emitted aerosols containing in vehicle exhaust emissions derived from the combustion of fuels such as diesel, premium gasoline, and unleaded gasoline was performed in this study by employing a range of analytical techniques (Laser granulometry, X-ray Fluorescence (XRF), Fourier Transform Infrared (FTIR), X-ray diffraction (DRX), Scanning Electron Microscopy (SEM), and Thermogravimetry). The X-ray diffractogram of diesel (D) derived aerosols showed an amorphous structure while those of unleaded gasoline (UG) and premium gasoline (PG), showed amorphous crystalline phases. The chemical composition of D, PG and UG derived particles was dominated by aliphatic C-H groups of alkanes with relatively low C=O groups of carboxylic acids, ketones, aldehydes, esters, lactones, and sulphate (SO_4^{2-}) inorganic salts. The nitrogen-containing functionality (NO_3-) was specific to particles of PG and UG. Laser particle size analysis showed fine particle sizes (Range) generated from diesel exhausts, thus making them dangerous when inhaled, as they can penetrate deeply into the human airways and become incorporated into the blood stream damaging more other viscera.

Keywords

Air pollution, combustion particles, physicochemical characterization, traffic emissions, environmental toxicity, exhaust gas suspension

Abbreviations

Diesel: D, unleaded gasoline: UG, premium gasoline: PG, FTIR: Fourier transform infrared spectroscopy; XRF: X-ray fluorescence; DRX: X-ray diffraction; SEM/EDX: Scanning Electron Microscopy/energy-dispersive X-ray analysis; TGA: Thermogravimetric analysis; DPM: Deposited particulate matter; PM: Particulate Matter.

Introduction

Suspended particulate matter (PM) is one of the major air pollutants affecting human health (Mico et al., 2015; Chernyshev et al., 2019; Lemou et al., 2020, Valavanidis et al., 2008, Popovicheva et al., 2014). The health effects of air pollution, observed in both indoor and outdoor environments, are of great concern because of the high risk of exposure even to relatively low concentrations of air pollutants (Rabhi et al., 2018). It is estimated that over 7 million deaths occur worldwide each year because of exposure to air pollution affecting the lungs and respiratory system (Lemou et al., 2020; Alves et al., 2023). The size, concentrations and compositions of the particles in the air, which can penetrate deep into the lungs and mix with the bloodstream have health implications (Yusuf et al., 2022). The aerosol particles are complex and heterogeneous in their physical features, chemical composition and origin. These physical and chemical features are different, due to the large variability of emission sources and formation and post-formation processes. Depending on the size of the particles, they remain in suspension long enough to penetrate the respiratory tract (Taunton et al., 2011). Information on particle size, shape, and elemental composition is essential for understanding the contribution of emission sources (Mico et al., 2015).

In this work, the microstructure and chemical composition of deposited particulate matter from different vehicle exhaust emissions was investigated. It has been reported that particulate matter from diesel engine exhaust is a complex mixture of organic molecules such as insufficiently oxidized carbon, metal oxides, sulphate and nitrate groups (Boughedaoui et al., 2004). The properties of these particles depend on some characteristic such as engine operating conditions, fuel composition, lubricating oil and exhaust gas filtration equipment (Taunton et al., 2011). Depending on the physical and chemistry properties of different sizes of particles, and the great variability of their emission, they remain suspended for enough time to penetrate into the respiratory tract (Taunton et al., 2011; Mico et al., 2015).

The environmental effects of rapid industrialisation, urbanisation and energy demand have resulted in countless incidents of air, soil, and water contamination with toxic pollutants, threatening humans and ecosystems with serious health risks (Manisalidis et al., 2020; Yusuf, et al., 2022). Transportation activities have become a significant source of pollution due to the large number of pollutants released during the incomplete combustion (Yusuf, et al., 2022).

In Algeria, where urbanization and motorization are developing rapidly during the last few decades, degradation of air quality and adverse health effects are already perceived (Yassaa et al., 2001a; Yassaa et al., 2001b; Boughedaoui et al., 2004; Kerbachi et al., 2006; Ladji et al., 2009a ; Ladji et al 2009b). The population growth in Algiers conurbation has resulted in about a 50% increase of the car fleet between 2002 and 2020. The National Office of Statistics (ONS) reported that Algiers had 1,731,664 vehicles for 3.1 million inhabitants in 2020. This important growth has contributed greatly to air pollution in the Algerian Capital (ONS, 2020).

In recent years, diesel engines have played an important role in transportation operations due to their low maintenance requirements, fuel economy and better performance (Yusuf, et al., 2022). However, diesel engines are classified as a major source of atmospheric pollutants, posing a serious risk to human health (Yusuf, et al., 2022).

Most diesel exhaust studies were aimed at obtaining information on the average chemical characteristics by bulk analysis techniques, while characterizing individual particles is important for health impact assessment, providing the chemical composition and morphological information at the microscopic level. The particle analysis can reveal the types of particles of major signatures in the vehicular exhausts (Toner et al., 2006; Chernyshev et al., 2018).

According to the investigation realized about market fuels in Algeria, five types fuels for gasoline and diesel engines are largely used: normal petrol, premium gasoline, unleaded gasoline, gas oil and liquefied petroleum gas fuel (LPG/F), which covers all automotive and industrial applications. The main fuels that are extensively used in Algeria are investigated in our work for determining their characteristics.

Automobile gasoline (normal petrol, premium gasoline and unleaded gasoline) is light hydrocarbon oil used as fuel in spark-ignition engines. Its distillation temperature is between 35 and 200°C, while diesel oil is a heavy oil composed of a mixture of hydrocarbons (paraffinic, naphthenic, aromatic and olefinic) whose distillation temperature is between 200 and 380°C, their flash point is always greater than 50 and their greater density than 0.82(unit) (Sarıkoç, 2020).

The premium is of the same nature as regular gasoline; its composition differs from gasoline by the higher benzene content (4 to 6%) due to the decrease in lead content; the higher sulfur content (0.5%). The unleaded super has totally supplanted the super lead. The additives used are MTBE (methyltertiobutyl ether), the most used additive, and benzene added to improve the octane number (Sarıkoç, 2020).

As regards to annual consumption, according to the Hydrocarbons Regulatory Authority (ARH) from the Ministry of Energy and Mines (Algeria), for the year 2022, diesel consumption has reached 10.1 million tonnes (MT), an increase of more than 4% compared to 2021, estimating that this consumption "should continue to increase thanks to the economic growth recorded in Algeria". On the other hand, the consumption of liquefied petroleum gas-fuel (LPG-c) increased by 20%, reaching 1.5 MT in 2022, compared to 1.2 MT in 2021, thanks to the efforts agreed upon by different actors for several years to promote this eco-responsible product offered to the consumer at a very attractive price (9 DA/litre) compared to other types of fuel, in addition to many other incentive measures. On the other hand, gasoline consumption fell by 2.26% to 3.3 MT, compared to 3.4 MT in 2021. A drop that is explained by the increase in LPG-c consumption, which should reach 6.8 million tonnes by 2050, knowing that the production capacity of this fuel is currently estimated at 4 million tonnes per year.

This paper aims to characterize the microstructure and chemical composition of deposited particulate matter (DPM) in the vehicular exhaust emissions of different engines; diesel, premium gasoline and unleaded gasoline vehicles. The actual parameters measured (e.g. size, chemical composition, etc.) was performed by Laser granulometry while the physicochemical analysis was achieved by X-ray Fluorescence (XRF), Fourier Transform Infrared (FTIR), X-ray diffraction (DRX), Scanning Electron Microscopy (SEM), and thermogravimetry (TG) were employed for spectral characterization.

Materials and methods

DPM collection

The process of DPM involved gathering particles left on the surfaces of various vehicles' exhaust pipes. The samples were protected from light and moisture until analysis. It is important to note that all surfaces or walls of the exhaust pipes were carefully cleaned using a dry cloth and the considered sample particulates were collected one year later, so these sample particles resulted from one-year accumulation on the walls of

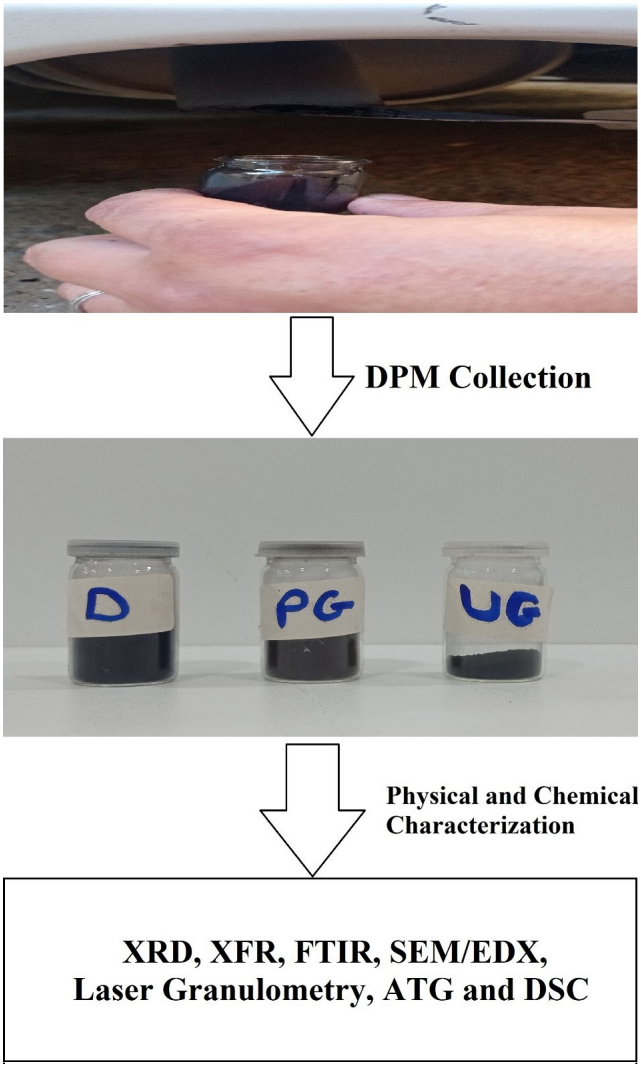


Figure 1: *Collection of DPM and the instruments used for their characterization.*

the exhaust pipes. For reproducibility, at least three samples for each fuel type were used for each analysis. It should be noted, the samples were homogenised and analysed as a single sample for both super gasoline and unleaded gasoline.

Description of cars and buses utilised in this study

The emissions of light cars of different manufacturers, with an age of between 2 and 10 years were investigated. In this contribution, the selection was made as follows: (i) two vehicles using super gasoline fuel, (ii) two others using unleaded gasoline fuel (UG). Also, DPM of diesel (D) was collected from a large (40 person) bus in the year 2013.

DPM characterization

XRD analysis of DPM was performed on a Siemens D-5000 diffractometer with Cu- Ka radiation (1.5418 A°).

DPM XRF analysis were conducted by Rigaku ZSX Primus II X-ray Fluorescence Spectrometer, elementary coverage: ^{4}Be to ^{92}U. Closing window, Rh-anode, 3kW or 4 kW, 60kV. Primary X-ray filter: Al25, Al125, Ni40 and Ni400. Heavy Element Detector: Scintillation Counter (SC).

FTIR analysis of the particle chemical composition was measured using Bruker Brand Spectrometer in a transmission mode, at 2 cm^{-1} resolution.

Individual DPM were examined using the SEM/EDX of the Brand Quanta 250 with tungsten filament produced by FEI field emission scanning electron microscope, with a maximum resolution of the images: 3584 x 3094 pixels (16 bits). EDX Bruker system EDS Quantax 200 for X-ray microanalysis images in secondary or backscattered electrons (topographic information and compositional contrast) with a lateral resolution of 0.1 µm approximately (magnification up to 20,000).

The equipment used for volumetric distribution measurements by laser granulometry was a MALVERN MATERSIZER 2000 granulometer, equipped with Scirocco as a dry dispersion accessory; sensitivity normal, absorption 0.1 and obscuration 5.68%.

Thermogravimetric (TG) analysis of DPM was done on a Thermal Analyst (Setaram Set Sys 16/18). Analyses were carried out in flowing air at a constant heating rate of 10 °C min^{-1} (25–950 °C).

The procedure for the collection of DPM and the instruments used for characterization were summarised in Figure 1.

Results and discussions

Mineralogical analysis and chemical composition by XRD and XRF techniques

As shown on Figure 2, three peaks are seen on Diesel diffractogram, ranged from 10 to 40°. The first one is weak, at 17.41°.

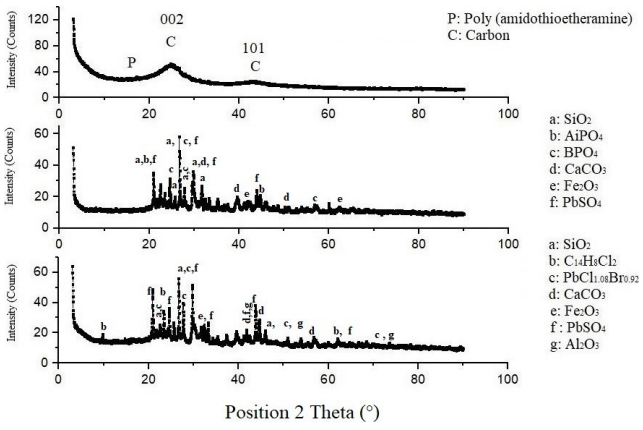


Figure 2: *XRD spectrum of engine particles s: a-Diesel (D), b- Premium gasoline (PG), c- unleaded gasoline (UG).*

The two others located at 24.76° (002) and 43.15° (101), can be attributed to the presence of carbonic phase (graphite) as the

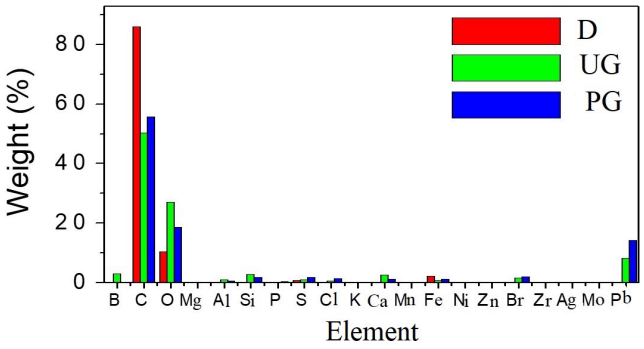


Figure 3: *Basic composition of engine emissions determined by XRF.*

major phase, which was confirmed by XRF (85.9% of carbon) (Guo et al., 2015), Figure 3.

In contrast, as indicated in Figure 1, unleaded gasoline (UG) and premium gasoline (PG) diffractograms confirm the presence of crystalline phases with greater intensities in the range [20° to 50 °]. The identification of these peaks reveals the presence of silicates (quartz: SiO_2), oxides (Fe_2O_3), sulphates ($PbSO_4$), phosphates (BPO_4, $AlPO_4$), carbonates ($CaCO_3$) and other compounds (Satsangi and Yadav, 2014).

By comparing UG to PG diffractograms, peaks attributing to $C_{14}H_8Cl_2$ (dichloroanthracene) and PbCl Br_2 occurred only in PG samples. This result can be explained by the existence of a non-burned fraction of PG. The presence of some peaks at low intensities in different positions in both PG and UG diffractograms can be attributed to impurities present during sampling, including dolomite ($CaMg(CO_3)_2$), iron-zinc oxide ($ZnFe_2O_4$), sulfate: magnesium (Mg_2SiO_4), calcium-aluminum ($KAlSi_3O_8$), and phosphate (BPO_4, $AlPO_4$, $PbSO_4$ $PbMoO_4$) (Figure 4 (a, b and c)).

Comprehensive XRF analysis of the particles from the PG and UG engines as depicted in Figure 3, were similar and consisted mainly of Carbon 55.8% and 50.3% by weight, respectively.

The composition of Pb in the particles of PG (14.3% by weight) was greater than that of UG (8.14%). This result can be attributed to the parallel use of two fuels. The presence of element B in vehicles without Pb (3.08% by mass) was also detected. Other trace elements such as Mg, P, K, Mn, Ni, Zn and Mo may be due to the impurities that can accumulate on the exhaust pipes of the various vehicles, which were derived essentially from the external environment such as the deposit of sludge, the products of maintenance during the washing of the vehicles.

The non-carbon content can be divided into two groups: (i) elements derived from motor oil additives; P, S, Ca, Zn, Mg, Mo, and probably Na; and (ii) elements of used metals; Fe, Cr, Al, Cu, and Br (Uy et al., 2014).

Fourier transform infrared spectroscopy (FTIR)

Figure 5 shows FTIR spectra of considered samples produced

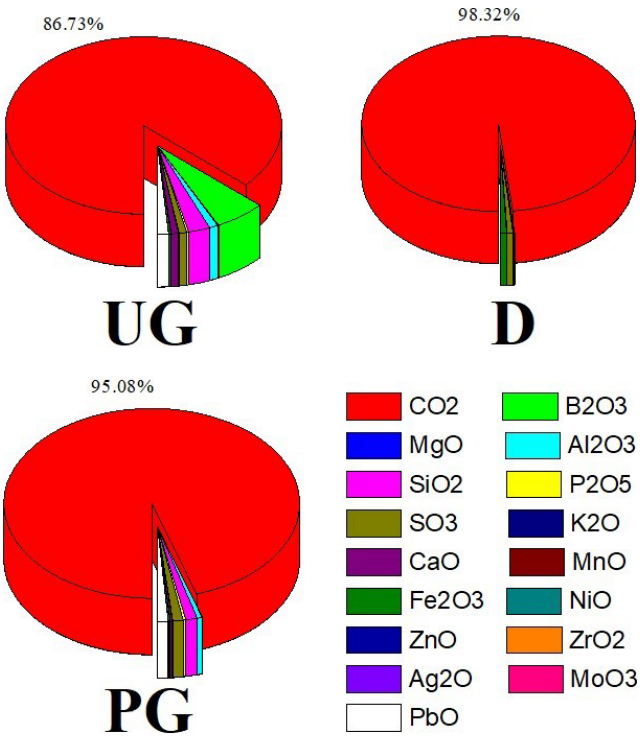


Figure 4: *Engine emissions of oxide composition by XRF: a-Diesel, b-Unleaded gasoline and c- Premium gasoline.*

from diesel engines, premium gasoline and unleaded gasoline vehicles. These spectra are similar, showing the same vibration frequencies with differences in peak intensities. Indeed, PG super FTIR spectra is characterized by a peak with high intensity at 2922.67 cm^{-1}, whereas it appears lower in the spectra of the two other samples. This can be explained by the presence of asymmetric vibration of CH_2 groups. These results are similar to those reported in the literature (Guerrero et al., 2013; Popovicheva et al., 2014; Sahu et al., 2016).

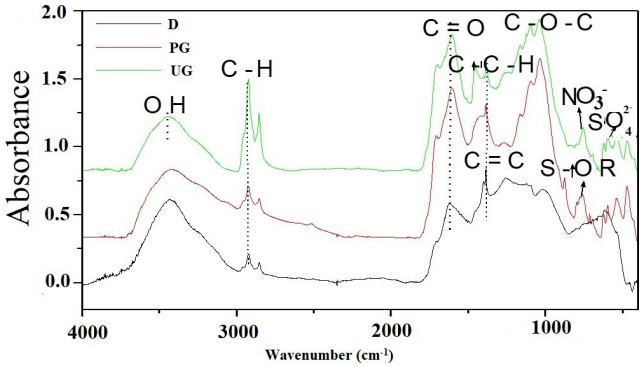


Figure 5: *Comparative FTIR Spectra of the engine emissions.*

Absorption between 1700 and 1590 cm^{-1} generally correspond to the C=O stretching vibrations. Also, other absorption peaks are seen in 1700-1000 cm^{-1} region. The most important ones correspond to the stretching vibrations of C=O of carboxylic acids (Manoj et al., 2012; Sahu et al., 2016). The peak at 1745 cm^{-1} corresponds to the carbonyl groups of esters while that at 1533 cm^{-1} corresponds to the C=C bond vibrations resulting

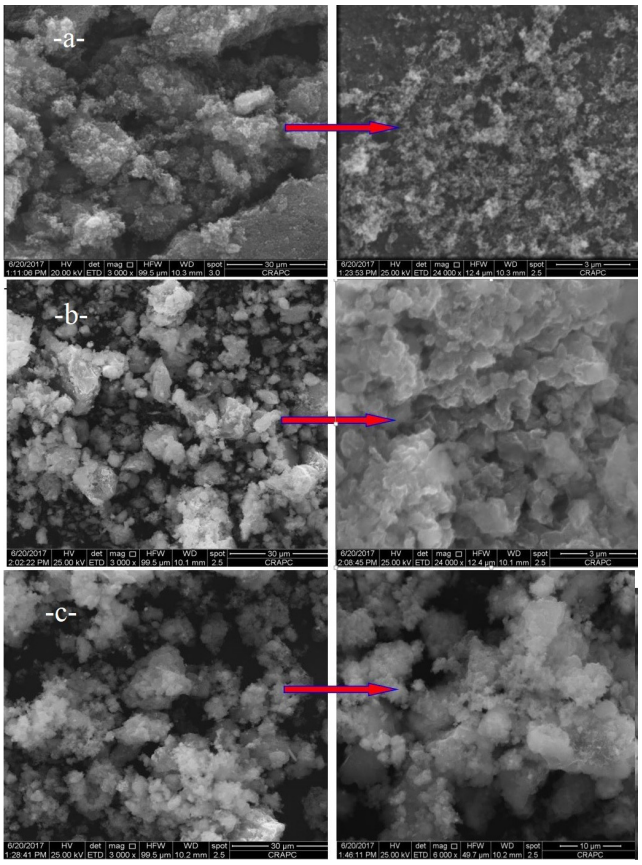


Figure 6: *SEM images of the engine particles: a-Diesel, b-Unleaded gasoline and c- Premium gasoline.*

from the aromatic ring or alkene functional groups (Guerrero et al., 2013). The massive absorption in 1400–1450 cm^{-1} region, and the peak at 1374.64 cm^{-1} can be attributed to the symmetric and asymmetric alkyl group vibrations (Guerrero et al., 2013; Popovicheva et al., 2014).

The bands lying between 1550 and 1380 cm^{-1} correspond to C=C bond vibrations of the aromatic group (Sahu et al., 2016). The aliphatic C-H plane deformation of CH_2/CH_3 groups were found in 1380 cm^{-1} and 1445 cm^{-1}, respectively (Manoj et al., 2012). The peak at 1162.17 cm^{-1} in both premium gasoline and unleaded gasoline correspond to the O-C function. The response around 1020 cm^{-1} can be attributed to the functions C-O-C and C = O, O-S, P-OR, Si-OR in all three samples (Sahu et al., 2016). The bands appeared in the 1000–1300 cm^{-1} region is a characteristic of C-C aromatic functions and C-H plane deformation, but the most important structure correspond to C-O-C ether (Manoj et al., 2012; Guerrero et al., 2013). The bands lying between 1100 and 1170 cm^{-1} correspond to the $C\text{-}CH_2\text{-}O$ vibrations, C-O-C asymmetric vibrations and C-C stretching groups (Guerrero et al., 2013). The two peaks at 594.26 and 626.46 cm^{-1} correspond to SO_4^{2-} ion (Popovicheva, 2014) produced due of sulfur contaminants in the fuel or oil.

Peak at 874.38 cm^{-1} in spectra of premium gasoline and unleaded gasoline can be probably explained by NO_3^- ions resulting from the higher nitrification of the particle surface in the fuel exhaust.

Scanning electron microscopy (SEM/EDX)

As it can be seen on Figure 5, the microscopic images of particles resulting from D, UG and PG indicate similarities in their morphological features. There are two forms, a spherical shape resulting from the combustion of organic matter (fuel) and a non-regular angular shape (irregular structure) which corresponds to the inorganic elements present in the fuel (Pb) and were originated from the external environment (aluminum silicate and calcium silicate). This can be explained by the fact that particles have been found spherical with the tendency to agglomerate and collect on the surface. Diesel particles are vague and can undergo some decomposition under the SEM beam.

Laser granulometry

From Figure 7 and Table 1, one can infer that size particles emitted by the different engines vary between 0.14 and 478.63 µm. The average diameter of diesel particles D was 4.49 µm, the maximum particle size was 15.13 µm and the minimum was 0.21 µm.

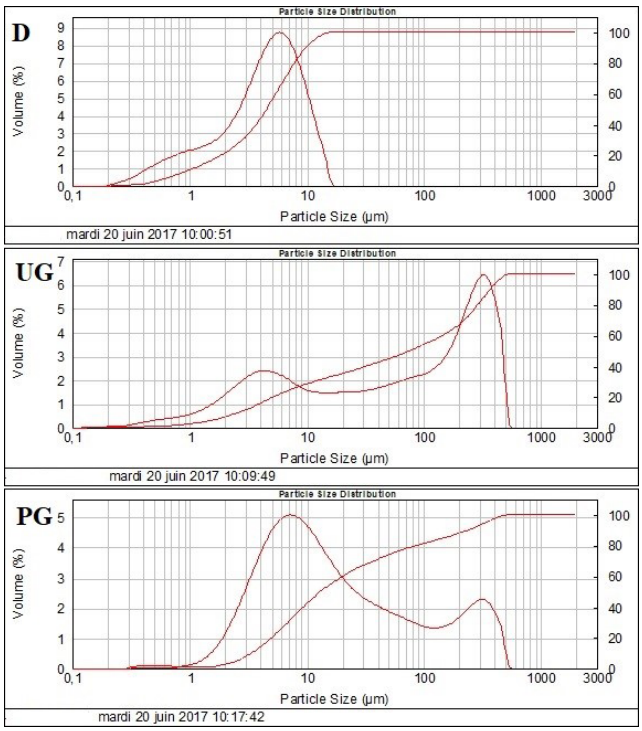


Figure 7: *Particle size distribution of particles: D, UG and PG.*

The average diameter of aerosol particles emitted from the combustion of premium gasoline (PG) and unleaded gasoline (UG) were 12.86 and 72.32 µm, respectively. It is worth noting that the particle size is finer in the case of Diesel, making them dangerous once inhaled, as they can reach deep inside the respiratory tract and get incorporated into the blood stream further damaging other viscera (Kim et al., 2015; Maricq and Xu, 2004; Mathis et al., 2005; Oh et al., 2011; Chiatti et al., 2016; Sahu et al., 2016).

Interestingly, it is important to highlight that analysis resulted from laser granulometry matched those very well performed by the SEM.

Table 1: *Particulate Size Distribution of Diesel (d10, d50 and d 90), Premium Gasoline (PG) and Unleaded Gasoline (UG).*

	Diesel	Premium Gasoline	Unleaded Gasoline
[Dmin-Dmax](µm)	0.209-15.136	0.275-478.630	0.138-416.869
d(0.1) (µm)	0.937	3.332	2.571
d(0.5) (µm)	4.495	12.863	72.329
d(0.9) (µm)	9.720	229.205	361.373

Thermal analysis (TGA and DSC)

TGA /DSC analyses were applied to determine the fraction of volatility and oxidation properties of the particles produced by the three types of engines. Figures 8 and 9 show the typical weight loss curves and DSC signal of the three types of particles, respectively.

As it can be noted from Figure 8, comparative thermogravimetry of particles shows a loss of mass between [100-1000°C], corresponding to hydrocarbon desorption (evaporable organic matter of 51.7%) for diesel (Simão et al., 2006). As it was also seen in the associated heat release curve (DSC) depicted in Figure 9, the increase in the released heat started at about 250 °C to 432 °C with an enthalpy of 9853 J/g. This can be explained by the endothermic evaporation of the volatile organic fraction (VOF). This implies that light hydrocarbons on the soot will evaporate above 200 °C (Oh et al., 2011).

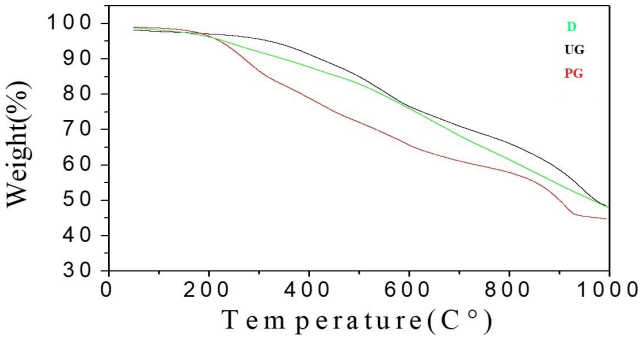


Figure 8: *Comparative spectra of thermogravimetric analysis (TGA) emissions from engines.*

For PG, the TGA curves show a loss of mass (53.65%) with the formation of two bearings. The first reached 16% in the range of [100–350 °C] which corresponds to the evaporation of water (Müller et al., 2006), and the 2nd level reached 37.5 % in the range of [350–900 °C] which corresponds to the removal of organic matter.

For UG, ATG curves show a total percentage of evaporable organic matter of 21% to 49.5% in the range of [100–550 °C] and 18% in the range of [550–950 °C]. This evaporable organic material corresponds to the non-burned hydrocarbon fraction (piled up) by the engines of the vehicles.

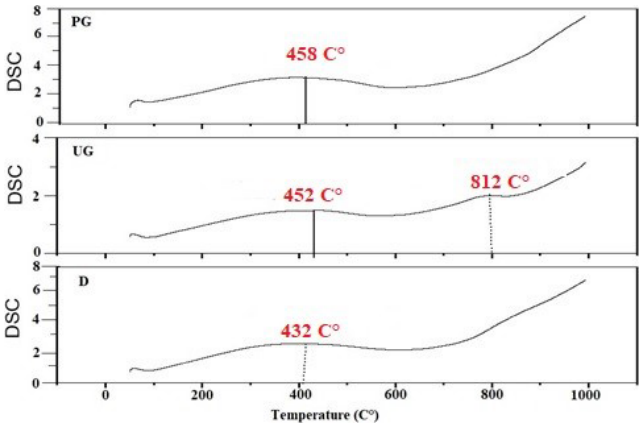


Figure 9: *Comparative spectra of DSC of motor emissions.*

The residual part of the particles corresponds to the oxides detected by XRF (SiO_2, CaO, P_2O_5, Fe_2O_3, etc.) (Simão et al., 2006).

The DSC curves of UG compared to that of PG, show the formation of a new endothermic peak at about 812 °C with enthalpy equal to 209J/g (from the rest of the organic matter) (Oh et al., 2011).

Conclusions

The microscopic and chemical characterizations of the particles emitted by the diesel engines, and petrol engines utilising premium and unleaded gasoline have allowed a comprehensive determination of their physicochemical properties with regards to the morphologies, the elemental compositions, the organic/inorganic contents and the particle size and structure.

The XRD and XFR analyses have proven to be powerful in providing useful insights about the microscopic structures of different fuels. They showed, in the case of particles generated by diesel engines, that many organic and inorganic compounds presented an amorphous structure with the appearance of two well-distinguished peaks, the first was the most intense at 24.76° (002), and the second at 43.15° (101). This can be attributed to carbonic (graphite) as the major phase, which was confirmed by XRF (85.9% of carbon). Unlike diesel (D), the unleaded gasoline (UG) and premium gasoline (PG) diffractograms showed a well-crystalline phase amorphous with greater intensities in the range [20° to 50°]; the identification of these peaks revealed silicates (quartz: SiO_2), oxides (Fe_2O_3), sulfates ($PbSO_4$), phosphates (BPO_4, $AlPO_4$), carbonates ($CaCO_3$), and others.

The chemical composition of diesel, premium gasoline, and unleaded gasoline particles was dominated by aliphatic CH groups in alkanes with relatively low C=O groups in carboxylic acids, ketones, aldehydes, esters, lactones, and sulphate inorganic salts.

Based on laser particle size analysis, it was found that the particle size was the finest in diesel, which makes them dangerous once inhaled, as they can reach deep inside the respiratory tract and get incorporated with the blood stream, thus damaging other

viscera. Prolonged exposure to these fine particles and highly toxic pollutants is likely to have a significant impact on health, resulting in symptoms of cancer, bronchitis, emphysema, and cardiovascular and pulmonary disease, as widely documented in the literature.

Declarations

Conflict of interest: The authors declare no competing interests.

About data availability statements

The datasets generated during and/or analysed during the current study are available from the corresponding author on reasonable request.

References

Alves, C., Evtyugina, M., Vicente, E., Vicente, A., Rienda, I. C., De La Campa, A. S., Tomé, M., & Duarte, I. (2023). PM2.5 chemical composition and health risks by inhalation near a chemical complex. *Journal of Environmental Sciences*, 124, 860-874. https://doi.org/10.1016/j.jes.2022.02.013

Boughedaoui, M., Kerbachi, R., Kessali, D., & Joumard, R. (2004). Mesure de la pollution plombifère dans l'air ambiant d'Alger. *Pollution atmosphérique*, N°181. https://doi.org/10.4267/pollution-atmospherique.1602

Chernyshev, V. V., Zakharenko, A. M., Ugay, S. M., Hien, T. T., Hai, L. H., Kholodov, A. S., Burykina, T. I., Stratidakis, A. K., Mezhuev, Ya. O., Tsatsakis, A. M., & Golokhvast, K. S. (2018). Morphologic and chemical composition of particulate matter in motorcycle engine exhaust. *Toxicology Reports*, 5, 224-230. https://doi.org/10.1016/j.toxrep.2018.01.003

Chernyshev, V. V., Zakharenko, A. M., Ugay, S. M., Hien, T. T., Hai, L. H., Olesik, S. M., Kholodov, A. S., Zubko, E., Kokkinakis, M., Burykina, T. I., Stratidakis, A. K., Mezhuev, Ya. O., Sarigiannis, D. A., Tsatsakis, A., & Golokhvast, K. S. (2019). Morphological and chemical composition of particulate matter in buses exhaust. *Toxicology Reports*, 6, 120-125. https://doi.org/10.1016/j.toxrep.2018.12.002

Chiatti, G., Chiavola, O., Recco, E., & Palmieri, F. (2016). Soot Particles Experimental Characterization During Cold Start of a Micro Car Engine. *Energy Procedia*, 101, 662-669. https://doi.org/10.1016/j.egypro.2016.11.084

Guerrero, A., Anguebes, F., Castelán, M., Morales, V., García, R., Córdova, A. V., & Zavala, J. C. (2013). FTIR-ATR and Multivariate Calibration for the Prediction of Biodiesel Concentration in Petrodiesel Blends. *American Journal of Analytical Chemistry*, 04(07), 343-347. https://doi.org/10.4236/ajac.2013.47043

Guo, M., Cai, Z., Zhang, Z., & Zhu, M. (2015). Characterization and lubrication performance of diesel soot nanoparticles as oil lubricant additives. *RSC Advances*, 5(123), 101965-101974. https://doi.org/10.1039/C5RA19522E

Kerbachi, R., Boughedaoui, M., Bounoua, L., & Keddam, M. (2006). Ambient air pollution by aromatic hydrocarbons in Algiers. *Atmospheric Environment*, 40(21), 3995-4003. https://doi.org/10.1016/j.atmosenv.2006.02.033

Kim, K.-H., Kabir, E., & Kabir, S. (2015). A review on the human health impact of airborne particulate matter. *Environment International*, 74, 136-143. https://doi.org/10.1016/j.envint.2014.10.005

Ladji, R., Yassaa, N., Balducci, C., Cecinato, A., & Meklati, B. Y. (2009a). Annual variation of particulate organic compounds in PM10 in the urban atmosphere of Algiers. *Atmospheric Research*, 92(2), 258-269. https://doi.org/10.1016/j.atmosres.2008.12.002

Ladji, R., Yassaa, N., Balducci, C., Cecinato, A., & Meklati, B. Y. (2009b). Distribution of the solvent-extractable organic compounds in fine (PM_1) and coarse (PM_{1-10}) particles in urban, industrial and forest atmospheres of Northern Algeria. *Science of the Total Environment*, 10.

Lemou, A., Rabhi, L., Cherifi, N., Ladji, R., & Yassaa, N. (2020). Temporal variation of PM_{10}, $PM_{2.5}$ and gaseous pollutants (NO_x, SO_2) particles suspended in the atmosphere of an urban sit in Bou-Ismail. *Algerian Journal of Environmental Science and Technology*, 5.

Lemou, A., Rabhi, L., Merabet, H., Ladji, R., Nicolas, J. B., Bonnaire, N., Mustapha, M. A., Dilmi, R., Sciare, J., Mihalopoulos, N., & Yassaa, N. (2020). Chemical characterization of fine particles ($PM_{2.5}$) at a coastal site in the South Western Mediterranean during the ChArMex experiment. *Environmental Science and Pollution Research*, 27(16), 20427-20445. https://doi.org/10.1007/s11356-020-08168-7

Manisalidis, I., Stavropoulou, E., Stavropoulos, A., & Bezirtzoglou, E. (2020). Environmental and Health Impacts of Air Pollution : A Review. *Frontiers in Public Health*, 8, 14. https://doi.org/10.3389/fpubh.2020.00014

Manoj, B., Sreelaksmi, S., Mohan, A. N., & Kunjomana, A. G. (2012). Characterization of Diesel Soot from the Combustion in Engine by X-ray and Spectroscopic techniques. *Int. J. Electrochem. Sci.*, 7, 8.

Maricq, M. M., & Xu, N. (2004). The effective density and fractal dimension of soot particles from premixed flames and motor vehicle exhaust. *Journal of Aerosol Science*, 35(10), 1251-1274. https://doi.org/10.1016/j.jaerosci.2004.05.002

Mathis, U., Mohr, M., Kaegi, R., Bertola, A., & Boulouchos, K. (2005). Influence of Diesel Engine Combustion Parameters on Primary Soot Particle Diameter. *Environmental Science & Technology*, 39(6), 1887-1892. https://doi.org/10.1021/es049578p

Mico, S., Tsaousi, E., Deda, A., & Pomonis, P. (2015). Characterization of Airborne Particles and Source Identification Using SEM/EDS. *Eur. Chem. Bull*, 6.

Müller, J.-O., Su, D. S., Jentoft, R. E., Wild, U., & Schlögl, R. (2006). Diesel Engine Exhaust Emission : Oxidative Behavior and Microstructure of Black Smoke Soot Particulate. *Environmental Science & Technology*, 40(4), 1231-1236. https://doi.org/10.1021/es0512069

Oh, K. C., Lee, C. B., & Lee, E. J. (2011). Characteristics of soot particles formed by diesel pyrolysis. J*ournal of Analytical and Applied Pyrolysis*, 92(2), 456-462. https://doi.org/10.1016/j.jaap.2011.08.009

ONS. (2020). *National Office of Statistics of Algier*s, http://www.ons.dz.

Popovicheva, O. B. (2014). Microstructure and Chemical Composition of Diesel and Biodiesel Particle Exhaust. *Aerosol and Air Quality Research*. https://doi.org/10.4209/aaqr.2013.11.0336

Popovicheva, O. B., Kireeva, E. D., Steiner, S., Rothen-Rutishauser, B., Persiantseva, N. M., Timofeev, M. A., Shonija, N. K., Comte, P., & Czerwinski, J. (2014). Microstructure and Chemical Composition of Diesel and Biodiesel Particle Exhaust. *Aerosol and Air Quality Research*, 14(5), 1392-1401. https://doi.org/10.4209/aaqr.2013.11.0336

Rabhi, L., Lemou, A., Cecinato, A., Balducci, C., Cherifi, N., Ladji, R., & Yassaa, N. (2018). Polycyclic aromatic hydrocarbons, phthalates, parabens and other environmental contaminants in dust and suspended particulates of Algiers, Algeria. *Environmental Science and Pollution Research*, 25(24), 24253-24265. https://doi.org/10.1007/s11356-018-2496-y

Sahu, R., Adak, P., & Elumalai, S. P. (2016). Characterization of hazardous solid waste (soot) accumulated in tailpipe of typical Indian share autos. *Current Science*, 111(3), 560-564.

Sarıkoç, S. (2020). Fuels of the Diesel-Gasoline Engines and Their Properties. In R. Viskup (Éd.), *Diesel and Gasoline Engines*. IntechOpen. https://doi.org/10.5772/intechopen.89044

Satsangi, P. G., & Yadav, S. (2014). Characterization of $PM_{2.5}$ by X-ray diffraction and scanning electron microscopy–energy dispersive spectrometer : Its relation with different pollution sources. I*nternational Journal of Environmental Science and Technology*, 11(1), 217-232. https://doi.org/10.1007/s13762-012-0173-0

Simão, J., Ruiz-Agudo, E., & Rodriguez-Navarro, C. (2006). Effects of particulate matter from gasoline and diesel vehicle exhaust emissions on silicate stones sulfation. *Atmospheric Environment*, 40(36), 6905-6917. https://doi.org/10.1016/j.atmosenv.2006.06.016

Taunton, A. E., Gunter, M. E., Nolan, R. P., & Phillips, J. I. (2011). Characterization of minerals in pleural plaques from lung tissue of non- human primates. *An International Journal of Mineralogy, Crystallography, Geochemistry, Ore Deposits, Petrology, Volcanology and Applied Topics on Environment, Archeometry and Cultural Heritage*, 13.

Toner, S. M., Sodeman, D. A., & Prather, K. A. (2006). Single Particle Characterization of Ultrafine and Accumulation Mode Particles from Heavy Duty Diesel Vehicles Using Aerosol Time-of-Flight Mass Spectrometry. *Environmental Science & Technology*, 40(12), 3912-3921. https://doi.org/10.1021/es051455x

Uy, D., Ford, M. A., Jayne, D. T., O'Neill, A. E., Haack, L. P., Hangas, J., Jagner, M. J., Sammut, A., & Gangopadhyay, A. K. (2014). Characterization of gasoline soot and comparison to diesel soot : Morphology, chemistry, and wear. *Tribology International*, 80, 198-209. https://doi.org/10.1016/j.triboint.2014.06.009

Yassaa, N., Meklati, B. Y., Brancaleoni, E., Frattoni, M., & Ciccioli, P. (2001). Polar and non-polar volatile organic compounds (VOCs) in urban Algiers and saharian sites of Algeria. *Atmospheric Environment*, 35(4), 787-801. https://doi.org/10.1016/S1352-2310(00)00238-7

Yassaa, N., Youcef Meklati, B., Cecinato, A., & Marino, F. (2001). Particulate n-alkanes, n-alkanoic acids and polycyclic aromatic hydrocarbons in the atmosphere of Algiers City Area. *Atmospheric Environmen*t, 35(10), 1843-1851. https://doi.org/10.1016/S1352-2310(00)00514-8

Yusuf, A. A., Dankwa Ampah, J., Soudagar, M. E. M., Veza, I., Kingsley, U., Afrane, S., Jin, C., Liu, H., Elfasakhany, A., & Buyondo, K. A. (2022). Effects of hybrid nanoparticle additives in n-butanol/waste plastic oil/diesel blends on combustion, particulate and gaseous emissions from diesel engine evaluated with entropy-weighted PROMETHEE II and TOPSIS : Environmental and health risks of plastic waste. *Energy Conversion and Management*, 264, 115758. https://doi.org/10.1016/j.enconman.2022.115758

Yusuf, A. A., Inambao, F. L., & Ampah, J. D. (2022). Evaluation of biodiesel on speciated PM2.5, organic compound, ultrafine particle and gaseous emissions from a low-speed EPA Tier II marine diesel engine coupled with DPF, DEP and SCR filter at various loads. *Energy*, 239, 121837. https://doi.org/10.1016/j.energy.2021.121837

Yusuf, A. A., Yusuf, D. A., Jie, Z., Bello, T. Y., Tambaya, M., Abdullahi, B., Muhammed-Dabo, I. A., Yahuza, I., & Dandakouta, H. (2022). Influence of waste oil-biodiesel on toxic pollutants from marine engine coupled with emission reduction measures at various loads. *Atmospheric Pollution Researc*h, 13(1), 101258. https://doi.org/10.1016/j.apr.2021.101258

Research article

Analysis of air quality issues and air quality management status in five major African cities

Rafael Borge[*1], **Sebastian Lange**[2], **Robert Kehew**[2]

[1]Universidad Politécnica de Madrid (UPM). Department of Chemical and Environmental Engineering
C/ José Gutiérrez Abascal, 2 28006, Madrid, Spain
[2]United Nations Human Settlements Programme (UN-HABITAT), Planning, Finance and Economy Section
United Nations Avenue, Gigiri P.O. Box 30030, Nairobi, Kenya
*Corresponding author: rafael.borge@upm.es

Received: 15 March 2023 - **Reviewed:** 5 June 2023 - **Accepted:** 12 July 2023
https://doi.org/10.17159/caj/2023/33/2.15605

Abstract

Poor air quality is one of the main dysfunctions of the rapid urbanisation process in Africa. Although the information is fragmented, the limited evidence available points out that air quality is a leading health risk in Africa, especially affecting the poorest, most vulnerable communities. In this study, we look into five cities in Africa to understand the nature of urban air quality issues and to delve into the initial responses. We report on the status of the main air quality management dimensions, including air quality standards and monitoring strategy/capabilities, emission inventories and air quality modelling, health impact assessment studies, communication practices, development and implementation of clean air action plans (in connection with other relevant strategies, i.e. climate change policies) and governance issues. We find that all cities have limited monitoring capabilities although communication strategies may differ substantially. While indoor pollution sources are declining in more developed economies, traffic is a growing concern in all five cities. In most cases, air quality issues are mostly related to $PM_{2.5}$ and natural contributions worsen air pollution from road transport, biomass and open waste burning. While nationally-driven strategies, often based on large-scale engineering projects and financial schemes, may report substantial gains in early stages, multi-level governance and planning is likely to maximize benefits and provide a useful framework for the complex problem of air quality management in the long run. The integration of air quality plans in overarching strategies to tackle persistent poverty and social inequity is urgently needed.

Keywords

urban air quality, air quality standards and monitoring, emissions inventories and modelling, communication, clean air action plans, governance

Introduction

Urbanisation has historically been accompanied by industrialisation, economic growth and better living standards for citizens. However, demographic growth requires public policies to steer the urbanization process and to help ensure a more equitable distribution of wealth (UN Habitat, 2010), especially in the context of rapid population movements towards urban areas. While Africa currently is the least urbanised of the settled continents, it is experiencing the fastest rate of urbanisation in the world, with an average annual growth rate of 2.55% in the period 2000 – 2015. Such trends are expected to cause the urban population of the continent to rise from 1.3 billion today to a projected 2.5 billion by 2050 (UN, 2019). Many parts of Africa seem to be experiencing urbanisation without significant productivity increases or adequate service provision, infrastructure and regulation, leading to the "urbanisation of poverty". At present in sub-Saharan Africa, 59% of the urban population live in slums (UN Habitat, 2016), while only 45% have access to basic sanitation facilities. Many cities have evolved as a collection of fragmented neighbourhoods not connected by efficient transportation or other networked infrastructure; this results in congestion and other dysfunctions such as poor air quality (Stanaway et al., 2018) and high population exposure in urban areas where both emissions from unplanned and unregulated activities occur.

While the major cities on the continent do not experience the same levels of chronic air pollution as some cities in Asia do (Lelieveld et al., 2015), available data suggests that African cities are bearing a considerable share of the global burden of air pollution-related disease. The limited literature on air pollution in Africa identifies particulate matter (PM) as the single most

relevant pollutant from a public health perspective (Burnett et al., 2018), even when PM chemical composition is usually not characterized. The review of PM source apportionment analyses carried out by Karagulian et al. (2015) identified eleven such studies for Africa (out of 419 for the entire world). On average, the largest contributor to $PM_{2.5}$ ambient concentration is household fuel burning, responsible for 34% of total concentration –the largest relative concentration on the planet. Natural sources (desert dust and sea salt) were the second source in importance, accounting for 22% of $PM_{2.5}$. (For comparison, natural sources represent the largest share of ambient concentrations in Middle-eastern countries, as much as 52%). Traffic was identified as the third largest contributor to $PM_{2.5}$ levels in Africa, with a 17% share. This sector was, however, the main cause of PM_{10} levels, contributing around 34% of the total ambient concentration of this pollutant on the continent. Other studies have identified open-field burning activities as a major source of this pollutant in Africa (Naidja et al., 2018). According to the continental DACCIWA inventory (Keita et al., 2018), PM emissions from the residential sector, mostly related to household solid fuel combustion, represent more than 70% of total combustion sources on the continent. As a consequence, African cities need to deal with both indoor and outdoor air pollution, in contrast with OECD country cities, where indoor air quality is driven by outdoor concentration levels (Tang et al., 2018).

Traffic is a relevant polluting sector in nearly all African cities, and its relative importance is expected to grow substantially in the near future (Liousse et al., 2014). Use of old cars, the poor quality of fuel or diesel fuel, underdeveloped infrastructures, and unorganized public transport (Assamoi & Liousse, 2010) are common challenges in African cities, contributing to worsening air pollution (Mbow-Diokhane, 2019). It should be noted that exhaust emissions are not the only source of PM from mobile sources – abrasion (road, brake and tyre wear) and dust re-suspension can be even larger contributors to PM emissions. Non-exhaust emissions are particularly relevant in Africa where unpaved roads make up the majority of the road network (Naidja et al., 2018).

While deaths attributable to other risk factors such as unsafe water, unsafe sanitation, and childhood malnutrition have decreased markedly since 1990, air pollution-related deaths have risen in Africa (Roy, 2016) and remain a major public health issue. Heft-Neal et al. (2018) suggest that modest reductions in the levels of fine particulate matter ($PM_{2.5}$) in the air in African cities may have larger health benefits to infants than most known health interventions. There is a clear economic case to deal with air pollution in Africa as well. The World Bank estimates that, in 2013, air pollution costs in Sub-Saharan Africa totalled 3.8 percent of GDP (WB, 2016). This means that the loss in economic output due to air pollution-related morbidity and mortality may be as high as USD 1.63 billion in countries like Ghana (Fisher et al. 2021). In addition, since the emissions of air pollutants often coincide with the release of greenhouse gases, worsening air quality is intimately related to global warming. The impacts of climate change have been identified as an additional, growing health threat in Africa, especially for the more vulnerable layers of society (Chersich et al., 2018).

This underpins the importance of prioritizing air quality interventions in the political agenda of African major cities. To contribute to that goal this study analyses the current air pollution issues as well as air quality management and initial responses in five illustrative cities in the continent. Comprehensive studies regarding air quality management status in Africa are very limited (Schwela, 2012) and there are no studies comparing approaches in specific urban areas that may illustrate effective options to improve air quality in the continent. This research is done in the framework of an initiative of UN Habitat to raise awareness about the issue of urban air quality in Africa (UN Habitat, 2023) and ultimately, aims at extracting general conclusions and identifying effective approaches to improve urban air quality in the whole African region. The specific scientific goals are to understand urban air quality issues (current ambient concentration levels of any pollutant relevant to air quality, sources, and impacts related to them) and analyse whether interventions from national, regional, or local administrations as well as any other initiatives driven by international institutions, donors, companies or the civil society are adequately targeted to those issues and draw conclusions and recommendations that may improve air quality management in Africa and curb harmful emissions in an efficient way considering limitations and constraints of real cities.

Methodology

Case studies

The case studies selected to investigate air quality issues and policies in African cities correspond to five large (capital) cities that cover a variety of geographical locations (Figure 1) and socio-economic development stages (Table 1) (in alphabetical order): Accra (Ghana), Cairo (Egypt), Cape Town (South Africa), Dakar (Senegal) and Nairobi (Kenya). While all of them face similar challenges related to rapid urbanization, social inequalities and population growth, each presents specific air quality issues and has different constraints and prospects and thus, they may contribute to building a more comprehensive picture and help identify policy approaches toward air quality issues in other urban areas in the continent. The selection criteria included the availability of information as well as previous cooperation track with UN-Habitat and an on-the-ground presence to assure the accuracy and relevance of the analysis. Further details on socio-economic and geophysical conditions and specific emission sources in each city can be found in UN Habitat (2023).

According to Stanaway et al. (2018) total premature deaths per annum from outdoor (ambient) air pollution are on the rise in the case study countries (Figure 2). In all of them, premature deaths from air pollution per annum outstrip deaths from unsafe water, sanitation, and underweight childhood (Roy, 2016). Since about 2005, the five countries under review have collectively had some success in curbing premature deaths from

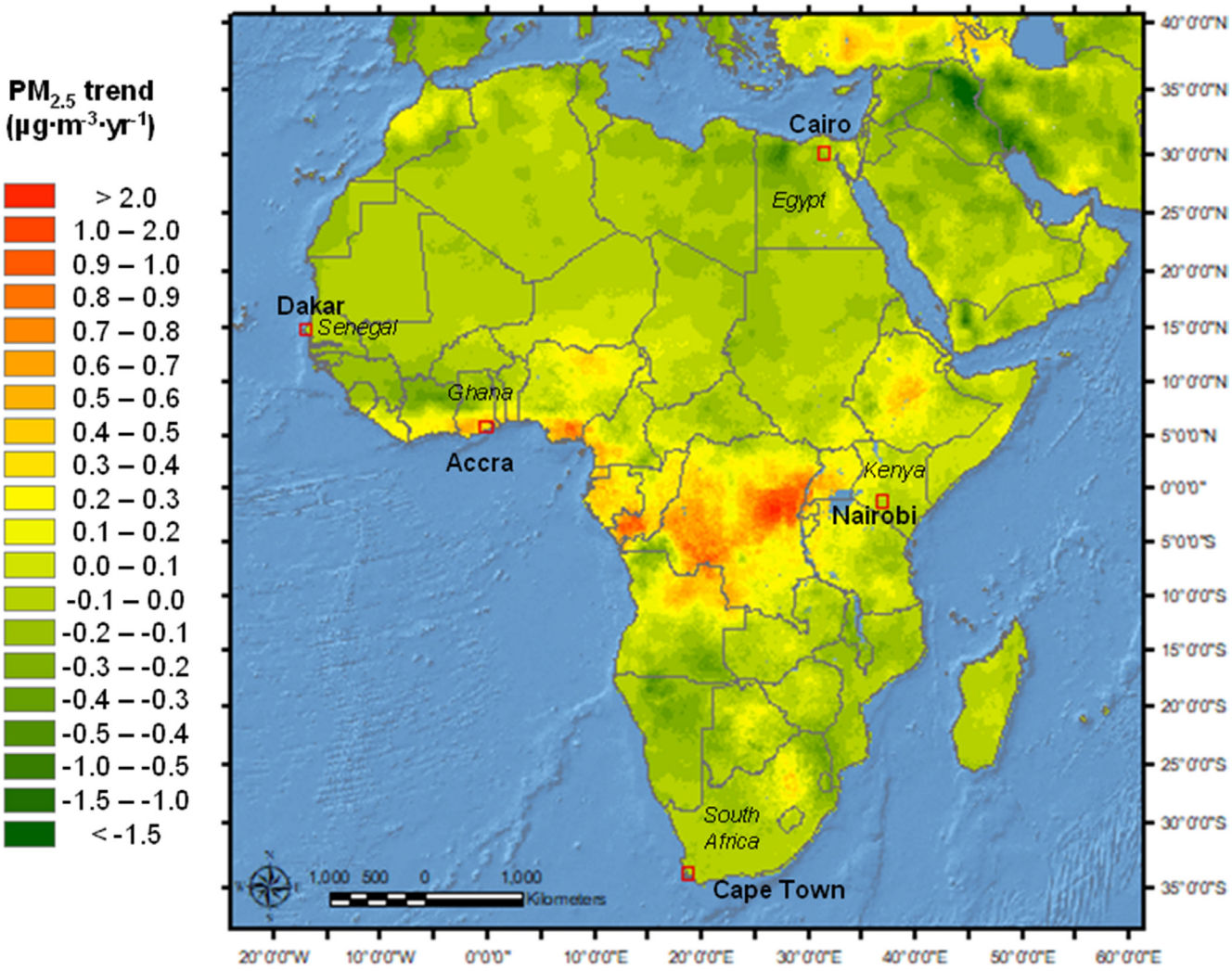


Figure 1: *Anthropogenic $PM_{2.5}$ ambient concentration trends in the 2000-2016 period and location of the five case studies (elaborated from van Donkelaar et al., 2018)*

Table 1: *Summary of socio-economic and demographic information of the five case studies*

City	Country, national GDP (USD)[1] and poverty headcount ratio (%)[2]	Population (million people)[3]	Recent annual growth rate (%) and future projections (%)[4]
Accra	Ghana, 4738 (56.9%)	2.3	2.1% / 2.2%
Cairo	Egypt, 12390 (61,9%)	18.8	2.2% / 2.0%
Cape Town	South Africa, 13730 (57.1%)	4.1	2.6% / 1.9%
Dakar	Senegal, 3776 (88.1%)	2.8	2.6% / 3.0%
Nairobi	Kenya, 3461 (86.5%)	3.9	3.8% / 3.9%

(1) GDP per capita, PPP (current international $) (WB, 2018)
(2) Poverty headcount ratio at $5.50 a day (2011 PPP) (% of population) (WB, 2018)
(3) 2018 population according to UN (population division) (UN, 2019)
(4) Average population growth rate (2000-2015) and growth rate projection (2015-2030) according to UN (population division) (UN, 2019)

indoor air pollution. More specifically, available data (Stanaway et al., 2018) suggest that household air pollution is generally a larger health concern in the lower income countries while high ambient air concentrations have greater relative importance in emerging economies. The trends in Egypt and South Africa clearly show that the burdens of disease have shifted towards outdoor (ambient) air quality over the last years while per capita GDP has grown; meanwhile, indoor air quality still dominates health impacts in Senegal, Kenya and Ghana (Figure 3).

Mortality rates attributed to the joint effects of household and ambient air pollution vary considerably among countries,

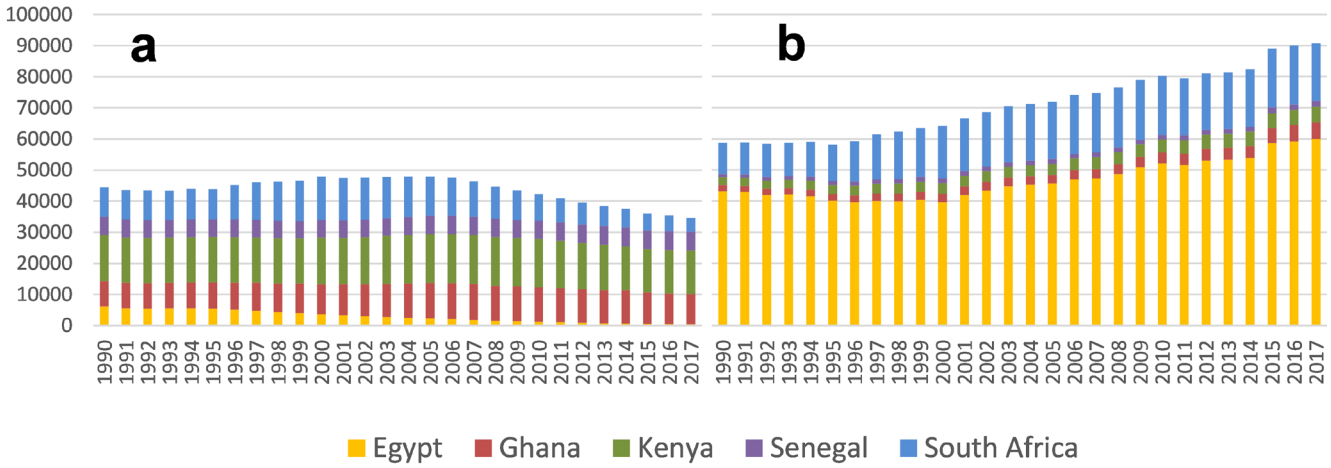


Figure 2: *Total deaths associated with air pollution in the countries where the five case study cities are located, according to the global burden of disease 2017, a) indoor, b) outdoor (elaborated from Stanaway et al., 2018)*

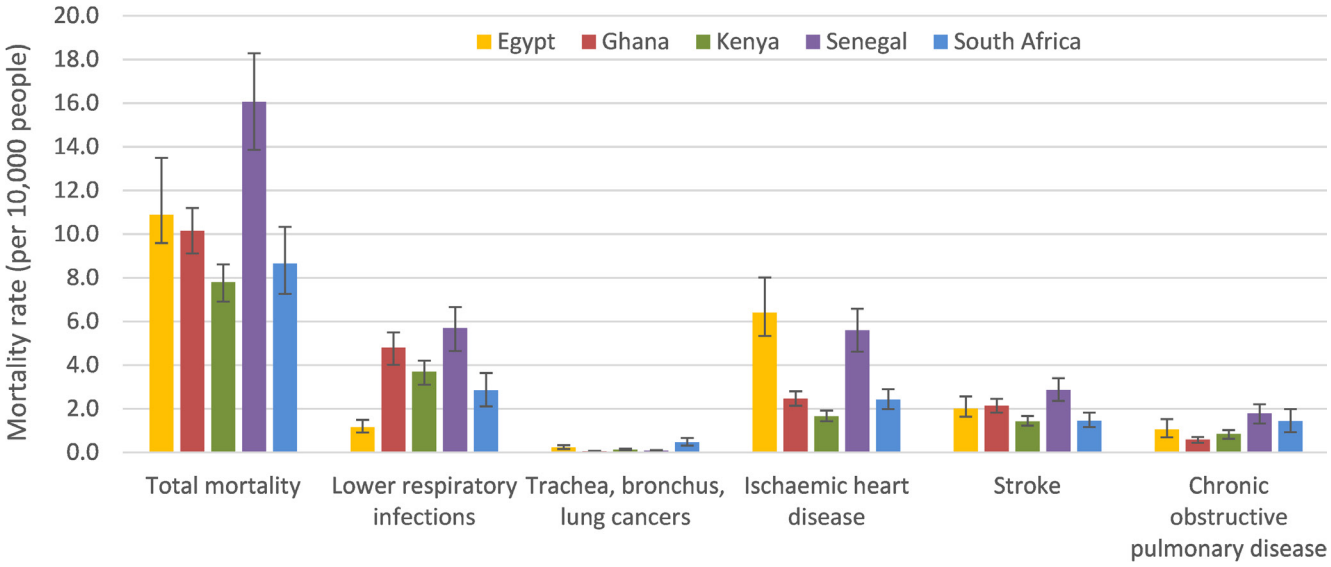


Figure 3: *Mortality rates attributed to joint effects of household and ambient air pollution (elaborated from the Global Health Observatory data repository, WHO, 2016). The bars show the 95% confidence intervals.*

as well as in the diseases that cause those death (Figure 3). According to the Global Health Observatory (WHO, 2016), lower respiratory infections are the leading cause of air pollution-related mortality in Ghana and Kenya, while ischemic heart disease dominates such deaths in Egypt. These two conditions have a similar contribution to premature deaths in Senegal and South Africa. Overall, Senegal presents the highest mortality rate due to air pollution.

Approach and data sources

The research relies on a desk review of scientific papers published in international journals as well as official reports and thematic studies from reference international organizations. No specific search or selection criteria were set regarding the type of documents or production dates, although preference was given to contrasted sources and updated information. Data scarcity (e.g. the lack of local emission inventories or reliable air quality data) prevents us from performing a fully consistent comparison and analysis. Therefore, any reliable source that may help understanding the current status and trends of emissions (any potentially relevant pollutant), air quality, or impacts has been considered. To present a synthetic and harmonized view of the status of the five cities analysed, we rely on the combination of: (i) fully comparable, globally available data with (ii) any other local or national specific data and references that may be relevant to illustrate the relative stages of progress towards a comprehensive air quality management strategy. The information discussed in the following section consists of:

- Emissions and air quality: estimated emissions of organic carbon (OC) and black carbon (BC) emissions elaborated from the most updated and comprehensive regional inventory available for Africa (Keita et al., 2018), available at the Emissions of atmospheric Compounds and Compilation of Ancillary Data (ECCAD) website. The DACCIWA (Dynamics-Aerosol-Chemistry-Cloud Interactions in West Africa) inventory provides emission fields at 0.125° x 0.125°

spatial resolution (broadly 10 km horizontal resolution) for the entire African continent from 1990 to 2015. It extends and improves the only previous regional inventory available (Liousse et al., 2014) by establishing an African database on fuel consumption and new emission factor measurements, including biofuel and fossil fuel emissions as well as open waste burning emissions. Primary (directly emitted) anthropogenic particles are largely made of OC and BC (Bond et al., 2004) and therefore, they constitute a good proxy of combustion-related PM emissions. The reference inventory provides emission estimates from the main anthropogenic sources (energy, industries, residential, transportation, waste, and other sectors). This is complemented with city-specific information and references including technical reports, journal papers, plans and strategies related to air quality measurements from permanent monitoring networks or ad hoc experimental monitoring campaigns, emission inventories, air quality modelling activities, source apportionment studies, health impact assessments, cost-benefit analyses, etc.

- Summary of the response given to the air quality issues portrayed in the previous point under a common structure (UNEP, 2016): relevant initiatives regarding 1) air quality standards, regulations and plans, 2) vehicle emissions, 3) public and non-motorized transport, 4) industrial emissions, 5) open burning of waste, and 6) indoor air pollution are briefly summarized. The main interest is to discuss the lessons learned from recent actions as well as ongoing initiatives that may be relevant for future planning or inspiring other urban areas with similar status. The discussion for each city includes national (even international) instruments and then those that are primarily driven by regional or local administrations and try to make links among them since very often they are tightly interrelated given the multi-level nature of air quality governance.

The Conclusions section is structured following the six framework areas proposed in the Guidance Framework for Better Air Quality in Asian Cities developed by UN Environment and Clean Air Asia (Clean Air Asia, 2016).

Discussion

Emissions and air quality issues

The sources of emissions vary considerably from city to city. While the five case study cities share some commonalities, they show considerable variations (Figure 4). The residential sector appears to be the main source of emissions of organic carbon for four of the five cities (Dakar, Accra, Cape Town and Nairobi); however, in the megacity of Cairo, the waste sector predominates. For black carbon, the dual importance of the waste and residential sectors appears evident in Dakar and Cape Town. In Cairo, nonetheless, the waste sector dominates BC emissions, with the residential sector assuming little importance, while the reverse is true in Nairobi. The reasons for such differences are complex but are partly related to the varying levels of socio-economic development achieved in their respective countries. The main implication of these differences is that, when addressing air quality at the city level, customized, well-targeted clean air action plans should be specifically designed.

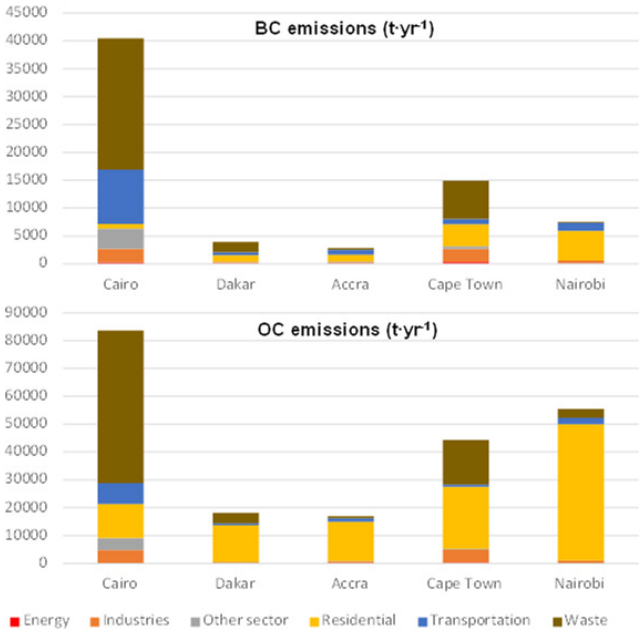


Figure 4: *PM emissions in the five target cities (t) in 2015. Elaborated from DACCIWA gridded inventory (Keita et al., 2018). BC = black carbon, OC = organic carbon*

Emission rates shown in Figure 4 originate from the overlapping of the DACCIWA gridded inventory in a 120x120 km geographical domain for each city. Total emissions in those reference areas are shown in Figure 5 (black carbon) and Figure 6 (organic carbon). The maps show the road network according to the information published by SEDAC from Global Roads Open Access Data Set, Version 1 (gROADSv1) (Columbia University and University of Georgia, 2013) and administrative boundaries, taken from the GADM database version 3.4 (GADM, 2018).

Accra

According to air quality data from Accra's monitoring network (EPA, 2018), annual average concentration in Accra in the 2006-2017 period ranged from 55-78 μg/m^3, and 85-130 μg/m^3 for $PM_{2.5}$ and PM_{10} respectively, without a clear trend (Appoh, 2018). According to Ghanese EPA (EPA, 2016), 90.2% of PM_{10} measurements from roadside monitoring sites levels recorded in 2015 exceeded the EPA guideline for that particulate matter (70 μg/m^3 for the 24-h mean). Although the network suffers from frequent breakdowns and data availability is below 50% (Schwela, 2012), $PM_{2.5}$ present a remarkable monthly variability reaching up to 209 μg/m^3 in December 2015 (EPA, 2018). Such high pollution episodes during the dry season have been related to the Harmattan wind, a natural phenomenon that contributes to the load of particulate matter travelling south from the Sahara Desert (Zhou et al., 2013).

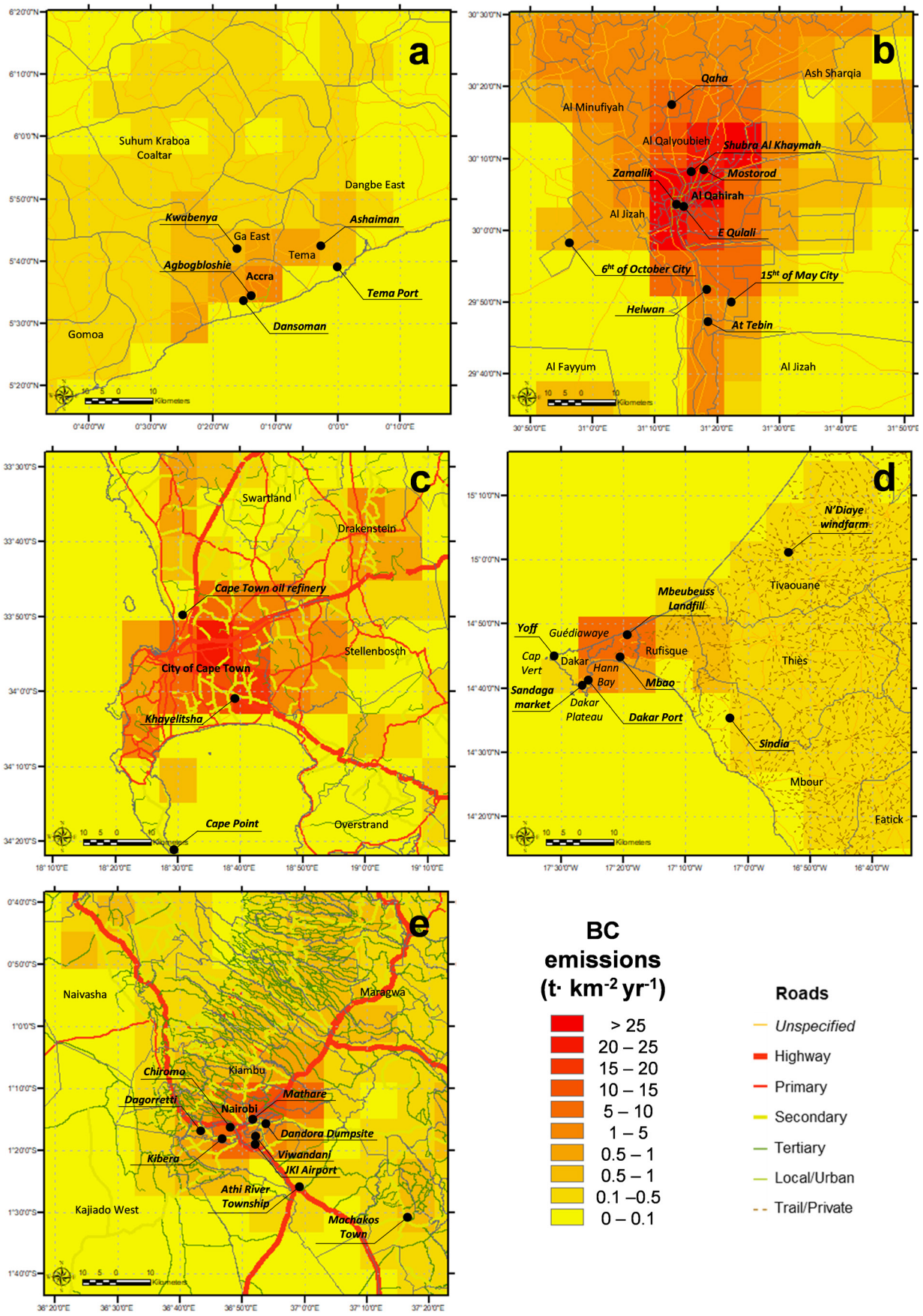


Figure 5: *Black carbon emissions in the five target cities (t) in 2015, a) Accra, b) Cairo, c) Cape Town, d) Dakar and e) Nairobi. Elaborated from DACCIWA gridded inventory (Keita et al., 2018)*

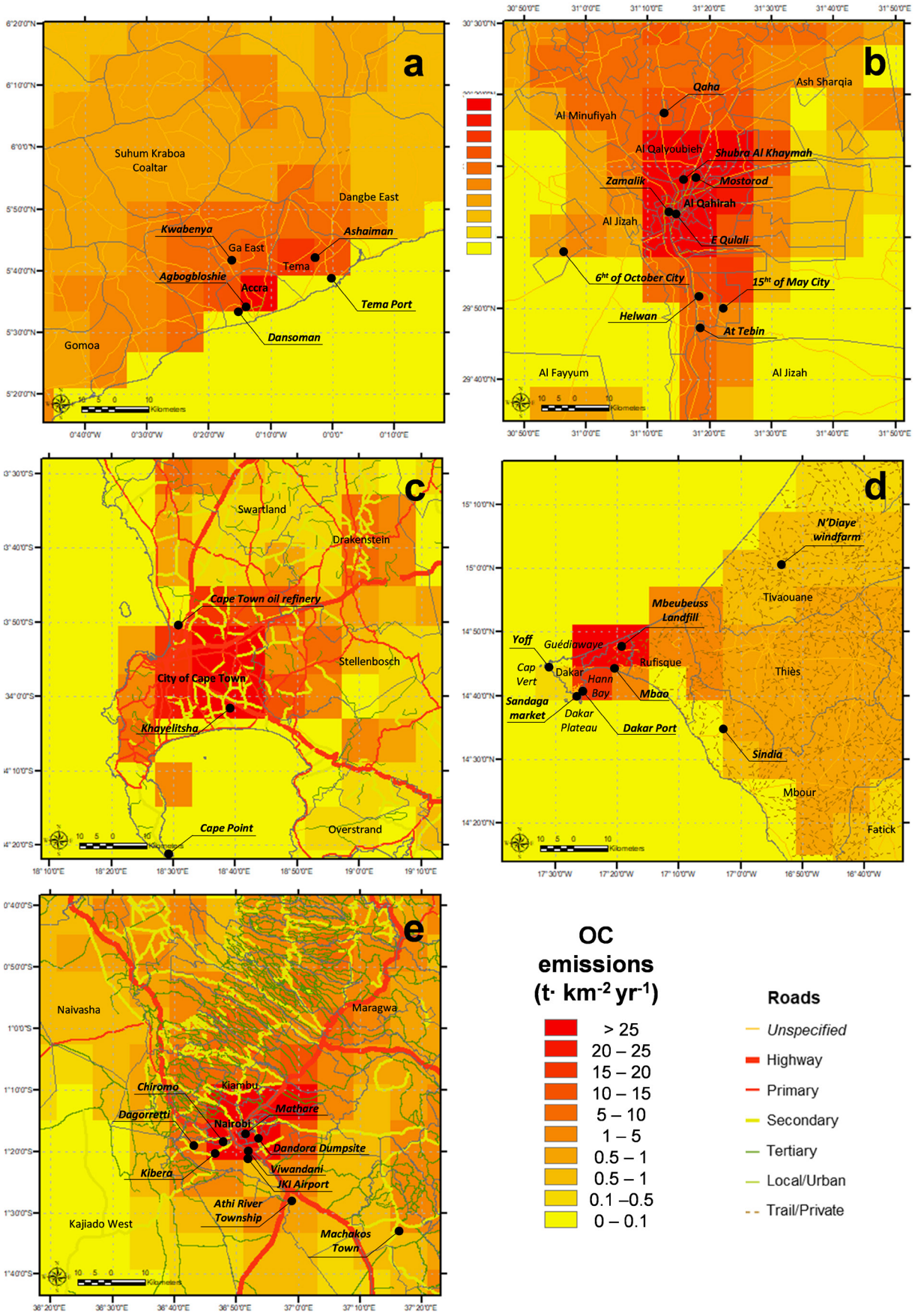


Figure 6: *Organic carbon emissions in the five target cities (t) in 2015, a) Accra, b) Cairo, c) Cape Town, d) Dakar and e) Nairobi. Elaborated from DACCIWA gridded inventory (Keita et al., 2018)*

Besides permanent monitoring schemes, air quality in Accra has been addressed by a number of limited time-frame studies and ad-hoc campaigns. Zhou et al. (2013) found that sea salt (up to 13.9 µg/m^3) and mostly mineral dust, were large contributors to ambient PM concentrations. During Harmattan wind episodes, crustal particles accounted for 55 µg/m^3 (37%) of $PM_{2.5}$ mass and 128 µg/m^3 (42%) of PM_{10} mass. This is consistent with the study of Aboh et al. (2009) that found PM crustal elements (dust) fraction to increase more than tenfold during the Harmattan period in Kwabenya, a suburb of Accra that is frequently exposed to dust from the Sahara–Sahel region during the dry season. Outside Harmattan, road traffic was found to be one of the largest contributors to PM, being responsible for approximately 12-33% of PM concentration. Biomass combustion was responsible for between 10.6 and 21.3 µg/m^3 of particulate matter in the study areas, affecting most of the poorest neighbourhoods.

Other relevant sources identified were vehicle emissions, tire and brake wear, road dust, and solid waste burning. While total PM_{10} concentration reached values as high as 179 µg/m^3, BC contents were only four µg/m^3 during 2006-2007 period analysed, highlighting the huge influence of natural PM. Arku et al. (2015) reported an average exposure of 56 µg/m^3 to $PM_{2.5}$ for school children in Accra (more than double the average exposure levels in the US or Europe) although there were large variations, from less than 10 µg/m^3 to more than 150 µg/m^3. They identified residential emissions (wood and charcoal burning for cooking) and dust emissions from unpaved roads as the most influential sources of exposure. Exposure to emissions from household cooking and garbage burning has been found to impact even the unborn. According to Amegah et al. (2012), these pollution sources significantly reduce birth weight in Accra. Studies carried out in cities like Kumasi (Bandowe et al., 2019) or Ashaiman (Ofosu et al., 2012) confirmed the influence of road traffic, including PM resuspension, particularly in low-income neighbourhoods. Industry (e.g. Tema Oil Refinery) emits considerable amounts of PM and other pollutants (NO_X, SO_2) and has a clear impact in closer urban areas such as the Tema Metropolis (Amoatey et al., 2018; Amoatey et al., 2019).

Cairo

As in many megacities in the world, air pollution has been a chronic problem in Greater Cairo. The first air quality measurements (SO_2 and smoke) were carried out by the Ministry of Health (MoH) in 1973 (Nasralla, 2001). In 1999, the Egyptian Environmental Affairs Agency (EEAA) initiated a more systematic air quality monitoring strategy towards the current National Air Quality Network of Egypt that includes 24 real-time continuous air monitoring and 25 sampling stations in the Greater Cairo Metropolitan Area (GCMA).

According to the PM_{10} levels recorded, air quality in Cairo met Egyptian air quality standards for that pollutant 72% of the time while $PM_{2.5}$ levels of 60-99 µg/m^3 have been reported for the city as an annual mean (EEAA, 2018). A significant drop between 1999-2009 has been observed for all pollutants, except for O_3 (WB, 2013). This trend has been observed in other urban areas where reduced NO_X emissions reductions from traffic lead to higher O_3 concentration levels due to non-linear photochemical processes (Saiz-Lopez et al., 2017).

Despite the declining trend, the rapid population growth (by more than 3 million from 1999 to 2010) has led to a net increase in population exposure to air pollution. Wheida et al. (2018) found an average PM_{10} of 155 (±35) µg/m^3 during the 2010-2015 period in GCMA, although large concentration gradients exist within the megacity. Similarly, average $PM_{2.5}$ concentrations varied from 50 to more than 100 µg/m^3. Both PM levels were affected by massive wind-blown desert dust since the city is within the Sahara region. With the exception of a few traffic hot spots, NO_2 concentrations were found to be below 40 µg/m^3, defined as an air quality standard in Egypt in accordance with the 2005 WHO guidelines (WHO, 2006). Some authors (Ndour et al., 2008) suggest that high natural dust loads in urban atmospheres may enhance the photocatalytic uptake of NO_2. Despite the increase of tropospheric O_3, further recent analyses confirm that current air quality issues in Cairo primarily concern PM (Mostafa and Zakey, 2018).

According to WB (2013), approximately 35–55% and 25-35% of the total concentration of PM_{10} is attributable to airborne geological material during the summer and winter, respectively. This is consistent with the findings of Boman et al. (2013) that also found a significant influence of marine aerosols. Earlier, Favez et al. (2008) observed maximum dust concentrations in spring and winter due to the higher frequency of dust storms, but noted high dust contribution throughout the year (around 50 µg/m^3). In addition, they reported that natural dust interacting with NO_X and SO_2 emissions adds secondary compounds to the atmospheric aerosol. Similarly, Hassanien and Abdel-Latif (2008) reported significant amounts of toxic polycyclic aromatic hydrocarbons (PAH), in road dust samples collected across Cairo.

An episodic deterioration in air quality through the so-called Black Cloud phenomenon is observed every October and November (WB, 2013) due to rice straw burning in the governorates around Cairo coinciding with typical autumn thermal inversions (Marey et al., 2010; Zakey et al., 2008). During these events, the aerosols originating from the burning of agricultural residue account for 12% of BC and up to 50% of OC (soluble fraction) (Favez et al., 2008). On the other hand, Mahmoud et al. (2008) concluded that traffic was the majority source of BC in Cairo during the daytime, even in autumn when biomass burning takes place in the Nile Delta.

Cape Town

The City of Cape Town (CCT) and the Department of Environmental Affairs and Development Planning (DEA&DP) operate an air quality network that measures criteria pollutants based on reference analytical techniques in accordance with the US EPA standardized methods (DEA&DP, 2019a). All the information is submitted to the South African Air Quality Information System (SAAQIS) where nearly real-time air quality

information can be visualized and downloaded (https://saaqis.environment.gov.za).

Historically, SO_2 and PM_{10} have been the air pollutants of concern in South Africa (UNEP, 2016). Recent data show a general decrease in SO_2 ambient concentration levels, while there is not a clear trend regarding PM_{10} and increasing exceedances of O_3 (DEA&DP, 2019a). Currently air quality in the Western Cape Province is generally good, below the South African National Ambient Air Quality Standards (SANS). However, some hot spots, such as the Khayelitsha informal settlement, remain. The eight-hour mean SANS for ozone was exceeded on 19 occasions in 2018 at Khayelitsha, with values up to 175 µg/m^3. High PM levels have been usually reported due to residential wood burning, waste burning and dust from unpaved roads, among other sources in this area (DEA&DP, 2016). A few, exceptionally high values (up to 387 µg/m^3 in 1 hour) have been linked to wildfires in the surroundings of the city as well as windblown dust episodes responsible for daily PM_{10} levels up to 82 µg/m^3 (DEA, 2012). Of note, no PM data from Khayelitsha station were available for the year 2018 due to insufficient measurements. Such information gaps are common throughout the country with less than 40% of the 94 monitors included in the national network meeting the minimum 70% data recovery rate for PM measurements (DEA, 2019) due to aging infrastructure and insufficient funding (DEA&DP, 2019a).

Wicking-Baird et al., (1997) studied the causes of pollution in the city using the Chemical Mass Balance (CMB) model (Watson et al., 1990) and concluded that visibility issues during brown haze episodes were mainly related to road traffic, responsible for 65% of total $PM_{2.5}$ mass. Other important sources of pollutants were combustion (14%), boilers (6%) and natural PM sources (5%). This situation has not changed significantly according to more recent studies that identified traffic, domestic burning of fuels, industry and waste burning as the major pollution sources in Cape Town (Mbow-Diokhane, 2019; Keen and Altieri, 2016).

Measurements of relevant anthropogenic VOC concentrations in Cape Town suggest that urban background concentrations of these pollutants are dominated by traffic or industrial sources in different areas of the city (Kuyper et al., 2020). Mumm et al. (2017) concluded that low socioeconomic districts generally experienced higher levels of air pollution. Similarly, Hersey et al. (2015) identified low-income townships and informal settlements as the worst-polluted areas in South Africa. This supports the claim made by Keen and Altieri (2016) that the largest health benefits of improving air quality in the city would be found in the Khayelitsha. Hersey et al. (2015) identified biomass burning (veld fires) and domestic burning in informal settlements as significant sources of PM_{10}. These authors found that dust enhancement was particularly high in May-July; contributions from the nearby Namib Desert may make up to 25% of total PM_{10}. Sea salt represented around 20% of total PM_{10} in Cape Town.

Higher-resolution modelling exercises confirm that high PM_{10} episodes in Cape Town are related to air masses travelling over major dust source regions such as the Kalahari or Namib Deserts (Molepo et al., 2019). The growing influence of windblown dust and wildfires on PM_{10} levels in Cape Town has been also linked to increasingly severe droughts in the Western Cape (DEA&DP, 2019a). In addition to negative air quality ramifications, climate change has been identified as a major health threat in South Africa, especially for vulnerable groups (Chersich et al., 2018). Nonetheless, greenhouse gas emissions in the country have raised by 20% in the 2000-2010 period (DEA, 2016) because of the coal dependency in the country (Klausbruckner et al., 2016).

Dakar

Senegal is the only country in West Africa that has set up a continuous air quality monitoring network (5 monitors) and created the Centre for Air Quality Management (Centre de Gestion de la Qualité de l'Air; CGQA) within the Urban Mobility Improvement Programme (PAMU). (Mbow-Diokhane, 2019). Frequent power failures (Cissokho & Seck, 2013) combined with specific technical and maintenance issues or the network (CGQA, 2018) result in low data availability. Nonetheless, data captured is made publicly available by CGQA (https://www.air-dakar.org) and through annual reports regarding air quality in Dakar (CGQA, 2018).

These reports suggest that PM is the most problematic pollutant, with annual average urban background levels as high as 182 µg/m^3. Although PM is strongly affected by marine aerosols and windblown dust from the Sahara, air quality can be classified as good or moderate 52% and 34% of the time, respectively. Concentration peaks have been associated with the westward propagation of Harmattan pulses loaded with mineral dust from the desert (Rodríguez et al., 2019). On average, levels of PM_{10} are twice as high during the nine driest months than in the three wet months. AQI interannual evolution does not exhibit a clear pattern, and data deficiencies (average data availability range from 5% to 67% depending on the monitoring station) make it difficult to further identify pollution trends.

There are several studies that delve further into the pollution levels and causes in Dakar. Ndong Ba et al. (2019a) reported ambient PM_{10} average concentrations around 100 µg/m^3 during the dry season in the 2017-2018 period. Previous campaigns revealed average $PM_{2.5}$ levels of 87 µg/m^3 in a traffic location in Dakar, much higher than the concentrations measured in a rural area some 40 km away from the city (32 µg/m^3) (Ndong Ba et al., 2019b). The coarser fraction of suspended PM, however, was quite constant in both locations (57 and 56 µg/m^3 respectively), probably due to natural dust contributions. A comparison with the results from Dieme et al. (2012) suggests an increase of $PM_{2.5}$ levels in Dakar and its surroundings during recent years due to an increasing influence of traffic and of biomass burning. Several studies have identified traffic as the main source of air pollution in the city (Schwela, 2012; Sarr et al., 2018) with clear health implications, especially for individuals working along major streets (Ndong Ba et al., 2019a). Adon et al., 2016 monitored the concentration of the main gaseous pollutants in Dakar for two years (2008 and 2009) using passive samplers

and found average NO_2 concentration levels of 31.7 ppb at a traffic site – three-fold higher than the corresponding value for urban background areas. In addition to the spatial variability, this study found that NO_2 concentrations over the dry season were 1.5 times higher than those of the wet period (three times higher for SO_2). The authors point out that besides favourable meteorological conditions, the wet period in Dakar coincides with a lower activity period (school vacation) and consequently, less emissions.

These findings are consistent with those of Doumbia et al. (2012) regarding particulate matter. Similarly, they identified traffic as the main pollution source, with an average contribution of 87% and 89% to total BC during the dry and wet seasons, respectively. The remaining 13%-11% was apportioned to biomass burning emissions. Besides anthropogenic sources, the contribution of natural windblown dust from the Sahara to PM levels in Dakar is well documented (Oluleye & Folorunsho, 2019). During Saharan dust episodes, PM_{10} and $PM_{2.5}$ ambient concentrations in Dakar can reach values in excess of 800 µg/m³ and 350 µg/m³, respectively (Jenkins & Diokhane, 2017). Although natural dust dominates PM concentration in Senegal, the largest burden of asthma and bronchitis in the country is found in Dakar (Toure et al., 2019), suggesting that urban emission sources also play a vital role on human health, and synergistic negative effects can be significantly diminished by cutting down anthropogenic emissions.

Nairobi

Despite the lack of a comprehensive, consistent and transparent city-wide monitoring strategy, some monitoring campaigns and pilot projects have been undertaken in Nairobi (Nthusi, 2017), including the use of novel sensor technology (Kumar et al., 2015). Despite technical limitations and issues related to data quality (WHO, 2018), such initiatives can contribute to increase awareness and thus, support for more effective air quality regulation (Ngo et al., 2017). This technology was used by de Souza et al. (2017) to monitor $PM_{2.5}$ and PM_{10} over an eight-month period (May 2016 to January 2017). They reported concentrations between 11 and 23 µg/m³ and 26 and 59 µg/m³, respectively. The NO_2 observed level oscillated between 8 and 12 ppb while the highest mean level of SO_2 was recorded at Viwandani (40 ppb), an informal settlement in the industrial area of Nairobi. Although these instruments were not calibrated with reference air quality monitors, the study suggests that pollution levels are usually higher in Kibera and Viwandani slums.

This is consistent with previous studies (Gulis et al., 2004; Egondi et al., 2016) that highlight that air quality issues are particularly serious in deprived neighbourhoods of Nairobi, a pattern also found in other African urban areas (Dionisio et al., 2010). Pope et al. (2018) also used low-cost sensors (calibrated against gravimetric measurements, in this case) to monitor $PM_{2.5}$ in Nairobi in February-March 2017 and found an average concentration of 36.6 µg/m³ at a roadside; considerably higher than the 24.8 µg/m³ observed in an urban background site and the 8.8 µg/m³ measured in a rural location more than 100 km away from the city. The relevance of traffic to PM levels in the city was also pointed out by Kinney et al. (2011). They estimated a 24-hour average $PM_{2.5}$ personal exposure of 45-85 µg/m³ along busy roads in Nairobi.

Similarly, van Vliet and Kinney (2017) estimated that roadway concentrations of $PM_{2.5}$ in Nairobi were approximately 20-fold higher than urban background levels (around 20 µg/m³). Gatari et al. (2019) found that BC represented 34-56% of the total $PM_{2.5}$ mass during a one-month monitoring campaign (July 2019), with levels reaching 24-hour average values as high as 43 µg/m³. This demonstrates that road traffic emissions are the dominant source of airborne $PM_{2.5}$ in curb-side locations. The proximity to traffic emissions was also found to be a key factor by Ngo et al. (2015). They found that a large fraction of $PM_{2.5}$ consisted of BC (vehicle smoke) due to poor engine maintenance of Nairobi's fleet. They further found that women in the Mathare slum experienced exposure to $PM_{2.5}$ at levels similar to roadside mechanics and street vendors (hawkers), since they spend a substantial amount of time near roadways.

This is again, directly related to environmental justice since poor pedestrians who cannot afford motorized transport are forced to walk long distances near traffic emissions (Klopp, 2012). Egondi et al. (2016) also demonstrated that residents in slums are continuously exposed to $PM_{2.5}$ levels exceeding hazardous levels according to the WHO guidelines, reaching average concentrations up to 166.4 µg/m³, considerably higher values than those reported in other studies, even at curbside sites (Pope et al., 2018; Kinney et al., 2011). High concentrations in poor neighbourhoods in Nairobi may be attributed to very local sources, such as street markets that increase traffic as well as street cooking using biomass fuels.

The strong seasonality of pollution that peaks during the dry season (July in particular) suggests that PM levels may also be affected by dust transport and PM resuspension processes. The longest period of recorded ambient PM concentration in Nairobi is that reported by Gaita et al. (2014) who found average $PM_{2.5}$ urban background and suburban pollution levels of 21 (± 9.5) µg/m³ and 13 (± 7.3) µg/m³, respectively (at 17 m and ten m above the ground level). This study identified traffic as the largest contributor to airborne PM (39%). Despite the large emission share attributed to regional inventories (Figure 5 and Figure 6), waste burning, open fires and charcoal burning, typical of low-income households were responsible for six percent of total $PM_{2.5}$. A very high contribution of mineral dust, presumably from unpaved roads (35% on average; up to 74% of $PM_{2.5}$ in Nairobi during the dry season) was also highlighted.

Air quality management and responses

Accra

According to Appoh & Terry (2018), air quality management in Ghana can be considered advanced and the recent evolution of the country can be considered as an instructive case study of air quality management planning in a sub-Saharan Africa

context. During recent decades, the country has developed an organizational structure and regulatory framework, including National Ambient Air Quality Guidelines (24-hour PM_{10} mean ≤ 70 µg/m³). Ghana EPA expected the full implementation of AQ standards by June 2019 (EPA, 2018), although they are not enforced so far. Together with partners, launched a series of initiatives, many of them targeting Accra.

The development of the air quality monitoring network in Accra sets up a good example of the potential benefits of international collaboration, in this case in the context of the Air Quality Monitoring Capacity Building Project in cooperation with the United States Environmental Protection Agency (USEPA), the United States Agency for International Development (USAID) and UNEP. In Accra, we also see the fledgling work of the WHO's Urban Health Initiative to raise the necessary awareness through community-based actions and campaigns.

The consolidation of a robust democracy in Ghana provides the enabling environment (appropriate institutional, legal, and regulatory structures) for the successful development and implementation of air quality management systems. Building on these technical capacities and multi-level governance scheme, Ghana's Environmental Protection Agency (EPA) launched the Greater Accra Metropolitan Areas Air Quality Management Plan in 2018 (EPA, 2018). The first comprehensive air quality strategy for the Greater Accra Metropolitan Area intends to deal with a number of potential sources of air pollution, especially road traffic, which is becoming the main source of air pollution in Accra (Naidja et al., 2018). The plan establishes links to other relevant environmental strategies that can provide the framework for the promotion of recent successful sector-specific measures implemented in Accra, such as the Bus Rapid Transit System (BRT) project (EPA, 2017) or the Greater Accra Scrap Dealers Association (GASDA) in Agbogbloshie (Pure Earth, 2015). The plan includes specific indicators, time frames, responsibilities and collaborations needed that intend to comply with the national ambient air quality standards. The plan also contemplates the involvement of stakeholders and the population. More importantly, it identifies the capacity gaps that should be addressed for a successful implementation. Other initiatives to install new improved clean stoves and to promote the use of liquefied petroleum gas (LPG) have demonstrated a high potential to reduce personal exposure to $PM_{2.5}$ also in rural areas (Piedrahita et al., 2017).

Cairo

During the last decades, Cairo has faced virtually all of the underlying drivers that contribute to air quality degradation in megacities: a rapidly growing population and urban expansion, uncontrolled rural-to-urban migration, inadequate land use planning, as well as the unsustainable growth of the industrial zones and other heavy infrastructure sectors in and around the city. However, most ministries of the Egyptian government have established an environmental unit or department to deal with air pollution in their respective sectors (WB, 2013). As a result, considerable progress has been made in tackling air quality issues in the GCMA. Egypt has been able to channel international collaboration (DANIDA, USAID) to develop a full-scale urban air quality monitoring system in the Greater Cairo Metropolitan Area (Nasralla, 2011).

Projects such as EIMP or CAIP have been instrumental to identify the main issues and orientate national policies. In addition to the development of an emission inventory (WB, 2013), Cairo is the only city in this paper that has done a strategic analysis of future emission scenarios. Health impact assessment studies revealed that 4 550 premature deaths would be avoided every year in the GCMA if the PM_{10} Egyptian standard (annual average concentration of 70 µg/m³) was met (Wheida et al., 2018), although other authors (Safar and Labib, 2010) suggest that the natural contribution to PM_{10} background will prevent Cairo from ever attaining this standard. Standards for NO_2, SO_2 and O_3 are relatively stringent for the region, comparable to the European ones. In addition, it has been estimated that air pollution represents 3 to 6% of the GDP (UNEP, 2004), making a clear economic case for emission abatement.

Conceptually, a multi-level governance scheme matches better the multi-scale reality of air quality. However, Egypt demonstrates that a strongly centralized management system may be more effective in the initial stages. Despite a lack of a specific plan per se, the massive introduction of natural gas in the domestic, industrial (EEAA, 2004), power generation (WB, 2013), and traffic sectors (Thomas, 2016), along with the development of large-scale engineering projects (Korkor, 2014) in Egypt has allowed improving air quality in Cairo despite intense population and traffic growth. Up to nine ministries have been actively developing legal and institutional mechanisms as well as programs and projects with international development cooperation partners to improve air quality in their respective sectors, often structured as Public Private Partnerships (WB, 2013). Some of them, such as the valorisation of agricultural waste in cement clinks have demonstrated the benefits of cross-sectoral measures to prevent pollution, improve the local economy and report climate co-benefits.

Cape Town

Air quality has drawn the attention of policymakers in South Africa for nearly two decades, making the country a reference point regarding air quality management in sub-Saharan Africa (Schwela, 2012). In addition to the national framework, the Provincial Government's 2016 Air Quality Management Plan (DEA&DP, 2016) is the current regional strategy that aims at improving air quality.

The South African air quality management system is arguably the most advanced one in the African continent. The administration decentralization in the country has allowed a multi-level governance scheme for air quality action, with a nationally set policy and regulatory framework (Republic of South Africa, 2013), and enforcement and compliance conducted by regional and local governments. The national air quality standards (SANS) are comprehensive and stringent (DEA, 2012), comparable to those of Europe (EC, 2008), currently under review.

The country has also made efforts towards effective ambient air quality monitoring and communication through the South African Air Quality Information System (SAAQIS) or the National Association for Clean Air (NACA) dissemination activities (Mbow-Diokhane, 2019). A number of projects have conducted meaningful modelling and health assessment studies (e.g. Muchapondwa, 2010). While this is the ideal situation for the implementation of mature air quality management systems, the experiences from Cape Town also inform us about the difficulties and potential shortcomings of this approach (Naiker et al., 2012). The Western Cape Province and Cape Town Municipality have been very active in developing regulations, operational frameworks, air quality management plans, climate strategies and mobility plans among others in the last two decades (TDA, 2005; CCT, 2007; CCT, 2011; DEA&DP, 2014; TDA, 2017; City of Cape Town / 100 Resilient Cities, 2019 among others).

However, progress regarding air quality is still lagging behind the strategic goals (Tshehla and Wright, 2019). Despite fragmented planning, this may be related to the lack of concretion of the majority of these strategies, lack of political will as well as insufficient funding (DEA&DP, 2019). Even in a city of relative prosperity on the continent, mobilizing the resources for all these activities is an ongoing challenge. In addition to the challenges of administrative transitions and a complex regulatory framework, Cape Town faces the problem of persistent poverty and social inequity that shapes its physical reality and results in an unbalanced burden of air pollution-related disease. It has been suggested that previous plans did not address the problems surrounding poverty as the underlying cause of environmental deterioration (Language et al., 2016).

Dakar

In the review provided by Schwela (2012), Senegal was depicted as a country at an intermediate stage of air quality management maturity. Among other gaps, Senegal was noted as lacking a comprehensive air quality management strategy while multi-level governance was considered blurred; a specific urban air quality plan for Dakar was also lacking. Being a country particularly vulnerable to the effects of climate change (Zamudio and Terton, 2016), current action plans pay more attention to Greenhouse Gases (GHG) mitigation (MEDD, 2015), with a focus on the promotion of renewable energies such as the 159 MW N'Diaye windfarm nearby Dakar (IISD, 2019a) and adaptation, such as the Dakar Resilience Strategy (Ville de Dakar/100 Resilient Cities, 2016).

However, significant advances have been made recently. Senegal has one of the oldest and more stable democracies in Africa and recent reforms favoured the decentralization and strengthening of the administration. This provides the enabling environment (appropriate institutional, legal, and regulatory structures) for the successful development and implementation of a comprehensive air quality management system in the future (WB, 2015). Presently, Senegal has a comprehensive and well-structured strategy to keep strengthening its administration and to foster collaboration with private stakeholders and the international community under the Emerging Senegal Plan (ESP), and overarching framework for the structural transformation of the country and the improvement of governance and living conditions in Senegal by 2035 (MEPC, 2014; MEPC, 2018). This instrument encompasses 27 key infrastructure projects, such as the development of the Bus Rapid Transit (BRT) system in Dakar, an 18.3 km separate corridor (currently serving 144 buses), with23 BRT stations and three terminals covering Dakar Plateau and Guédiawaye (IISD, 2019b). At the local level, the Dakar Master Plan (MURHLE & JICA, 2016) is another key instrument to improve living conditions and it is also expected to provide large air quality co-benefits. Previous strategies, mainly the Urban Mobility Improvement Program (PAMU) have contributed to tackling air quality issues and have been key for the development of incipient monitoring capabilities in the city (WB, 2009).

Despite considerable technical limitations that seriously hinder data availability, Dakar's air quality network can be pointed out as an example of transparency and environmental data. Still, Dakar is aware of the pressing need to further control emissions from road traffic (Ndong Ba et al., 2019a; Adon et al., 2018; Sylla et al., 2018). Furthermore, officials understand that the transport and waste management problems (Yaah, 2018) cannot be seen exclusively from a technological point of view but must also be addressed from an urban planning perspective, since they are linked to the rapid expansion of the city.

Nairobi

Kenya lacks a comprehensive urban air quality management programme. The air quality management responsibility is with the Ministry of Environment and the National Environment Management Authority (NEMA), in collaboration with other partners. Despite weaknesses detected in terms of enforcement and compliance practices (Schwela, 2012), there are some recent actions undertaken by the national government as well as local initiatives that are expected to benefit air quality in Nairobi.

Until very recently, most efforts to address air pollution in Nairobi were undertaken at the national level (MoEF, 2014; MoEP, 2016). A profuse and fragmented environmental regulation in Kenya has hindered the effective linkage with operational programs and strategies at the local level, making air quality management difficult (Shilenje, 2014). On the other hand, the paucity of administrative capacity, weakness of institutions and high corruption levels (Transparency International, 2019) have been identified as essential factors for the lack of progress. Lack of public awareness (Ngo et al., 2017) and very limited monitoring capacity are also major barriers to developing a comprehensive, multi-level air quality strategy.

Nairobi has now the necessary regulatory framework and instruments to tackle air quality issues in the city. The Nairobi Integrated Urban Development Master Plan (NIUPLAN) (NCC & JICA, 2014) includes a comprehensive and accurate diagnosis of the environmental challenges in the city and establishes the general strategy for a balanced and sustainable urban

development, in line with the general national strategy given reflected in Kenya's Vision 2030 (Government of Kenya, 2018). More recently, Nairobi launched the first Air Quality Plan in Kenya, the Air Quality Plan 2019-2023, along with the Nairobi City County Air Quality Policy to tackle air quality issues specifically (NCC & UNEP, 2019).

Conclusions and recommendations

The lack of consistent information prevents from a formal comparison of the status of air pollution and policies in Africa. Nonetheless, the findings of air quality management in the 5 case studies included in this research are summarized in Table 2 with the aim of assessing whether interventions are adequately targeted to air quality issues and identifying action needed within the 6 framework areas Clean Air Asia, (2016). More research is needed to fully characterize the drivers of air pollution action in the region and formulate evidence-based plans to improve air quality in affected cities in coordination with climate policies and general development strategies. Nonetheless, some general conclusions and recommendation can be drawn from the 5 major cities scrutinized based on the available information discussed in the previous section. More specific recommendations and particularly interesting interventions for each city are discussed in UN Habitat (2023).

Air quality standards and monitoring

- Air quality standards are typically set up at the national level and most of the countries have moved from air quality guidelines to formal and rather comprehensive air quality standards. As much as possible, $PM_{2.5}$ standards should be considered a priority. However, more than the lack of air quality standards the problem in Africa seems to be the lack of effective enforcement. Consequently, providing the enable conditions and resources for actual enforcement should be encouraged.
- Air quality data in the continent is very limited. Even where monitoring stations exist, maintenance or reliable power supply are lacking, and data is therefore of poor quality or consistency. International collaboration seems to be key for the initial development of monitoring capabilities, but it is essential to secure adequate funding to guarantee the sustainability of urban air quality monitoring networks.
- New technologies based on low-cost sensors may help improving monitoring capabilities but significant investments for capacity building and maintenance and closer collaboration with local universities and research institutes are still needed.

Emissions inventories and modelling

- While national GHG inventories are usually available, city-scale emission inventories are mostly missing and should be developed to prioritise action on the variety of emissions sources affecting urban air quality and to assess the efficiency of plans and measures.
- Some cities have been able to channel international cooperation into the development of incipient modelling capabilities. The literature discussed throughout this work illustrate relatively simple modelling exercises that may be useful as a first step to understand local air quality issues for specific sources or neighbourhoods. Further support and scientific collaboration are needed to move towards more complex tools to deal with emerging air quality issues such as tropospheric ozone or secondary aerosols. A stronger network of researchers in Africa would be instrumental to foster scientific knowledge and capabilities in this area.

Health and other impacts

- While some general estimates exist, more localised epidemiological and cost-benefit studies are needed for a better understanding of the health impacts of poor air quality in African cities, prioritize measures and make a stronger case for action. Health impact models based on assumptions drawn from higher income countries may underestimate impacts in Africa due to synergies with weather risks, communicable diseases and food security issues. Meeting the WHO air quality guidelines, or even the national air quality standards, would bring extensive health benefits, especially for the most vulnerable communities.
- The evidence offered by the 5 case studies point out that low-income citizens in African cities may disproportionally bear the impacts of air pollution. Emissions from indoor dirty fuels combustion, waste open burning and unpaved roads among others create pollution hot spots in informal settlements. Exposure to air pollution from traffic is particularly high among the poor as well, since they mostly rely on non-motorized transport routes along heavily polluted environments. Tackling these issues will contribute not only to improve air quality but also environmental justice.

Communication

- Education and public awareness towards air pollution processes and air quality health effects is alarmingly low in Africa and it may be a major hindrance to implement emission abatement measures. Any intervention or strategy needs to emphasise communication actions for a successful result.
- Open access to air quality data, emission inventories and health indicators is an essential need to involve policy makers and to engage relevant stakeholders and the general public in improving air quality. Channels for communicating the impacts of poor air quality are also essential since only an informed citizenry can demand additional measures to preserve public health.

Clean Air Action Plans

- Air quality action can be driven by dedicated Air Quality Action Plans but also by integrating air pollution actions into national and city development plans that guide infrastructure development. The choice of adopting either approach will depend on the local context. While both approaches will yield positive effects, it is important to explicitly consider emissions and exposure in all the plans

and strategies, including urban planning instruments, as an effective way to maximize health benefits.

- Ad-hoc experimental campaigns and research projects highlight that, despite the diversity of city-specific conditions, PM-related pollution is common factor to all of them and should be prioritized.
- For countries and cities in which data and financial resources are lacking, a strategic focus on sector-targeted, short-term actions to reduce air pollution may be a wise investment, even before the creation of long-term monitoring and management capacity. Effective banning of open burning, accompanied by a transition to a more responsible recycling local industry, or improved fuel standards may represent illustrative 'no regrets' actions.
- Road traffic is a key sector in all the cities analysed. Stronger emission inspection schemes and interventions to promote clean, affordable and efficient public transport must be prioritized. In this context, soft mobility offers significant health benefit potentials and it should be promoted in the local agendas.
- While rapidly growing cities pose a major challenge in terms of mobility demand and resource consumption, this also provides an opportunity to integrate air quality criteria in the general urban planning of new settlements and city enlargement.

Governance

- While multi-level governance and planning may be complicated at initial stages of air quality management, it is a powerful combination that matches the multi-scale reality of air pollution and may provide the most effective and sustainable response in the long term.
- Switching to renewable energy sources constitutes an example of national-scale strategy that will always report local air quality co-benefits. A harmonized and coordinated response to climate change and air quality is a pressing need to meet national climate commitments and local air quality standards.
- It is key to improve local governance to achieve real changes by strengthening institutions and collaborations among administrations, companies and other stakeholders. In addition to official plans and strategies, there is a breadth of projects and initiatives in collaboration with multiple international organisms that would benefit from a closer coordination. The creation of centralized emissions and air quality department in the cities may facilitate the allocation of responsibilities and an effective allocation of resources.
- Public-private partnerships have demonstrated to be an effective way to promote social development and air quality improvement in the cities analysed. The case studies also inform that involving actors of the informal economy is essential to provide a consistent response to air pollution and social issues in urban areas.
- A multi-national organism to coordinate national air quality plans and measures into a common African strategy may be instrumental to boost cooperation among countries, share experiences, exploit synergies and to deal with transboundary pollution issues. This may provide a fruitful framework to exchange experiences and harmonize criteria and methodologies.

Although information regarding emissions and air quality data in African cities is limited, the findings of this paper support the urgency to advocate form immediate action. Improving air quality in Africa involves addressing sprawling, dysfunctional cities, and urban planning in the context of a wider social and environmental considerations since poverty and air pollution are closely intertwined and the causes of social inequity and environmental deterioration must be dealt with consistently. The magnitude of the challenge requires long-term collaborations with international institutions, the private sector and the civil society at large. This applies to the scientific community as well. Cooperation among African universities and research groups would be instrumental to fill knowledge gaps and support better informed policies.

Note

An earlier version of this paper was presented at the International Conference on Air Quality in Africa (ICAQ' Africa 2022) in October 2022 and was published in its Proceedings.

References

Adon, M., Yoboue, V., Galy-Lacaux, C., Liousse, C., Diop, B., Doumbia, E. H. T., Gardrat, E., Ndiaye, S. A., Jarnot, C., 2016. Measurements of NO_2, SO_2, NH_3, HNO_3 and O_3 in West African urban environments. Atmospheric Environment 135, 31-40. https://doi.org/10.1016/j.atmosenv.2016.03.050

Adon, M., Yoboue, V., Galy-Lacaux, C., Liousse, C., Diop, B., Doumbia, E. H. T., Gardrat, E., Ndiaye, S. A., Jarnot, C., 2016. Measurements of NO_2, SO_2, NH_3, HNO_3 and O_3 in West African urban environments. Atmospheric Environment 135, 31-40. https://doi.org/10.1016/j.atmosenv.2016.03.050

Amoatey, P., Omidvarborna, H., Affum, H. A., Baawain, M., 2019. Performance of AERMOD and CALPUFF models on SO_2 and NO_2 emissions for future health risk assessment in Tema Metropolis. Human and Ecological Risk Assessment: An International Journal 25(3), 772-786. https://doi.org/10.1080/10807039.2018.1451745

Amoatey, P., Omidvarborna, H., Baawain, M., 2018. The modeling and health risk assessment of $PM_{2.5}$ from Tema Oil Refinery. Human and Ecological Risk Assessment: An International Journal 24 (5), 1-16. https://doi.org/10.1080/10807039.2017.1410427

Appoh, E., 2018. U.S. Overview of Air Quality Monitoring in Accra, Ghana-2. UC Davis Air Sensors International Conference, September 2018, Oakland, CA. [Available online at: https://asic2018.aqrc.ucdavis.edu/sites/g/files/dgvnsk3466/files/inline-files/Emmanuel%20Appoh%20-%20Ghana%20%20International%20Plenary%20Presentation%20at%20%208am%20of%2014%20Sept%202018.pdf]

Appoh, E., Terry, S., 2018. Clean Air for Ghana. Building on success. The Magazine for Environmental Managers. A&WMA. [Available online at: http://pubs.awma.org/flip/EM-Jan-2018/appoch.pdf]

Arab Republic of Egypt Ministry of Environment (EEAA), 2004. National Strategy for Cleaner Production in Egyptian Industry. [Available at: http://www.eeaa.gov.eg/portals/0/eeaaReports/IndUnit/CP.pdf]

Arab Republic of Egypt Ministry of Environment (EEAA), 2018. Annual report of the Ministry (2016). http://www.eeaa.gov.eg/portals/0/eeaaReports/achievements2016/%D8%A5%D9%86%D8%AC%D8%A7%D8%B2%D8%A7%D8%AA%D9%88%D8%B2%D8%A7%D8%B1%D8%A9%D8%A7%D9%84%D8%A8%D9%8A%D8%A6%D8%A9%D9%A2%D9%A0%D9%A1%D9%A6.pdf

Assamoi, E.M., Liousse, C., 2010. A new inventory for two-wheel vehicle emissions in West Africa for 2002. Atmospheric Environment 44, 3985-3996. https://doi.org/10.1016/j.atmosenv.2010.06.048

Bandowe, B. A. M., Nkansah, M. A., Leimer, S., Fischer, D., Lammel, G., Han, Y., 2019. Chemical (C, N, S, black carbon, soot and char) and stable carbon isotope composition of street dusts from a major West African metropolis: Implications for source apportionment and exposure. Science of The Total Environment 655, 1468 – 478. https://doi.org/10.1016/j.scitotenv.2018.11.089

Boman, J., Shaltout, A. A., Abozied, A. a. M., and Hassan, S. K., 2013. On the elemental composition of $PM_{2.5}$ in central Cairo, Egypt, X-Ray Spectrum., 42, 276–283, doi:10.1002/xrs.2464. https://doi.org/10.1002/xrs.2464

Burnett, R., Chen, H., Szyszkowicz, M., Fann, N., Hubbell, B., Pope, C.A., Apte, J.S., Brauer, M., Cohen, A., Weichenthal, S., Coggins, J., Di, Q., Brunekreef, B., Frostad, J., Lim, S.S., Kan, H., Walker, K.D., Thurston, G.D., Hayes, R.B., Lim, C.C., Turner, M.C., Jerrett, M., Krewski, D., Gapstur, S.M., Diver, W.R., Ostro, B., Goldberg, D., Crouse, D.L., Martin, R. V, Peters, P., Pinault, L., Tjepkema, M., van Donkelaar, A., Villeneuve, P.J., Miller, A.B., Yin, P., Zhou, M., Wang, L., Janssen, N.A.H., Marra, M., Atkinson, R.W., Tsang, H., Quoc Thach, T., Cannon, J.B., Allen, R.T., Hart, J.E., Laden, F., Cesaroni, G., Forastiere, F., Weinmayr, G., Jaensch, A., Nagel, G., Concin, H., Spadaro, J. V, 2018. Global estimates of mortality associated with long-term exposure to outdoor fine particulate matter. Proc. Natl. Acad. Sci. U. S. A. 115, 9592. https://doi.org/10.1073/pnas.1803222115

Center for International Earth Science Information Network (CIESIN), Columbia University, and Information Technology Outreach Services (ITOS), University of Georgia, 2013. Global Roads Open Access Data Set, Version 1 (gROADSv1). Palisades, NY: NASA Socioeconomic Data and Applications Center (SEDAC). https://doi.org/10.7927/H4VD6WCT

Chersich, M. F., Wright, C. Y., Venter, F., Rees, H., Scorgie, F., Erasmus, B., 2018. Impacts of Climate Change on Health and Wellbeing in South Africa. International Journal of Environmental Research and Public Health 15, 1884. https://doi.org/10.3390/ijerph15091884

Cissokho, L., Seck, A., 2013. Electric Power Outages and the Productivity of Small and Medium Enterprises in Senegal. ICBE-RF Research Report N0. 77/13 for the Investment Climate and Business Environment Research Fund (ICBE-RF). [Available online at: http://www.trustafrica.org/en/publications-trust/icbe-research-reports?download=354:electricpower-outages-and-the-productivity-of-small-andmedium-enterprises-in-senegal]

City of Cape Town (CCT) Environmental Planning Department, 2007. Cape Town Energy and Climate Change Strategy. ISBN: 0-9584719-3-2 [Available online at: http://resource.capetown.gov.za/documentcentre/Documents/City%20strategies,%20plans%20and%20frameworks/Energy_+_Climate_Change_Strategy_2_-_10_2007_301020079335_465.pdf]

City of Cape Town (CCT), 2011. Cape Town's Action Plan for Energy and Climate Change. ISBN 978-0-9802784-9-1. [Available online at: http://resource.capetown.gov.za/documentcentre/Documents/City%20strategies,%20plans%20and%20frameworks/Cape%20Town%20Energy%20and%20Climate%20Action%20Plan%20(ECAP).pdf]

City of Cape Town / 100 Resilient Cities, 2019. City of Cape Town Resilience Strategy. [Available online at: http://resource.capetown.gov.za/documentcentre/Documents/City%20strategies,%20plans%20and%20frameworks/Resilience_Strategy.pdf]

Clean Air Asia, 2016. Guidance Framework for Better Air Quality in Asian Cities. Pasig City, Philippines. ISBN: 978-621-95434-0-8. https://cleanairasia.org/ibaq/wp-content/themes/ibaq/pdf/GASOURCEBOOK.pdf

Clean Air Asia, 2016. Guidance Framework for Better Air Quality in Asian Cities. Pasig City, Philippines. ISBN: 978-621-95434-0-8. [Available online at: https://cleanairasia.org/ibaq/wp-content/themes/ibaq/pdf/GASOURCEBOOK.pdf]

De Souza, P., Nthusi, V., Klopp, J.M., Shaw, B.E., Ho, W.O., Saffell, J., Jones, R., Ratti, C., 2017. A Nairobi experiment in using low cost air quality monitors. Clean Air Journal 27 (2), 12-42. http://dx.doi.org/10.17159/2410-972X/2017/v27n2a6

Dieme, D., Cabral-Ndior, M., Garçon, G., Verdin, A., Billet, S., Cazier, F., Courcot, D.,Diouf, A., Shirali, P., 2012. Relationship between physicochemicalcharacterization and toxicity of fine particulate matter ($PM_{2.5}$) collected inDakar city (Senegal). Environmental Research 113, 1–13 https://doi.org/10.1016/j.envres.2011.11.009

Dionisio, K. L., Arku, R. E., Hughs, A. F., Vallarino, J., Carmichael, H., Spengler, J. D., Agyei-Mensah, S., Ezzati, M., 2010. Air pollution in Accra neighbourhoods: spatial, socioeconomic, and temporal patterns. Environmental Science & Technology 44, 2270-6. DOI: DOI: 10.1021/es903276s

Doumbia, E. H. T., Liousse, C., Galy-Lacaux, C., Ndiaye, S. A., Diop, B., Ouafo, M., Assamoi, E. M., Gardrat, E., Castera, P., Rosset, R., Akpo, A., Sigha, L., 2012. Real time black carbon measurements in West and Central Africa urban sites. Atmospheric Environment 54, 529-537. https://doi.org/10.1016/j.atmosenv.2012.02.005

Egondi, T., Muindi, K., Kyobutungi, C., Gatari, M., Rocklöv, J., 2016. Measuring exposure levels of inhalable airborne particles ($PM_{2.5}$) in two socially deprived areas of Nairobi, Kenya. Environmental Research 148, 500–506. http://dx.doi.org/10.1016/j.envres.2016.03.018

Environmental Protection Agency (EPA) Ghana, 2016. 2015 Annual Progress Report. [Available at: http://www.epa.gov.gh/epa/sites/default/files/downloads/publications/2015%20Annual%20Report.pdf]

Environmental Protection Agency (EPA) Ghana, 2017. Roadmap for the Promotion of Cleaner Buses in Accra, Ghana. [Available at: https://wedocs.unep.org/bitstream/handle/20.500.11822/21447/Final%20Roadmap%20for%20the%20Promotion%20of%20Cleaner%20Buses%20in%20Accra%20Ghana.pdf]

Environmental Protection Agency (EPA) Ghana, 2018. The Greater Accra Metropolitan Areas Air Quality Management Plan. ISBN 978-9988-2-8509-8. [Available at: http://www.epa.gov.gh/epa/sites/default/files/downloads/publications/Greater%20Accra%20Region%20Air%20%20Quality%20Management%20Plan%202%20Oct%20%202018%20updated.pdf]

European Commission (EC), 2008. Directive 2008/50/EC of the European Parliament and of the Council of 21 May 2008 on ambient air quality and cleaner air for Europe. [Available online at: https://eur-lex.europa.eu/legal-content/EN/TXT/PDF/?uri=CELEX:32008L0050&from=EN]

Favez, O., Cachier, H., Sciare, J., Alfaro, S., El-Araby, T. M., Harhash, M. A., Abdel Wahab, M. M., 2008. Seasonality of major aerosol species and their transformations in Cairo megacity. Atmospheric Environment 42 (7), 1503–1516. http://dx.doi.org/10.1016/j.atmosenv.2007.10.081.

Fisher, S., Bellinger, D.C., Cropper, M.L., Kumar, P., Binagwaho, A., Koudenoukpo, J.B., Park, Y., Taghian, G., Landrigan, P.J., 2021. Air pollution and development in Africa: impacts on health, the economy, and human capital. The Lancet Planetary Health 5 (10), 681-688. https://doi.org/10.1016/S2542-5196(21)00201-1

GADM database (www.gadm.org), version 3.4, April 2018.

Gaita, S. M., Boman, J., Gatari, M. J., Pettersson, J. B. C., Janhäll, S., 2014. Source apportionment and seasonal variation of $PM_{2.5}$ in a Sub-Saharan African city: Nairobi, Kenya. Atmospheric Chemistry and Physics 14, 9977–9991. https://doi.org/10.5194/acp-14-9977-2014

Gatari, M. J., Kinney, P. L., Yan, B., Sclar, E., Volavka-Close, N., Ngo, N. S., et al., 2019. High airborne black carbon concentrations measured near roadways in Nairobi, Kenya. Transportation Research Part D: Transport and Environment 68, 99–109. https://doi.org/10.1016/j.trd.2017.10.002

Government of the Republic of Kenya, 2018. Kenya Vision 2030. Third Medium Term Plan 2018 – 2022. Transforming Lives: Advancing socio-economic development through the "Big Four". [Available online at: http://vision2030.go.ke/inc/uploads/2019/01/THIRD-MEDIUM-TERM-PLAN-2018-2022.pdf]

Gulis, G., Mulumba, J. A. A., Juma, O., Kakosova, B., 2004. Health status of people of slums in Nairobi, Kenya. Environmental Research 96, 219-27. DOI: 10.1016/j.envres.2004.01.016

Hassanien, M. A., Abdel-Latif, N. M., 2008. Polycyclic aromatic hydrocarbons in road dust over Greater Cairo, Egypt. Journal of Hazardous Materials 151 (1) 247-254. https://doi.org/10.1016/j.jhazmat.2007.05.079

Heft-Neal, S., Burney, J., Bendavid, E., Burke, M., 2018. Robust relationship between air quality and infant mortality in Africa. Nature 559 (7713), 254–258. https://doi.org/10.1038/s41586-018-0263-3

Heft-Neal, S., Burney, J., Bendavid, E., Burke, M., 2018. Robust relationship between air quality and infant mortality in Africa. Nature 559 (7713), 254–258. https://doi.org/10.1038/s41586-018-0263-3

Hersey, S. P., Garland, R. M., Crosbie, E., Shingler, T., Sorooshian, A., Piketh, S., Burger, R., 2015. An overview of regional and local characteristics of aerosols in South Africa using satellite, ground, and modeling data. Atmospheric Chemistry and Physics 15, 4259–4278. https://doi.org/10.5194/acp-15-4259-2015

Jenkins, G.S., Diokhane, A. M., 2017. WRF prediction of two winter season Saharan dust events using PM_{10} concentrations: Boundary versus initial conditions. Atmospheric Environment 167, 129-142. http://dx.doi.org/10.1016/j.atmosenv.2017.08.010

Karagulian, F., Belis, C. A., Dora, C. F. C., Prüss-Ustün, A. M., Bonjour, S., Adair-Rohani, H., Amann, M., 2015. Contributions to cities' ambient particulate matter (PM): A systematic review of local source contributions at global level. Atmospheric Environment 120, 475-48. https://doi.org/10.1016/j.atmosenv.2015.08.087

Keen, S., Altieri, K., 2016. The health benefits of attaining and strengthening air quality standards in Cape Town. Clean

Air Journal 26 (2), 22 – 27. http://dx.doi.org/10.17159/2410-972X/2016/v26n2a9

Keita, S., Liousse, C., Yoboúe, V., Dominutti, P., Guinot, B., Assamoi, E.-M., Borbon, A., Haslett, S. L., Bouvier, L., Colomb, A., Coe, H., Akpo, A., Adon, J., Bahino, J., Doumbia, M., Djossou, J., Galy-Lacaux, C., Gardrat, E., Gnamien, S. , Léon, J. F., Ossohou, M., N'Datchoh, E. T., Roblou, L., 2018. Particle and VOC emission factor measurements for anthropogenic sources in West Africa. Atmospheric Chemistry and Physics 18 (10), 7691-7708. https://www.atmos-chem-phys.net/18/7691/2018

Kinney, P. L., Gichuru, M. G., Volavka-Close, N., Ngo, N., Ndiba, P. K., Law, A., Gachanja, A., Gaita, S. M., Chillrud, S. N., Sclar, E., 2011. Traffic Impacts on $PM_{2.5}$ Air Quality in Nairobi, Kenya. Environmental Science & Policy 14, 369–378. https://doi.org/10.1016/j.envsci.2011.02.005

Klausbruckner, C., Annegarn, H., Henneman, L. R. F., Rafaj, P., 2016. A policy review of synergies and trade-offs in South African climate change mitigation and air pollution control strategies. Environmental Science & Policy 57, 70–78. https://doi.org/10.1016/j.envsci.2015.12.001

Klopp J. M., 2012. Towards a political economy of transportation policy and practice in Nairobi. Urban Forum 23(1), 1-21. https://doi.org/10.1007/s12132-011-9116-y

Korkor, H., 2014. Promoting energy efficiency investments for Climate Change mitigation and sustainable development. Case study: Egypt. Policy reforms that were implemented to Promote Energy Efficiency in the Transportation Sector. Prepared for UN Economic and Social Commission for Western Asia. [Available online at: https://www.unescwa.org/sites/www.unescwa.org/files/page_attachments/escwa-casestudy-ee_transport-egypt_final.pdf]

Kumar, P., Morawska, L., Martani, C., Biskos, G., Neophytou, M., Di Sabatino, S., Bell, M., Norford, L., Britter, R., 2015. The rise of low-cost sensing for managing air pollution in cities. Environment International 75, 199–205. https://doi.org/10.1016/j.envint.2014.11.019

Kuyper, B., Wingrove, H., Lesch, T., Labuschagne, C., Say, D., Martin, D., Young, D., Khan, M. A. H., O'Doherty, S., Davies-Coleman, M. T., Shallcross, D. E., 2020. Atmospheric Toluene and Benzene Mole Fractions at Cape Town and Cape Point and an Estimation of the Hydroxyl Radical Concentrations in the Air above the Cape Peninsula, South Africa. ACS Earth and Space Chemistry 4 (1), 24-34. https://doi.org/10.1021/acsearthspacechem.9b00207

Lelieveld, J., Evans, J. S., Fnais, M., Giannadaki, D., Pozzer A., 2015. The contribution of outdoor air pollution sources to premature mortality on a global scale. Nature 525, 367. https://doi.org/10.1038/nature15371

Liousse C., E. Assamoi, P. Criqui, C. Granier and R. Rosset, 2014. African combustion emission explosive growth from 2005 to 2030. Environ. Res. Lett. 9 035003. doi:10.1088/1748-9326/9/3/035003

Mahmoud, K.F., Alfaro, S.C., Favez, O., Abdel Wahab, M. M., Sciare, J., 2008. Origin of black carbon concentration peaks in Cairo (Egypt). Atmospheric Research 89, 161–169. https://doi.org/10.1016/j.atmosres.2008.01.004

Marey, H. S., Gille, J. C., El-Askary, H. M., Shalaby, E. A., El-Raey, M. E., 2010. Study of the formation of the "black cloud" and its dynamics over Cairo, Egypt, using MODIS and MISR sensors. Journal of Geophysical Research 115, D21206. https://doi.org/10.1029/2010JD014384

Mbow-Diokhane, A., 2019. Air Quality in African Cities. In: Mboup G., Oyelaran-Oyeyinka B. (eds) Smart Economy in Smart African Cities. Advances in 21st Century Human Settlements. Springer, Singapore. ISBN 978-981-13-3470-2. https://doi.org/10.1007/978-981-13-3471-9_9

Mbow-Diokhane, A., 2019. Air Quality in African Cities. In: Mboup G., Oyelaran-Oyeyinka B. (eds) Smart Economy in Smart African Cities. Advances in 21st Century Human Settlements. Springer, Singapore. ISBN 978-981-13-3470-2. https://doi.org/10.1007/978-981-13-3471-9_9

Ministry of Environment and Forestry (MoEF), 2014. Cabinet Secretary for Environment, Water and Natural Resources. The Environmental Management and Co-ordination Act (No.8 of 1999). Legal Notice No. 34 Kenya Gazette Supplement No.41. [Available online at: http://kenyalaw.org/kl/fileadmin/pdfdownloads/LegalNotices/34-EnvironmentalManagementandCo-ordination(AirQuality)Regulations2014.pdf]

Molepo, K. M., Abiodun, B. J., Magoba, R. N., 2019. The transport of PM_{10} over Cape Town during high pollution episodes. Atmospheric Environment 213, 116-132. https://doi.org/10.1016/j.atmosenv.2019.05.041

Mostafa, A. N., Zakey, A. S., 2018. Analysis of the surface Air Quality Measurements in The Greater Cairo (Egypt) Metropolitan. Global Journal of Advanced Research 5 (6), 207-214. ISSN: 2394-5788. [Available online at: http://gjar.org/publishpaper/vol5issue6/d843r9.pdf]

Muchapondwa, 2010. A cost-effectiveness analysis of options for reducing pollution in Khayelitsha township, South Africa. The Journal for Transdisciplinary Research in Southern Africa 6 (2), 333 - 358. https://doi.org/10.4102/td.v6i2.268

Mumm, R., Diaz-Monsalve, S., Hänselmann, E., Freund, J., Wirsching, M., Gärtner, J., Gminski, R., Vögtlin, K., Körner, M., Zirn, L., Wittwer-Backofen, U., Oni, T., Kroeger, A., 2017. Exploring urban health in Cape Town, South Africa: an interdisciplinary analysis of secondary data. Pathogens and Global Health 111 (1), 7-22. https://doi.org/10.1080/20477724.2016.1275463

Naidja, L., Ali-Khodja, H., Khardi, S., 2018. Sources and levels of particulate matter in North African and Sub-Saharan cities: a literature review. Environmental Science and Pollution Research 25, 12303–12328. https://doi.org/10.1007/s11356-018-1715-x

Naiker, Y., R.D.Diab, R. D., Zunckel, M., Hayes, E. T., 2012. Introduction of local Air Quality Management in South Africa: overview and challenges. Environmental Science & Policy 17, 62-71. https://doi.org/10.1016/j.envsci.2011.11.009

Nairobi City Country (NCC) supported by United Nations Environmental Programme (UNEP), 2019. Air Quality Action Plan (2019-2023). [Available online at: https://www.eci-africa.org/wp-content/uploads/2019/05/Nairobi-Air-Quality-Action-Plan_Final_ECI_31.12.2018.pdf]

Nairobi City County (NCC) and the Japan International Cooperation Agency (JICA), 2014. The Project on Integrated Urban Development Master Plan for the City of Nairobi in the Republic of Kenya [Available online at: http://www.kpda.or.ke/documents/Nairobi%20Integarted%20Urban%20Development%20Master%20Plan.pdf]

Nasralla, M. M., 2001. Greater Cairo Air Quality profile. Egyptian Environmental Policy Program Support Unit. Prepared under the Environmental Policy and Institutional Strengthening Indefinite Quantity Contract (EPIQ) for U.S. Agency for International Development, Cairo and Egyptian Environmental Affairs Agency (EEAA). [Availableonlineat:https://rmportal.net/library/content/tools/environmental-policy-and-institutional-strengthening-epiq-iqc/egyptian-environmental-policy-program-eepp-program-support-unit-psu-task-order-cd-vol-2/epiq-reports-technical-area-dissemination-of-policy-knowledge-environmental-communication/034-cairoairprofile-nasrala.pdf/at_download/file]

Ndong Ba, A., Cazier, F., Verdin, A., Garçon, G., Cabral, M., Courcot, L., Diouf, A., Courcot, D., Gualtieri, M., Fall, M., 2019b. Physico-chemical characterization and in vitro inflammatory and oxidative potency of atmospheric particles collected in Dakar city's (Senegal). Environmental Pollution 245, 568-581. https://doi.org/10.1016/j.envpol.2018.11.026

Ndong Ba, A., Verdin, A., Cazier, F., Garçon, G., Thomas, J., Cabral, M., Dewaele, D., Genevray, P., Garat, A., Allorge, D., Diouf, A., Loguidice, J.M., Courcot, D., M., Courcot, L., Fall, M., Gualtieri, 2019a. Individual exposure level following indoor and outdoor air pollution exposure in Dakar (Senegal). Environmental Pollution 248, 397-407. https://doi.org/10.1016/j.envpol.2019.02.042

Ndour, M., D'Anna, B., George, C., Ka, O., Balkanski, Y., Kleffmann, J., Stemmler, K., Ammann, M., 2008. Photo-enhanced uptake of NO_2 on mineral dust: laboratory experiments and model simulations. Geophysical Research Letters 35, L05812. https://doi.org/10.1029/2007GL032006

Ngo, N. S., Kokoyo, S., Klopp, J., 2017. Why participation matters for air quality studies: risk perceptions, understandings of air pollution and mobilization in a poor neighbourhood in Nairobi, Kenya. Public Health 142, 177–185. https://doi.org/10.1016/j.puhe.2015.07.014

Ngo, N.S., Gatari, M., Yan, B., Chillrud, S.N., Bouhamam, K., Kinney, P.L., 2015. Occupational exposure to roadway emissions and inside informal settlements in sub-Saharan Africa: a pilot study in Nairobi, Kenya. Atmospheric Environment 111, 179–184. https://doi.org/10.1016/j.atmosenv.2015.04.008

Nthusi, V., 2017. Nairobi Air Quality Monitoring Sensor Network Report. Technical report published April 2017 via Science Division, UN Environment. [Available online at: https://doi.org/10.13140/rg.2.2.10240.64009]

Ofosu, F. G., Hopke, P. K., Aboh, I. J. K., Bamford, S. A., 2012. Characterization of fine particulate sources at Ashaiman in Greater Accra, Ghana. Atmos. Pollut. Res. 3, 301–310. https://doi.org/10.5094/APR.2012.033

Oluleye, A., Folorunsho, A., 2019. Influence of Atmospheric Dynamic Factors on Dust Aerosol Mobilization Over West Africa: Simulations from WRF-Chem. Aerosol Science and Engineering 3, 132–149. https://doi.org/10.1007/s41810-019-00048-z

Piedrahita, R., Kanyomse, E., Coffey, E., Xie, M., Hagar, Y., Alirigia, R., Agyei, F., Wiedinmyer, C., Dickinson, K. L., Oduro, A., Hannigan, M., 2017. Exposures to and origins of carbonaceous $PM_{2.5}$ in a cookstove intervention in Northern Ghana. Science of The Total Environment 576, 178-192. https://doi.org/10.1016/j.scitotenv.2016.10.069

Pope, F. D., Gatari, M., Ng'ang'a, D., Poynter, A., Blake, R., 2018. Airborne particulate matter monitoring in Kenya using calibrated low-cost sensors. Atmospheric Chemistry and Physics 18, 15403–15418. https://doi.org/10.5194/acp-18-15403-2018

Pure Earth, 2015. Project Completion Report: Making Electronic Waste Recycling in Ghana Safer Through Alternative Technology, Accra Ghana. [Available online at: https://www.pureearth.org/wp-content/uploads/2014/01/Ghana-Pilot-PCR-2015.pdf]

Republic of Kenya. Ministry of Energy and Petroleum (MoEP), 2016. Sustainable Energy for All. Kenya Action Agenda. Pathways for Concerted Action toward Sustainable Energy for All by 2030. [Available online at: https://www.se4all-africa.org/fileadmin/uploads/se4all/Documents/Country_AAs/Kenya_SE4ALL_AA_January_2016.pdf]

Republic of South Africa, 2013. The 2012 National Framework for Air Quality Management in the Republic of South Africa. Government Gazette No. 36161 (volume 572). Pretoria, 15 February 2013. [Available online at: https://www.environment.gov.za/sites/default/files/gazetted_notices/nema_national_framework_aqm_gn115.pdf]

Republic of South Africa. Department of Environmental Affairs (DEA), 2012. National Ambient Air Quality Standard for particulate

matter with aerodynamic diameter less than 2.5 micron metres ($PM_{2.5}$). 29 June, 2012. [Available online at: https://cer.org.za/wp-content/uploads/2010/03/National-Ambient-Air-Quality-Standard_PM2.5.pdf]

Republic of South Africa. Department of Environmental Affairs (DEA), 2019. Department of Environmental Affairs annual report 2018/2019. ISBN: 978-0-621-45872-5. [Available online at: https://www.environment.gov.za/sites/default/files/reports/annualreport201819.pdf]

République du Sénégal. Ministere de L'environnement et du Developpement Durable. Centre de Gestion de la Qualite de L'air (CGQA), 2018. Suivi de la qualité de l'air à Dakar Rapport annuel 2017. [Available online at: https://www.air-dakar.org/images/pdf/bulletins/rapport_annuel_2017.pdf]

Republique du Senegal. Ministry for Urban Renewal, Housing and Living Environment (MURHLE) and the Japan International Cooperation Agency (JICA), 2016. Project for Urban Master Plan of Dakar and Neighboring Area for 2035. [Available online at: http://open_jicareport.jica.go.jp/618/618/618_526_12250007.html]

Republique du Senegal. Ministry of Economy, Planning and Cooperation (MEPC), 2014. Plan for an Emerging Senegal. Phase I PSE 2014 2018. [Available online at: https://www.economie.gouv.sn/sites/default/files/fichier/Phase%20I%20PSE%202014-2018.pdf]

Republique du Senegal. Ministry of Economy, Planning and Cooperation (MEPC), 2018. Plan for an Emerging Senegal. Phase II PSE 2019 2023. [Available online at: https://www.economie.gouv.sn/sites/default/files/fichier/Phase%20II%20PSE%202019-2023.pdf]

Rodríguez, S., Calzolai, G., Chiari, M., Nava, S., García, M. I., López-Solano, J., Marrero, C., López-Darias, J., Cuevas, E., Alonso-Pérez, S., Prats, N., Amato, F., Lucarelli, F., Querol, X., 2019. Rapid changes of dust geochemistry in the Saharan Air Layer linked to sources and meteorology. Atmospheric Environment 27, 117186. https://doi.org/10.1016/j.atmosenv.2019.117186

Roy, R., 2016. The cost of air pollution in Africa, OECD Development Centre Working Papers, No. 333, OECD Publishing, Paris. ISSN: 18151949. https://www.oecd-ilibrary.org/the-cost-of-air-pollution-in-africa_5jlqzq77x6f8.pdf?itemId=%2Fcontent%2Fpaper%2F5jlqzq77x6f8-en&mimeType=pdf

Safar, Z., Labib, M. W., 2010. Assessment of particulate matter and lead levels in the Greater Cairo area for the period 1998-2007. Journal of Advanced Research 1, 53-63. https://doi.org/10.1016/j.jare.2010.02.004

Saiz-Lopez, A., Borge, R., Notario, A., Adame, J.A., de la Paz, D., Querol, X., Artíñano, B., Gomez-Moreno, F.J., Cuevas, C.A., 2017. Unexpected increase in the oxidation capacity of the urban atmosphere of Madrid, Spain. Scientific Reports 7, 45956. https://doi.org/10.1038/srep45956

Sarr, D., Diop, B., Farota, A. K., Sy, A., Diop, A. D., Wade, M., Diop, A. B., 2018. Atmospheric Circulation in Low Troposphere-Effect On NO_x and CO Emissions in Traffic and Air Quality in Dakar. Journal of Pollution Effects & Control 6 (3). DOI: 10.4172/2375-4397.1000230

Schwela, D., 2012. Review of urban air quality in Sub-Saharan Africa region - air quality profile of SSA countries (English). Washington, DC: World Bank. [Available online at: http://documents.worldbank.org/curated/en/936031468000276054/Review-of-urban-air-quality-in-Sub-Saharan-Africa-region-air-quality-profile-of-SSA-countries]

Shilenje, Z. W., 2014. Development in Air Pollution Measurement Technologies in Kenya. TECO-2014. WMO technical conference on meteorological and environmental instruments and methods of observation. [Available online at: https://www.wmo.int/pages/prog/www/IMOP/publications/IOM-116_TECO-2014/Session%201/P1_47_Shilenje_Air_Pollution_Measurement.pdf]

Stanaway, J. D., Afshin, A., Gakidou, E., Lim, S. S., Abate, D., Abate, K. H. et al., 2018. Global, regional, and national comparative risk assessment of 84 behavioural, environmental and occupational, and metabolic risks or clusters of risks for 195 countries and territories, 1990–2017: A systematic analysis for the Global Burden of Disease Study 2017. Lancet 392, 1923–1994. https://doi.org/10.1016/S0140-6736(18)32225-6

Sylla, F. K., Faye, A., Diaw, M., Fall, M., Tal-Dia, A., 2018. Traffic Air Pollution and Respiratory Health: A Cross-Sectional Study among Bus Drivers in Dakar (Senegal). Open Journal of Epidemiology 8, 1-13. https://doi.org/10.4236/ojepi.2018.81001

Tang, C., Garshick, E., Grady, S., Coull, B., Schwartz, J., Koutrakis, P., 2018. Development of a modeling approach to estimate indoor-to-outdoor sulfur ratios and predict indoor PM2.5 and black carbon concentrations for Eastern Massachusetts households. Journal of Exposure Science & Environmental Epidemiology 28, 125–130. https://doi.org/10.1038/jes.2017.11

The City of Cape Town's Transport and Urban Development Authority (TDA), 2017b. City of Cape Town Cycling Strategy. [Available online at: https://tdacontenthubfunctions.azurewebsites.net/Document/1403]

The City of Cape Town's Transport and Urban Development Authority (TDA), 2005. Non-Motorised Transport (NMT) Strategy Volume 1 (Status Quo Assessment) & Volume 2 (Policy Framework). [Available online at: https://tdacontenthubfunctions.azurewebsites.net/Document/30 and https://tdacontenthubfunctions.azurewebsites.net/Document/31]

The International Institute for Sustainable Development (IISD), 2019a. Sustainable Asset Valuation (SAVi) of the N'Diaye Wind Farm in Senegal. Summary of results. [Available online at: https://www.iisd.org/sites/default/files/publications/savi-senegal-ndiaye-wind-farm-en.pdf]

The International Institute for Sustainable Development (IISD), 2019b. Sustainable Asset Valuation (SAVi) of the Bus Rapid Transit Project in Senegal. Summary of results. [Available online at: https://www.iisd.org/sites/default/files/publications/savi-senegal-bus-rapid-transit-en.pdf]

The World Bank (WB), 2009. Senegal - Urban Mobility Improvement Program Project (English). Washington, DC: World Bank. Implementation Completion and Results Report. Report No: ICR0000955 [Available online at: http://documents.worldbank.org/curated/en/576841468103767226/pdf/ICR9550P0554721C0Disclosed051221091.pdf]

The World Bank (WB), 2013. The Arab Republic of Egypt. For better or for worse: air pollution in Greater Cairo. A sector note. Sustainable Development Department. Middle East and North Africa Region. Report No. 73074-EG. [Available online at: http://documents.worldbank.org/curated/en/972321468021568180/pdf/730740ESW0P09700Final0April02202013.pdf]

The World Bank (WB), 2015. Senegal Urbanization Review. How can cities contribute to the "Senegal Emerging Plan"? [Available online at: https://knowledge.uclga.org/IMG/pdf/senegalurbanizationreview.pdf]

The World Bank (WB), 2016. The cost of air pollution: strengthening the economic case for action (English). Washington, D.C.: World Bank Group. [Available online at: http://documents.worldbank.org/curated/en/781521473177013155/pdf/108141-REVISED-Cost-of-PollutionWebCORRECTEDfile.pdf]

The World Bank (WB), 2019. World Development Indicators [available online at: http://datatopics.worldbank.org/world-development-indicators/]

Thomas, A., 2016. Reducing Air Pollution in Cairo: Raise User Costs and Invest In Public Transit. Economic Research Forum. Policy Brief. ERF Policy Brief No. 12 | May 2016. [Available online at: http://erf.org.eg/wp-content/uploads/2016/05/PB12_2016.pdf]

Toure, N. O., Gueye, N. R. D., Mbow-Diokhane, A., Jenkins, G. S., Li, M., Drame, M. S., Coker, K. A. R., Thiam, K., 2019. Observed and modeled seasonal air quality and respiratory health in Senegal during 2015 and 2016. GeoHealth, 3. https://doi.org/10.1029/2019GH000214

Transparency International, 2019. Corruption Perceptions Index 2018. ISBN: 978-3-96076-116-7. [Available online at: https://files.transparency.org/content/download/2383/14554/file/2018_CPI_Executive_Summary.pdf]

Tshehla C., Wright, C. Y., 2019. 15 Years after the National Environmental Management Air Quality Act: Is legislation failing to reduce air pollution in South Africa? South African Journal of Science 115 (9/10), Art. #6100. https://doi.org/10.17159/sajs.2019/6100

United Nations (UN), Department of Economic and Social Affairs, Population Division (2019). World Population Prospects 2019, Online Edition. Rev. 1.

United Nations Environment Programme (UNEP), 2004. Air Quality and Atmospheric Pollution in the Arab Region. Manama: Joint Technical Secretariat composed of the League of Arab States; the United Nations Economic and Social Commission for Western Asia; and the United Nations Environment Programme, Regional Office for West Asia. [Available online at: http://www.un.org/esa/sustdev/csd/csd14/escwaRIM_bp1.pdf]

United Nations Environment Programme (UNEP), 2016. Actions on air quality. Policies & Programmes for Improving Air Quality around the World. UNEP Transport Unit. https://wedocs.unep.org/bitstream/handle/20.500.11822/17203/AQ_GlobalReport_Summary.pdf

United Nations Human Settlements Programme (UN Habitat), 2023. Air Quality in African Cities. [Available online at: https://unhabitat.org/sites/default/files/2023/03/un-habitat_2023_-_air_quality_in_african_cities.pdf]

United Nations Human Settlements Programme (UN Habitat), 2010. The State of African Cities 2010. Governance, Inequality and Urban Land Markets. ISBN Number: (Volume) 978-92-1-132291-0. http://www.citiesalliance.org/sites/citiesalliance.org/files/UNH_StateofAfricanCities_2010.pdf

United Nations Human Settlements Programme (UN Habitat), 2016. Urbanization and Development. Emerging Future. World Cities Report, 2016. https://unhabitat.org/sites/default/files/download-manager-files/WCR-2016-WEB.pdf

United Nations, Department of Economic and Social Affairs, Population Division (UN), 2019. World Urbanization Prospects: The 2018 Revision (ST/ESA/SER.A/420). New York: United Nations. ISBN: 978-92-1-148319-2. [Available online at: https://population.un.org/wup/Publications/Files/WUP2018-Report.pdf]

van Donkelaar, A. et al. 2018. Global Annual $PM_{2.5}$ Grids from MODIS, MISR and SeaWiFS Aerosol Optical Depth (AOD) with GWR, 1998-2016. Palisades, NY: NASA Socioeconomic Data and Applications Center (SEDAC). https://doi.org/10.7927/H4ZK5DQS

van Vliet, E. D. S., Kinney, P. L., 2007. Impacts of roadway emissions on urban particulate matter concentrations in sub-Saharan Africa: new evidence from Nairobi, Kenya. Environmental Research Letters 2 (4), 045028. https://doi.org/10.1088%2F1748-9326%2F2%2F4%2F045028

Ville de Dakar / 100 Resilient Cities, 2016. Dakar Resilience Strategy. [Available online at: https://www.100resilientcities.org/wp-content/uploads/2017/07/Dakar_Resilience_Strategy.pdf]

Watson, J. G., Robinson, N. F., Chow, J. C., Henry, R. C., Kim, B. M., Pace, T. G., Meyer, E. L., Nguyen, Q., 1990. The US EPA/DRI chemical mass balance receptor model, CMB 7.0. Environmental Software 5 (1), 38-49. https://doi.org/10.1016/0266-9838(90)90015-X

Western Cape Government. Department of Environmental Affairs and Development Planning (DEA&DP), 2019. Western Cape State of Air Quality Management Report 2018. [Available online at: https://www.westerncape.gov.za/eadp/files/atoms/files/SoAR_2018.pdf]

Western Cape Government. Department of Environmental Affairs and Development Planning (DEA&DP), 2016. Western Cape Air Quality Management Plan. ISBN: 978-0-621-45636-3. [Available online at: https://www.westerncape.gov.za/eadp/files/atoms/files/Western%20Cape%20Air%20Quality%20Management%20Plan.pdf]

Western Cape Government. Department of Environmental Affairs and Development Planning (DEA&DP), 2014. Western Cape Climate Change Response Strategy. [Available online at: https://www.westerncape.gov.za/text/2015/march/western_cape_climate_change_response_strategy_2014.pdf.]

Wheida, A., Nasser, A., El Nazer, M., Borbon, A., El Ata, G., Wahab, M., Alfaro, S., 2018. Tackling the mortality from long-term exposure to outdoor air pollution in megacities: Lessons from the Greater Cairo case study. Environmental Research 160, 223-231. https://doi.org/10.1016/j.envres.2017.09.028

Wicking-Baird, M. C., De Villiers, M. G., Dutkiewicz, R., 1997. Cape Town Brown Haze Study. Report No. Gen 182. Energy Research Institute, University of Cape Town, Cape Town. [Available online at: https://open.uct.ac.za/bitstream/handle/11427/23910/Wicking-Baird_ERC_1997.pdf]

World Health Organization (WHO), 2006. WHO Air quality guidelines for particulate matter, ozone, nitrogen dioxide and sulphur dioxide. Global update 2005. Summary of risk assessment. WHO/SDE/PHE/OEH/06.02. [Available online at: https://apps.who.int/iris/bitstream/10665/69477/1/WHO_SDE_PHE_OEH_06.02_eng.pdf]

World Health Organization (WHO), 2016. Ambient air pollution: A global assessment of exposure and burden of disease. ISBN: 978 92 4 151135 3. [Available online at: http://apps.who.int/iris/bitstream/handle/10665/250141/9789241511353-eng.pdf]

World Meteorological Organization (WHO), 2018. Low-cost sensors for the measurement of atmospheric composition: overview of topic and future applications. WMO-No. 1215. Editors: Lewis, A.C., von Schneidemesser, E., Peltier, R.E. ISBN 978-92-63-11215-6. [Available online at: https://www.ccacoalition.org/en/file/4645/download?token=RprgoCCg]

Yaah, V. B. K., 2018. Improvement of the Waste Management System in Senegal. Mediterranean Journal of Basic and Applied Sciences (MJBAS) 2 (3), 105-126. [Available online at: http://mjbas.com/data/uploads/4008.pdf]

Zakey, A. S., Abdel-Wahab, M. M., Pettersson, J. B. C., Gatari, M. J., Hallquist, M., 2008. Seasonal and spatial variation of atmospheric particulate matter in a developing megacity, the Greater Cairo, Egypt. Atmosfera 21(2), 171–189. [Available online at: http://www.scielo.org.mx/pdf/atm/v21n2/v21n2a4.pdf]

Zamudio, A. N., Terton, A., 2016. Review of current and planned adaptation action in Senegal. CARIAA Working Paper no. 18. International Development Research Centre, Ottawa, Canada and UK Aid, London, United Kingdom. ISSN: 2292-6798. [Available online at: www.idrc.ca/cariaa] Republique du Senegal. Ministre de l'Environnement et du Developpement Durable (MEDD), 2015. Strategie Nationale de Developpement Durable (SNDD). [Available online at: https://chm.cbd.int/api/v2013/documents/1B1ECE54-E678-582A-3206-64226B5FF510/attachments/SNDD%202015_Fersion%20Finale.pdf]

Zhou, Z., Dionisio, K. L., Verissimo, T. G., Kerr, A. S., Coull, B., Arku R. E., Koutrakis, P., Spengler, J. D., Hughes, A. F., Vallarino, J., Agyei-Mensah, S., Ezzati, M., 2013. Chemical composition and sources of particle pollution in affluent and poor neighborhoods of Accra, Ghana. Environ Res Lett, 8–4. doi:10.1088/1748-9326/8/4/044025

Table 2: *Status of progress in addressing air quality & promising practices. Colours indicate the stage of development: green=mature, yellow=intermediate and red=early. Source: UN Habitat, 2023*

City (Country)	Stage in Air Quality Management Process						
	Air quality standards & monitoring		Emissions inventories and modelling	Health (and/or other) impact assessments	Communication	Clean Air Action plans	Governance
	Monitoring (# and type of stations)	National air quality standards**					
Accra (Ghana)	5 fixed reference stations*, 23 low-cost monitors	Air Quality Guidelines proposed only for PM_{10}	Some research-oriented emission estimates. Limited modelling capabilities	Some studies focussing on indoor air quality issues. Relevant impact from e-waste open burning traffic too	AirNow-Ghana system being developed to share AQ data	2018 Greater Accra Metropolitan Areas Air Quality Management Plan (Ghana's EPA)	Comprehensive legal and institutional framework. Lack of integration and operational capacity
Cairo (Egypt)	42 fixed reference stations and 25 sampling stations in GCMA	TSP, PM_{10} NO_2, SO_2 O_3 and Pb ($PM_{2.5}$ missing)	Urban emission inventory available for 2010, need to be updated. Limited modelling capabilities	Some studies highlight the impact of traffic	AQ reports issued but data not publicly available	Several national plans and instruments but no AQ-specific strategy	Poor ministerial coordination and lack of a clear institutional framework for local AQ management
Cape Town (South Africa)	17 fixed reference stations*	SO_2, NO_2, PM_{10}, O_3, C_6H_6, Pb, CO and $PM_{2.5}$	Some research-oriented emission estimates and industrial inventories. Limited modelling capabilities	Some studies quantify the impact of meeting air quality standards, particularly beneficial for vulnerable groups	South African Air Quality Information System (SAAQIS) and National Association for Clean Air (NACA)	2016 Western Cape Province Air Quality Management Plan along with many other national and local strategies	Multi-level distribution of competences and profuse regulation. Lack of centralized local AQM authority
Dakar (Senegal)	5 fixed reference stations*, 1 mobile lab	SO_2, NO_2, CO, PM_{10} and Pb ($PM_{2.5}$ missing)	Some research-oriented emission estimates. Limited modelling capabilities	Studies point out that indoor air quality is a major health issue. Traffic is also particularly relevant in Dakar	Centre de Gestion de la Qualité de l'Air (CGQA) oficial AQ data, forecast and information	National development strategy (ESP) and several relevant local instruments but no specific AQ strategy	Comprehensive strategy to keep strengthening administrations and increase cooperation. Limited integration.
Nairobi (Kenya)	2 fixed reference stations*, 1 mobile lab, 6 low-cost monitors	SO_x, NO_x, CO, O_3, Pb, PM_{10} and $PM_{2.5}$. (Not effectively enforced)	Some research-oriented emission estimates. No modelling activities	Some studies highlight the impact of traffic household combustion and open burning	Clean Air Nairobi platform to share AQ data. Small-scale demonstrative projects	Air Quality Action Plan (2019-2023), complementary to other local and national relevant plans	Instruments and structures available. Institutional weakness and corruption

* *Despite the existence of these stations, data is often outdated, inconsistent or missing.*
** *Not in cities' control; included to indicate existing enabling/legislative framework at national level.*

Research article

Assessing anthropogenic contribution to $PM_{2.5}$ from an urban residential area of Lagos, Nigeria using aliphatic hydrocarbon compounds as indicators

Oluwabamise L Faboya[1,2], Khanneh W Fomba[2], Godwin C Ezeh[3], and Hartmut Herrmann[2*]

[1]Department of Chemistry, Federal University Oye-Ekiti, Ekiti State, Nigeria
[2]Atmospheric Chemistry Department (ACD), Leibniz Institute for Tropospheric Research (TROPOS), Permoserstraße 15, 04318 Leipzig, Germany
[3]Atmospheric Research and Information Analysis Laboratory (ARIAL), Centre for Energy Research and Development (CERD), Obafemi Awolowo University, Ile-Ife, Nigeria
* Corresponding author: herrmann@tropos.de

Received: 27 March 2023 - **Reviewed:** 18 August 2023 - **Accepted:** 8 October 2023
https://doi.org/10.17159/caj/2023/33/2.15790

Abstract

Aliphatic hydrocarbons in $PM_{2.5}$ samples from a residential area of Lagos, Nigeria, were quantified using a Curie-point pyrolysis–gas chromatography–mass spectroscopy (CPPGC/MS). The total concentrations of ∑n-alkanes (C_{20}-C_{34}), hopanes ($\sum_4$Hopanes), and steranes ($\sum_3$Steranes) ranged from 1.8 to 146.6 ng m^{-3} (avg. 43.6 ± 35.1 ng m^{-3}), 0.0 to 89.9 ng m^{-3} (avg. 18.7 ± 25.9 ng m^{-3}), and 0.0 to 26.0 ng m^{-3} (avg. 3.3 ± 6.7 ng m^{-3}), accounting for 0.2%, 0.1%, and 0.01% of the total $PM_{2.5}$ mass concentrations, respectively. They also accounted for, 1%, 0.4% and 0.1% of the organic matter content, respectively, making them non-negligible component of the organic matter in this region. An evaluation of the n-alkanes source diagnostic indices revealed that engines powered by gasoline and diesel were the main sources of particulate matter aliphatic hydrocarbons in this area of Lagos, consistent with the results obtained from steranes and hopanes profiles. Based on the air mass trajectories, both local and regional air mass transports showed a significant influence on the amounts of carbonaceous materials and aliphatic hydrocarbon compounds arriving at the study site. The results from this study established the occurrence of anthropogenic air pollution in Lagos residential area with a major contribution from petrogenic sources.

Keywords

ambient air quality, anthropogenic contaminants, n-alkanes, hopanes, and steranes

Introduction

Atmospheric particulate matter (PM) with an aerodynamic diameter of 2.5 µm or less ($PM_{2.5}$) is considered a major contributor to severe air pollution and has significant impacts on human health, atmospheric visibility and climate (Russell and Brunekreef, 2009; Javed et al., 2019; Tian et al., 2020; Sun et al., 2021). The composition of $PM_{2.5}$ is very complicated and includes organic carbon (OC) and elemental carbon (EC), often referred to as carbonaceous components, trace metals, inorganic salts and some specific organic compounds such as aliphatic hydrocarbons including n-alkanes, steranes and hopanes, n-alkanoic acids and polycyclic aromatic hydrocarbons (PAHs) (Mandalakis et al., 2002; Javed et al., 2019; Sun et al., 2021). However, the most prevalent and significant class of organic compounds in the atmosphere are the aliphatic hydrocarbons (Fu et al., 2008; Ren et al., 2016; Boreddy et al., 2018). Due to their specificity, hydrophobicity, and prolonged persistence in the environment, these aliphatic hydrocarbon compounds have been used as organic molecular markers for pollution source identification (Simoneit et al., 1991; Mudge and Duce, 2005; Andreou et al., 2007; Javed et al., 2019; Sun et al., 2021). For example, n-alkanes primarily come from both anthropogenic and biological sources, such as petroleum residue, diesel engine exhaust (Simoneit,1984), and biomass burning (Simoneit and Elias, 2000, 2001), and also from terrestrial plant wax, fungi, bacteria, algae, and plankton (Brown et al., 2002). Additionally, plant wax n-alkanes are employed directly as a proxy for both the atmospheric transport and genesis of organic aerosol (OA) in the atmosphere (Tang et al., 2006; Lyu et al., 2017). Hopanes and steranes are ubiquitous in crude petroleum and are enhanced in the lubricating oils used in diesel and gasoline-powered motor vehicles, and their existence in the environment confirms the contribution of the burning of fossil fuels in the area (Simoneit et al., 2004; Rushdi et al. 2017; Javed et al., 2019). EC is mainly emitted from anthropogenic combustion sources, whereas

OC, as a complex mixture of many groups of compounds, has both primary and secondary origins. Primary organic carbon (POC) is generated during the combustion process, while secondary organic carbon (SOC) is produced in the atmosphere through the gas to particle conversion processes of volatile precursors. Exposure to high levels of these urban carbonaceous components has been linked with cardiovascular mortality and morbidity (Samara et al., 2013).

At present, improving the ambient air quality is one of Lagos's greatest environmental challenges. Studies relating to the $PM_{2.5}$ pollution status in Lagos have been reported by various authors (Ezeh et al., 2012; Obioh et al., 2013; Owoade et al., 2013; Ezeh et al., 2014; Alani et al., 2019; Zeng et al., 2019). However, reports have focused on the mass concentrations and elemental characterisation of $PM_{2.5}$, while information on specific organic compound classes such as aliphatic hydrocarbons in $PM_{2.5}$ that can be utilised as source markers in identifying the primary sources of organic aerosols (OA) are very scarce. For instance, Ezeh et al. (2012) and (2014) determined $PM_{2.5}$ mass concentrations at different locations in Lagos during the rainy season. Although their results showed that $PM_{2.5}$ mass concentrations were below the World Health Organization (WHO) limit of 25 µg m^{-3}, they argued on the basis of correlation and enrichment factor analyses that most trace elements in $PM_{2.5}$ originated from anthropogenic sources. Similarly, Owoade et al. (2013) determined the $PM_{2.5}$ mass concentrations from different classes of receptor sites in Lagos. They showed that some residential areas have higher $PM_{2.5}$ levels due to contributions from local sources and transport of pollutants from neighbouring sites more than industrial areas. Common anthropogenic sources for $PM_{2.5}$ in their study include traffic-related, traffic emissions, marine salt, and industrial emissions based on the principal component factor analysis (PCFA) of the detected elements. More recently, Zhen et al. (2019) reported the chemical characteristics of $PM_{2.5}$ based on $PM_{2.5}$ mass concentrations, OC, EC, water-soluble ions and elemental compositions. According to their results, $PM_{2.5}$ pollution was higher in Lagos than in Hong Kong. They also found that vehicular emissions contributed the most to $PM_{2.5}$ pollution.

Aliphatic hydrocarbon compounds have been used in other parts of the world to identify the main sources of $PM_{2.5}$ pollution and to understand the important impact of regional transport on the characterisation of $PM_{2.5}$ (Andreou et al., 2007; Javed et al., 2019; Sun et al., 2021). For instance, Andreou et al. (2007) investigated the organic chemical composition of $PM_{2.5}$ in Athens to determine the emission sources. They found that all samples in Athens contained n-alkanes, biomarkers for fossil fuels, and a mixed origin (petrogenic and biogenic) of Athenian $PM_{2.5}$. Vehicular emissions were the main source of aliphatic hydrocarbon compounds, with biogenic sources contributing less. Recently, Sun et al. (2021) determined the sources of $PM_{2.5}$-associated PAHs and n-alkanes in Changzhou, China. Their results showed that biogenic sources are the main source of n-alkanes and PAHs in $PM_{2.5}$. The authors also observed that variations in the concentration of n-alkanes and PAHs from different air mass transports were not consistent with the changes in $PM_{2.5}$ mass, and concluded that regional transport has important effects on the characterisation of $PM_{2.5}$. This study therefore aimed at determining the $PM_{2.5}$ mass concentration and characterising the chemical composition of $PM_{2.5}$ collected at a residential location in Agege area of Lagos, Nigeria. The chemical analysis included the elemental carbon (EC), organic carbon (OC), aliphatic hydrocarbon compounds comprising of n-alkanes, hopanes and steranes. The impact of local and regional air mass transports on the amount of $PM_{2.5}$, elemental carbon (EC), organic carbon (OC), and aliphatic hydrocarbon compounds arriving at the study site was also assessed using the air mass trajectory analysis. In addition, the sources of $PM_{2.5}$ pollution were identified based on the molecular diagnostic indices of n-alkanes, hopanes and steranes profiles. It is expected that this work will provide useful information on the characteristic nature of $PM_{2.5}$ and in particular the sources and profiles of aliphatic hydrocarbons in $PM_{2.5}$ in residential areas of Lagos, which is currently scarce.

Materials and methods

Description of sampling site

The sampling site was an urban residential area located at 3/5 Morenike Carena Close, Orile Agege, Lagos (6.635008N and 3.302543E). The site was located within the city, and thus the anthropogenic emissions from various sources was quite high. Lagos is the former administrative capital of Nigeria, and the biggest metropolis in West Africa with a population of over 21 million (Alani et al., 2019). It remains the economic and industrial hub of the nation, where a range of activities are taking place that might be detrimental to the city's air quality. Lagos is susceptible to particulate matter pollution arising from vehicular traffic, diesel and gasoline generator emissions, open dumpsites, illegal waste burning, infrastructure construction, and household cooking using polluting fuel and stoves (Ibitayo 2012; Oseni 2016; Adegboye 2018; Adam 2018; Ozoh et al., 2018).

Sample collection

Thirty-six (36) $PM_{2.5}$ samples (24-h samples) and blank samples (4) were taken between June and July 2021 during the peak of the rainy season to provide an overview of the extent of the impact of the rainy season on the $PM_{2.5}$ mass concentrations and aliphatic hydrocarbon compounds characteristics in $PM_{2.5}$. Samples were collected on high-purity quartz fibre filters (Φ = 47 mm) using a low-vol "Gent" stacked $PM_{2.5}$ air sampler (Maenhaut et al., 1994). The device was placed at a height of 4.65 m above the ground to minimise blockage of air parcels from surrounding buildings and was operated at a constant flow rate of 18 L/min. The samples were stored in a refrigerator at 5°C and subsequently cooled in a container packed with frozen ice packs and transported to TROPOS in Leipzig, Germany within 24 h. Upon arrival, the samples were stored in the freezer at -20 °C prior to laboratory analysis.

Gravimetric and organic/elemental carbon (OC/EC) analysis

The mass of the $PM_{2.5}$ samples were quantified using a microbalance (Mod. AT261 Delta Range, Mettler-Toledo) after being allowed to equilibrate in the weighing room for 72 h under a climate-controlled conditions (temperature: 20 ± 1 °C and relative humidity: 50 ± 5%) before and after sampling. To determine the mass concentration, the weight difference of the filters before and after sampling was computed and divided by total sampling volume. Organic carbon and elemental carbon were analysed using a thermo-optical method (Sunset Laboratory Inc., USA) at a maximum temperature of 850 °C using the normalised temperature programme EUSAAR2 (EUropean Supersites for Atmospheric Aerosol Research) as described in the literature (Cavalli et al., 2010; Yttri et al., 2019). The method is in accordance with the standard proposed by the European networks (ACTRIS, EMEP). Samples were thermally desorbed from the filter medium under an inert He atmosphere followed by an oxidising O_2-He atmosphere, under carefully controlled heating ramps. A flame ionisation detector was used to quantify methane after catalytic methanation of CO_2. With the EUSAAR2 protocol, different OC fractions in an inert atmosphere (He) were measured at 200°C for 120s, at 300°C for 150s, at 450°C for 180s and at 650°C for 180s. Subsequently, the EC fractions were measured in an oxidising atmosphere (O_2-He) at 500°C for 120s, at 550°C for 120s, at 700°C for 70s and at 850°C for 80s (Cavalli et al., 2010). Lower OC/EC ratios are usually the result of charring processes' tendency to overestimate the EC and underestimate the OC. In order to adjust for the charring process, an optical correction was used. A laser with a wavelength of 678 nm was used to measure the sample's transmission in order to calculate the optical correction for charring for pyrolytic carbon. For OC and EC measurements, the detection threshold was 0.2 µg cm^{-2}.

Turpin and Lim (2001) evaluated organic matter (OM) using the conversion factor fOM/OC. The organic matter was around twice as much as the organic carbon (OM = 2.1 × OC). Because the conversion factor varies depending on the specific conditions at each site, the factor fOM/OC = 2.1 is recommended as it also takes the age of the aerosol into account (Turpin and Lim, 2001). Thus, the OM in the present study was estimated accordingly. The POC content can be estimated using EC as a tracer by multiplying the minimum OC/EC ratio determined for the entire study period by the EC content as follows (Huang et al., 2012; Deabji et al., 2021):

$$\mathrm{POC} = (\mathrm{OC/EC})_{\mathrm{min}} \times \mathrm{EC} \quad (1)$$

Consequently, the amount of SOC that contributes to total organic carbon can be determined as the difference between the concentrations of primary and secondary organic carbon using the following equation (Huang et al., 2012; Deabji et al., 2021):

$$\mathrm{SOC} = \mathrm{OC} - \mathrm{POC} \quad (2)$$

Organic compounds analysis

The analysis of aliphatic hydrocarbon compounds such as n-alkanes (C_{20}-C_{34}), hopanes, and steranes were carried out using a Curie-point pyrolyser (JPS-350, JAI Inc., Japan) coupled to a gas chromatography-mass spectrometry system (6890 N GC, 5973 inert MSD, Agilent Technologies, CA, USA). A detailed description of the analytical procedures has been previously documented (Neusüss et al. 2000). Briefly, filter aliquots were spiked with a solution of internal standards, including perdeuterated tetracosane-d_{50} (C_{24}-D_{50}) and dotriacontane-d_{66} (C_{32}-D_{66}). Two circular filter punches (6 mm; 56.5 mm^2) from each sample were then coated with an alloy of iron and nickel (50% Fe; 50% Ni) and immediately evaporated in a pyrolysis chamber at 510 °C for 4 s in a helium environment. The compounds were separated using a capillary column coated with a stationary phase of 5 % phenyl-arylene and 95 % dimethylpolysiloxane (ZB-5MS, 30 m × 0.25 mm i.d. × 0.25 µm film thickness). The GC and MS conditions had been previously described by Khedidji et al. (2020). Individual compounds were identified by comparing the mass spectra with literature and library data and validated with standard compounds. Internal calibration was performed using the respective internal standards. The GC-MS data were acquired using dedicated software (ChemStation) from Agilent. For each compound analysed, the sample concentration was calculated by subtracting the average concentration of the blank filter from the measured concentration in the sample.

Molecular diagnostic ratios

To investigate the origin of the n-alkanes in the $PM_{2.5}$ samples, the following diagnostic ratios were used.

The carbon preference index (CPI): CPI is a diagnostic technique that depicts the relationship of proportionality between alkanes with odd and even carbon chains in a given sample. Based on several studies (Bray and Evans, 1961; Simoneit et al.,1991; Boreddy et al., 2018), the CPI for n-alkanes (odd to even ratio) in the $PM_{2.5}$ was calculated as follows:

$$\mathrm{CPI} = \frac{\sum \mathrm{odd}\ (\mathrm{C21} - \mathrm{C33})}{\sum \mathrm{even}\ (\mathrm{C20} - \mathrm{C34})} \quad (3)$$

The average chain length (ACL): ACL is based on the prevalence of odd high homologs. It is estimated as the mean number of carbon atoms per molecule (Caumo et al., 2020). ACL has been used to identify the origins of n-alkanes in a given sample (Kawamura et al., 2003; Boreddy et al., 2018; Caumo et al., 2020), and based on these studies, the ACL for n-alkanes in $PM_{2.5}$ was obtained according to the following equation:

$$\mathrm{ACL} = \frac{[25(\mathrm{C25}) + 27(\mathrm{C27}) + 29(\mathrm{C29}) + 31(\mathrm{C31}) + 33(\mathrm{C33})]}{[\mathrm{C25} + \mathrm{C27} + \mathrm{C29} + \mathrm{C31} + \mathrm{C33}]} \quad (4)$$

Odd-even predominance (OEP): The odd-even computation also known as the odd-even carbon number by dividing the concentration of odd-numbered n-alkanes by the concentration of even-numbered n-alkanes as follows:

$$\mathrm{OEP} = \frac{\mathrm{C21} + 6(\mathrm{C23}) + \mathrm{C25}}{4(\mathrm{C22} + \mathrm{C24})} \quad (5)$$

Air mass back trajectory

96-hour air mass back trajectories for each sample collection time interval were determined using the HYSPLIT model version 5 (www.ready.noaa.gov/HYSPLIT_traj.php, last use February 2023, Stein et al., 2015) to evaluate the origin of the $PM_{2.5}$ during the sampling period. The choice of using 96-h was to provide a representation for fine mode particles ($PM_{2.5}$) that have longer atmospheric lifetimes as coarse mode particles. The Global Data Assimilation System meteorological data set was used for the HYSPLIT trajectory calculations with a 1°-degree grid size. The obtained trajectories were further processed using the r studio software open-air package (https://davidcarslaw.github.io/openair) for graphical presentation. The computed trajectories also contained information of the air mass history, its residence time (average time the air parcel stays over a given grid cell or geographical landscape during its transport prior to arrival at the receptor site) over given geographical landscapes, different mixing depts (height at which the potential temperature of the air mass starts to change), radiation and general meteorology information that the air mass experienced during its transport. Using the trajCluster function of the open-air r package, a cluster analysis of the total determined trajectories was performed. Here, the trajectories were grouped according to the least Euclidian distance between the trajectory's points. After several iterations amongst pairs of trajectories, the average trajectory representative of the minimum distance between the trajectories with a low total spatial variance (TSV) was determined and assigned as a cluster. The equation applied to determine the Euclidian distance (d (1,2)) between two points of two trajectories 1 and 2 at positions (X1, Y1) and (X2, Y2) respectively, is given as (Stein et al., 2015):

$$d_{1,2} = \left(\sum_{i=1}^{n}((X_{1i} - X_{2i})^2 + (Y_{1i} - Y_{2i}))^2\right)^{1/2} \quad (6)$$

The selection of the final number of clusters involves a mathematical process wherein the total spatial variance is compared with the model estimated number of clusters. The level at which a change in the number of clusters significantly increases the TSV is considered as the point of the optimum number of clusters for the total evaluated back trajectories. In this study, 5-cluster solution provided a reasonable spatial representation of the entire computed trajectories and was considered the best solution of this analysis. The clusters were labelled C1 to C5 and were assigned to each sample. The samples chemical composition including their $PM_{2.5}$ mass was averaged according to their respective clusters and the relative contributions of the clusters to the chemical components were evaluated.

Results and discussion

$PM_{2.5}$ mass concentration and its carbonaceous components

Table 1 summarises the 24-h average $PM_{2.5}$ mass concentrations and its carbonaceous chemical components for the sampling period (June to July 2021). The 24-h $PM_{2.5}$ mass concentrations ranged from 8.7 to 56.5 μg m^{-3} (avg. 27 ± 11.1 μg m^{-3}), surpassing the daily threshold of 15 μg m^{-3} recommended by World Health Organisation (WHO, 2021). The OC concentration ranged from 1.0 to 4.9 μg m^{-3} (avg. 2.2 ± 1.0 μg m^{-3}). On average, OC accounted for 8% of the total $PM_{2.5}$ mass and contributed 73% of the carbonaceous aerosol while the majority of OC consisted of POC (68%), indicating that the OC was a crucial component of $PM_{2.5}$ and the primary emission sources of OC in the study area need to be controlled. The EC concentration ranged from 0.4 to 1.8 μg m^{-3} (avg. 0.8 ± 0.3 μg m^{-3}). On average, EC constituted about 3% of the total $PM_{2.5}$ mass. Though this value could be viewed as only a small portion of ambient $PM_{2.5}$, EC has been recognised as a crucial sign of unfavourable health impacts (Janssen et al., 2011). With a correlation value of r^2 = 0.81 (Fig. 1a), OC and EC demonstrate a strong relationship, indicating that the majority of the OC at the site was produced as a primary aerosol with EC. This revealed that EC and OC, as well as the precursor that resulted in SOC formation, could have an identical source. The positive bi-modal correlation (r^2 = 0.76 and 0.40) between SOC and EC indicated that the primary combustion sources have an influence on the generation of SOC (Fig. 1b), and also suggested two distinct associations such as sources or atmospheric processes linking the two different carbonaceous components. The OC/EC ratios ranged from 2 to 5.1 μg m^{-3} (avg. 2.9 ± 0.7 μg m^3). Hildemann et al. (1991), reported 2.2 μg m^{-3} for light-duty gasoline vehicles while Cao et al. (2005) observed 4.1 μg m^{-3} for vehicle exhaust. This suggested that the most probable common source of $PM_{2.5}$ emission in the area was gasoline-powered engines. Generally, OC/EC ratio > 2 is an indicator of SOC formation in the atmosphere (Javed et al., 2019).

Table 1: *Summary of 24-h average $PM_{2.5}$ and carbonaceous constituents (μg m^3) in Lagos, Nigeria*

	Minimum	Maximum	Average	Standard deviation
$PM_{2.5}$ mass	8.7	56.5	27.0	11.1
EC	0.4	1.8	0.8	0.3
OC	1.0	4.9	2.2	1.0
OC/EC	2.0	5.1	2.9	0.7
POC	0.8	3.7	1.5	0.7
SOC	0.0	1.7	0.6	0.5
OM	2.2	10.3	4.5	2.1

n-alkanes in $PM_{2.5}$

The concentrations of aliphatic hydrocarbons detected in the $PM_{2.5}$ samples from the residential area in Lagos and the molecular diagnostic indices are presented in Table 2. The aliphatic fraction of the $PM_{2.5}$ was dominated by n-alkanes from C_{20} to C_{34}, and maximizing at C_{25}, followed by C_{26} and C_{20}, respectively (Fig. 2). The total concentrations of the n-alkanes $\sum(C_{20}\text{-}C_{34})$ in the samples ranged from 1.8 to 146.6 ng m^{-3} (avg. 43.6 ± 35.1 ng m^{-3}) (Table 2). On average, n-alkane contributed 0.2% and 1.0% of the total $PM_{2.5}$ mass and OM budget,

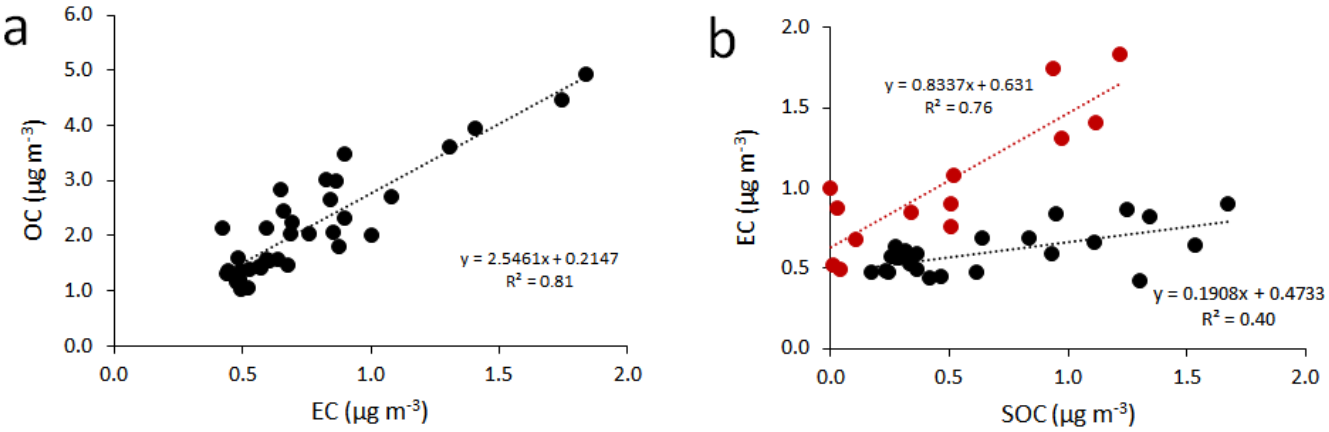


Figure 1: *(a, left) Plot of OC versus EC, and (b, right) EC versus SOC showing the relationship between the parameters. Samples with the same colour indicate a similar association (source or atmospheric processes).*

Table 2: *Aliphatic hydrocarbon concentrations (ng m^{-3}) and molecular indices of n-alkanes in $PM_{2.5}$ from a residential area in Lagos, Nigeria*

Compound	Minimum	Maximum	Average	Standard deviation
∑n-alkanes (C_{20}-C_{34})	1.8	146.6	43.6	35.1
Indices				
CPI	0.0	2.3	0.7	0.5
ACL	25.0	30.8	25.7	1.6
OEP	0.0	1.9	0.5	0.5
Hopanes				
17α(H), 21β(H)-30-Norhopane	0.0	22.1	4.4	7.0
17α(H), 21β(H)-Hopane	0.0	31.7	5.6	8.9
17α(H), 21β(H)-22S-Homohopane	0.0	89.9	2.9	15.1
17α (H), 21β(H)-22R-Homohopane	0.0	14.1	1.9	3.7
$\sum_4$Hopanes	0.0	89.9	18.7	25.9
Steranes				
α, α, α 20R-Cholestane	0.0	3.1	0.3	0.8
α, β, β (20R 24S)-24-Ethylcholestane	0.0	15.1	1.7	3.9
α, α, α (20R 24S)-24-Ethylcholestane	0.0	7.9	1.3	2.1
$\sum_3$Steranes	0.0	26.0	3.3	6.7

respectively, indicating a sizeable portion of the ambient $PM_{2.5}$ and organic matter concentration in the area. A comparable range of total n-alkanes concentrations in $PM_{2.5}$ were recently found in densely populated and traffic-intensive city of Beijing, China (4.51-153 ng m^{-3}) (Yang et al., 2023), but higher than the average total concentrations recorded in $PM_{2.5}$ in a residential/commercial area in Doha, Qatar (8.04 ± 7.74 ng m^{-3}) (Javed et al., 2019), an urban residential area (Aglantizia) in Nicosia, Cyprus (13 ± 12 ng m^{-3}) (Iakovides et al., 2021), and roadsides in Lisbon, Portugal (22.4 ± 17.9 ng m^{-3} in winter and 23.3 ± 2.83 ng m^{-3} in summer) (Alves et al., 2016). In contrast, the average total concentration of n-alkanes obtained in this work was lower than some urban cities in China (Changzhou: 252.37 ± 184.02 ng m^{-3}; Nanjing: 205 ng m^{-3}; Changchun: 209 ng m^{-3}; Shanghai: 259 ng m^{-3} in $PM_{2.5}$) (Wang et al., 2006; Haque et al., 2019; Sun et al., 2021) and Hong Kong (195 ng m^{-3}) (Wang et al., 2006). The prevalence of low carbon number (C_{20}-C_{26}) in the aerosol particles showed a predominance of anthropogenic sources, such as incomplete combustion of fossil fuels (Zhang et al., 2018). Fossil fuel-derived n-alkanes with the dominant of C_{25} and C_{20} homologs are considered to be from gasoline engines while C_{20}-dominated n-alkanes are regarded as from heavy and medium-duty diesel engines (Rogge et al., 1993; Schauer et al., 1999). The preponderance of C_{25} and C_{20} in the samples (Fig. 2) showed that anthropogenic emissions resulting from gasoline and diesel engines were probably the dominant sources of n-alkanes in the area under study.

The carbon preference index (CPI) values calculated from n-alkane distribution have been extensively employed as

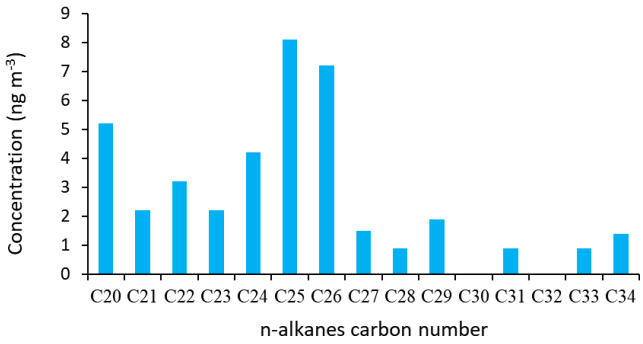


Figure 2: *Average concentrations of n-alkanes in the $PM_{2.5}$ samples.*

a source marker in aerosol samples (Simoneit et al., 1991; Andreous et al., 2008; Alves, 2008; Alves et al., 2016; Boreddy et al., 2018; Sun et al., 2021). The CPI values obtained from n-alkane distribution ranged from 0.0 to 2.3 (avg. 0.7 ± 0.5) (Table 2). CPI values close to 1 were attributed to pollution from petrogenic sources, while values above 3 were attributed to biogenic sources (Omar et al., 2007; Alves, 2008). Based on the average CPI value determined in this work, the n-alkanes in $PM_{2.5}$ originated mainly from a petrogenic source. The ACL has also been effectively used to pinpoint aerosol pollution sources (Boreddy et al., 2018; Caumo et al., 2020). The ACL index for n-alkanes in the present $PM_{2.5}$ samples ranged from 25.0 to 30.8 (avg. 25.7 ± 1.6) (Table 2); the wide variation in the index suggested that the primary source of n-alkanes in $PM_{2.5}$ stemmed from petrogenic contamination (Jeng, 2006). The OEP is an efficient tool that can give details about the various n-alkane sources (Ladji et al., 2009). The OEP results computed from the n-alkanes in the $PM_{2.5}$ samples ranged from 0.0 to 1.9 (avg. 0.5 ± 0.5) (Table 2). In general, OEP values of less than 1.3 are attributed to contamination by crude oil (Scalan and Smith, 1970). The low average OEP value suggested aerosol pollution arising from petroleum sources (Scalan and Smith, 1970; Sojinu et al., 2012).

Hopanes and steranes in $PM_{2.5}$

The concentrations of hopanes and steranes detected in the $PM_{2.5}$ samples during the entire campaign period are listed in Table 2. Four (4) hopanes including 17α(H),21β(H)-30-norhopane, 17α(H),21β(H)-hopane, 17α(H),21β(H)-22S-homohopane, and 17α(H),21β(H)-22R-homohopane and three (3) steranes including α, α, α 20R-Cholestane, α, β, β (20R 24S)-24-Ethylcholestane and α, α, α (20R 24S)-24-Ethylcholestane, were detected in the $PM_{2.5}$ samples (Table 2). The total concentrations of hopanes (∑4 hopanes) and steranes (∑3 steranes) varied from 0.0 to 89.9 ng m^{-3} (avg. 18.7 ± 25.9 ng m^{-3}) and 0.0 to 26.0 ng m^{-3} (avg. 3.3 ± 6.7 ng m^{-3}), respectively, in $PM_{2.5}$. On average, hopanes accounted for 0.1% and 0.4% of the total $PM_{2.5}$ and OM, while steranes contributed 0.01% and 0.1% of the total $PM_{2.5}$ and OM, respectively, making them a non-negligible part of the organic matter content in the study area. The average total hopanes and steranes concentrations in this study were higher than those reported in $PM_{2.5}$ in a residential/commercial area in Doha, Qatar (0.521 ± 0.677 ng m^{-3}) and (0.162 ± 0.25 ng m^{-3}), respectively (Javed et al., 2019). Similarly, lower average hopanes and steranes concentrations were observed in an urban residential area in Nicosia, Cyprus (0.83 ± 1.04 ng m^{-3}) and (0.27 ± 0.39 ng m^{-3}), respectively. The average hopanes level in this study was likewise higher than those observed in roadsides areas in Lisbon, Portugal in winter (1.71 ± 1.55 ng m^{-3}), but comparable to those observed in roadside areas in Lisbon, Portugal in summer (3.23 ± 1.15 ng m^{-3}) (Alves et al., 2016). The presence of hopanes and steranes in $PM_{2.5}$ samples suggested fossil fuels combustion in the area (Simoneit et al., 2004; Javed et al., 2019). The steranes were dominated by α, β, β (20R 24S)-24-Ethylcholestane (Table 2). The stereochemical configurations at the C17 and C21 positions of hopanes are frequently used to assess the thermal maturity levels of fossil fuels, and have proved useful in evaluating the source of hopanes in aerosol studies (Simoneit et al., 2004; Barakat, 2002; Mikuška, Křůmal, and Večeřa 2015; Javed et al., 2019). Hopanes with configurations of 17β(H), 21β(H) are considered immature; 17β(H), 21α(H) are of low maturity while 17α(H), 21β(H) are fully mature (Mikuška, Křůmal, and Večeřa 2015; Javed et al., 2019). The main homologs of hopanes detected in the samples were the thermodynamically more stable 17α(H), 21β(H) configurations, with a preponderance of the 17α(H), 21β(H)-hopane compound (C30αβ; 36% of total hopanes). This was followed by 17α(H), 21β(H)-norhopane (C29αβ; 28%) (Fig. 3), a fingerprint for vehicle emissions (Mikuška, Křůmal, and Večeřa 2015; Javed et al., 2019). The dominance of αβ configuration indicated that atmospheric contamination in the study area emanated from matured petroleum. Again, the higher concentration of the 22S hopane relative to the corresponding 22R epimer (Table 2; Fig. 3), further supported the fact that the hopanes were primarily derived from gasoline and diesel engine exhaust (Rushdi et al. 2017; Javed et al., 2019). The enhanced levels of these petroleum biomarkers (Barakat, 2002; Mikuška, Křůmal, and Večeřa 2015; Javed et al., 2019), indicated a significant impact of petrogenic source emission in the study area.

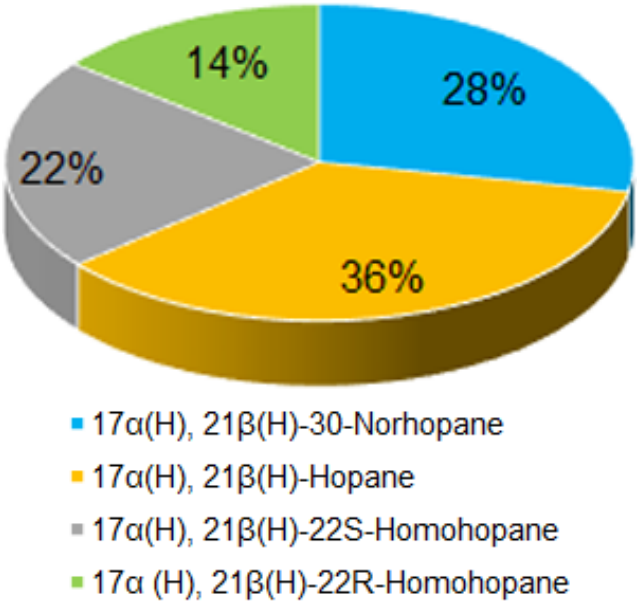


Figure 3: *The percentage composition of hopanes in the $PM_{2.5}$.*

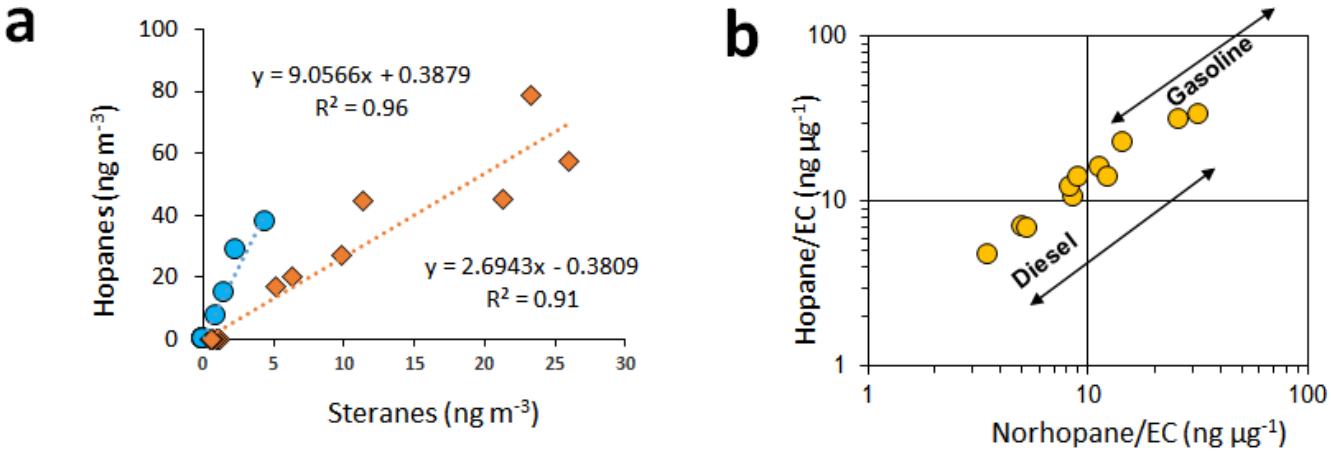


Figure 4: *(a, left) Cross plot of hopanes versus steranes. Samples with the same colour indicate a similar association (source or atmospheric processes) between hopanes and steranes in Fig. 4a, and (b, right) Ratio-ratio plot of hopane and norhopane normalised by EC (Modified after Yu et al., 2011).*

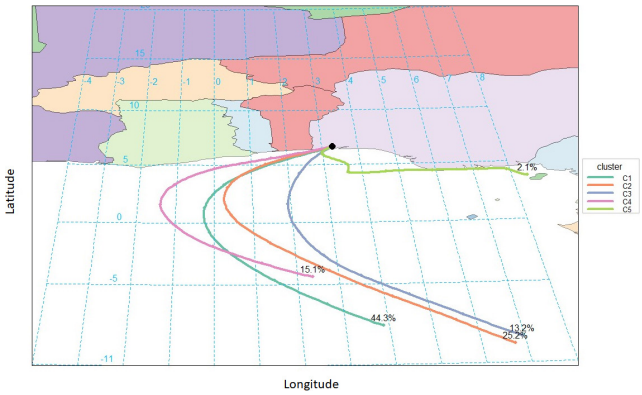


Figure 5: *96-hours backward air mass trajectories arriving at the sampling site during the sampling period in the rainy season.*

Two strong positive correlations (r^2 = 0.96 and 0.90) were observed between hopanes and steranes (Fig. 4a), indicating two separate associations such as sources or atmospheric processes linking the two different types of compounds. Hopanes and EC have been used to allocate gasoline and diesel emissions (Yu et al., 2011). Diesel engines are strong contributors to both EC and hopanes, whereas gasoline engines are huge sources of hopanes but only contribute moderate amounts of EC. As a result, diesel-powered engines have substantially lower hopanes to EC ratios than engines that run on gasoline (Yu et al., 2011). The ratio-ratio plot of hopane (17α(H),21β(H)-hopane) and norhopane (17α(H),21β(H)-30-norhopane) obtained for $PM_{2.5}$ in this study, normalised by EC is shown in Fig. 4b, modified from Yu et al. (2011). The plot corroborated the sources of hopanes as being from diesel and gasoline engines due to the strong correlation between the groups of compounds, which might have emanated from vehicle emissions, motorbikes, and fumes emitted by diesel and gasoline engines.

Air mass source regions

As shown in Fig. 5, air masses originated from different regions of the Atlantic Ocean crossing through various densely populated areas in the coastal settlements in Lagos at different altitudes before arriving at the site. The clusters C1 and C2, represent 44% and 25%, respectively, of the total air mass inflow during the sampling period and were characterised by their different origins, but with a similar profile in the mixing depth and trajectory heights (air masses altitudes) close to their arrival at the sampling site. C3 (13%) and C4 (15%) were representative of air masses that spent more time over the Atlantic Ocean and little time over terrain with C4 passing closer to the coast of neighbouring Benin republic while C3 was confined within the coast of Nigeria and originated from the Gulf of Guinea region. C5 made up only about 2% of the trajectories but represented air masses originating from the coastal area of Nigeria with higher residence time over land and vegetation regions with low mixing depth. These characteristics made it contributed significantly to the composition of the particles at the sampling site.

Influence of source regions on $PM_{2.5}$ mass and carbonaceous components

Fig. 6 gives the quantified contributions of the different directions of air masses to the $PM_{2.5}$ mass and its carbonaceous components. Interestingly, air mass for cluster 5 (C5) with few episodes of air transports contributed more to $PM_{2.5}$ mass (22%), EC (21.4%), OC (24.7%), OC/EC (23%), POC (21.4%), SOC (32%), and OM (24.7%) (Fig. 6a), as compared to air masses for clusters 1 to 4 (C1 to C4) which passed mainly through the Atlantic Ocean before crossing the mainland to the sampling site. Cluster 5 after emerging from the sea has a relatively long residence time in the continental coastal areas of the Atlantic Ocean before returning to the marine area and eventually crossing the mainland to the sampling site. The relatively low mixing depth of the air mass, high exposure to different continental environments and natural vegetation during transports may have raised the levels of the $PM_{2.5}$ mass and its chemical components in the C5 trajectory (Fig. 6b).

Influence of source regions on aliphatic hydrocarbon compounds

The contributions of the different direction of air mass to aliphatic hydrocarbon compounds are shown in Fig. 6. As previously observed in the carbonaceous materials and $PM_{2.5}$ mass

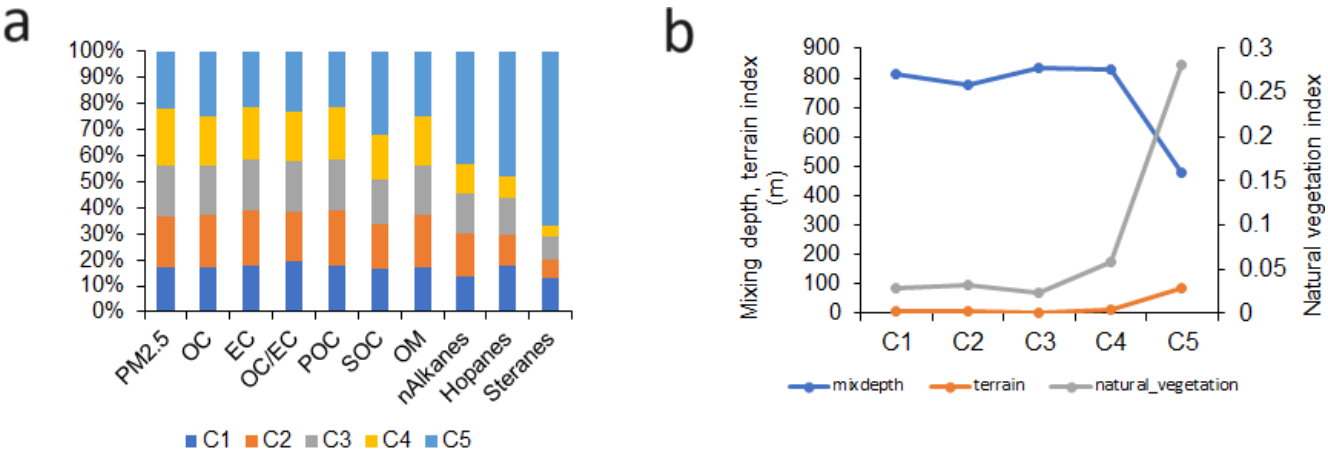


Figure 6: *(a, left) Plot showing the various contributions of different clusters to $PM_{2.5}$ mass concentration, its carbonaceous components, and aliphatic hydrocarbons, (b, right) Plot showing the average mixing depth, terrain, and natural vegetation of the air mass trajectories.*

concentrations, cluster C5 with few episodes also contributed significantly to the budgets of the aliphatic hydrocarbon compound classes as compared to the other clusters. C5 air mass contributed 70%, 50%, and 43% of steranes, hopanes, and n-alkanes concentrations in this region, respectively. The contributions of C1 and C3 were almost similar for the three classes ranged between 12% and 18% for C1 and 9% to 15% for C3. This was likely due to the similar trajectory routes the air masses experienced before arriving at the sampling site. C4 contributed the least to the aliphatic compounds with relative contributions ranged from 3% to 6% for steranes and n-alkanes, respectively. High impact of steranes by long distant transport (e.g., C5) indicated the stability of these compounds in the atmosphere. The differences in the cluster contributions and the stronger dominance of the C5 air mass can be ascribed to the relatively low mixing depth of the air mass, less residence time over the ocean, and high exposure to different continental environments as well as natural vegetation during the air mass transport in comparison to the C1 to C4 clusters. It was also interesting to observe the similar contribution of C1 across the components, indicating a constant influence of local sources in this air mass trajectory. Transport over continental and vegetated regions allowed local emissions to interact with the transported air masses, resulting in enhanced levels of these compounds in such air mass trajectories.

Conclusion

The total concentrations of ∑n-alkanes (C_{20}-C_{34}), hopanes ($\sum_4$Hopanes), and steranes ($\sum_3$Steranes) determined from $PM_{2.5}$ samples collected from a residential area of Lagos ranged from 1.8 to 146.6 ng m^{-3}, 0.0 to 89.9 ng m^{-3}, and 0.0 to 26.0 ng m^{-3}, amounting to 0.2%, 0.1%, and 0.01% of the total $PM_{2.5}$ mass concentrations, and 1%, 0.4% and 0.1% of the organic matter contents, respectively. N-alkanes diagnostic indices, hopanes, and steranes profiles showed that gasoline and diesel engines were the major contributors to particulate matter pollution in the studied area. Therefore, controlling gasoline and diesel emission sources are key steps to minimising air pollution in the area. Based on the air mass trajectories, the levels of $PM_{2.5}$, carbonaceous materials, and aliphatic hydrocarbon compounds arriving at the study site have been considerably impacted by both local and regional air transports. This study could help decision-makers and environmental stakeholders to develop specific, focused, and efficient management strategies for mitigating local air pollution challenges in Lagos, Nigeria.

Author contribution

OLF: Conceptualisation, Performed the fieldwork, Formal analysis, and interpretation, Writing - original draft. HH and KWF: Methodology, Resources, Supervision, Writing – review & editing. GCE: Resources. All the authors contributed to the final version of this paper.

Acknowledgment

We are very grateful to the anonymous reviewers for their helpful suggestions, which helped to improve the original version of this manuscript. We also thank the Alexander von Humboldt (AvH) Foundation for granting the first author (OLF) a Humboldt Research Fellowship for Postdoctoral Researchers towards this research. The laboratory assistant of Sylvia Haferkorn of TROPOS ACD is greatly appreciated. GCE acknowledges the Centre for Energy Research & Development, Obafemi Awolowo University for their all-round support.

Note

An earlier version of this paper was presented at the International Conference on Air Quality in Africa (ICAQ' Africa 2022) in October 2022 and was published in its Proceedings.

References

Adama O, 2018, 'Urban Imaginaries: Funding Mega Infrastructure Projects in Lagos, Nigeria', GeoJournal, 83, 2, 257-274, doi. org/10.1007/s10708- 016-9761-8.

Adegboye K, 2018, "Why Problem of Waste Management Persists in Lagos." Vanguard. https://www.vanguardngr.com/2018/10/why-problem-of-waste-management-persists-in-lagos-stakeholders/.

Alani R.A, Ayejuyo O.O, Akinrinade O.E, Badmus G.O, Festus C.J, Ogunnaike B.A, Alo B.I, 2019, 'The level $PM_{2.5}$ and the elemental compositions of some potential receptor locations in Lagos, Nigeria', Air Quality, Atmosphere and Health, 12,1251-1258, doi.org/10.1007/s11869-019-00743-3.

Alves C.A, 2008, 'Characterisation of solvent extractable organic constituents in atmospheric particulate matter: An overview', Anais da Academia Brasileira de Ciências, 80, 21-82.

Alves C.A, Oliveira C, Martins N, Mirante F, Caseiro A, Pio C, Matos M, Silva H.F, Oliveira C, Camões F, 2016, 'Road tunnel, roadside, and urban background measurements of aliphatic compounds in size-segregated particulate matter', Atmospheric Research, 168:139-148, doi.org/10.1016/j.atmosres.2015.09.007.

Andreou G, Alexiou S.D, Loupa G, Rapsomanikis S, 2008, 'Identification, Abundance and Origin of Aliphatic Hydrocarbons in the Fine Atmospheric Particulate Matter of Athens', Greece, Water, Air, and Soil Pollution: Focus, 8, 99-106, doi.org/10.1007/s11267-007-9142-x.

Barakat A.O, 2002, 'PAHs and petroleum markers in the atmospheric environment of Alexandria City, Egypt' Water, Air, and Soil Pollution 139, 289-310, doi.org/10.1023/A:1015894520672.

Boreddy S.K.R, Haque M.M, Kawamura K, Fu P, Kim, Y, 2018, 'Homologous series of n-alkanes (C_{19}-C_{35}), fatty acids (C_{12}-C_{32}) and alcohols (C_8-C_{30}) in atmospheric aerosols from central Alaska: Molecular distribution, seasonality and source indices', Atmospheric Environment, 184, 87-97, doi.org/10.1016/j.atmosenv.2018.04.021.

Bray E.E, Evans E.D, 1961, 'Distribution of n-paraffins as a clue to recognition of source beds', Geochimica Cosmochimica, Acta 22, 2-15, doi.org/10.1016/0016-7037(61)90069-2.

Brown S.G, Herckes P, Ashbaugh L, Hannigan M.P, Kreidenweis S.M, Collett J.L, 2002, 'Characterization of organic aerosol in Big Bend National Park, Texas', Atmospheric Environment, 36, 5807-5818. doi.org/10.1016/S1352-2310(02)00578-2.

Cao J.J, Wu F, Chow J.C, Lee S.C, Li Y, Chen S.W, An Z.S, Fung K.K, Watson J.G, Zhu C.S, Liu S.X, 2005, 'Characterization and source apportionment of atmospheric organic and elemental carbon during fall and winter of 2003 in Xi'an, China', Atmospheric Chemistry and Physics, 5, 3127-37. doi.org/10.5194/acp-5-3127-2005.

Caumo S, Bruns R.E, Vasconcellos P.C, 2020, 'Variation of the Distribution of Atmospheric n-Alkanes Emitted by Different Fuels' Combustion', Atmosphere, 11, 643, doi:10.3390/atmos11060643.

Cavalli F, Viana M, Yttri K.E, Genberg J, Putaud J.P, 2010, 'Toward a standardised thermal-optical protocol for measuring atmospheric organic and elemental carbon: the EUSAAR protocol' Atmospheric Measurement Techniques, 3, 1, 79–89, doi.org/10.5194/amt-3-79-2010.

Deabji N, Fomba K.W, El Hajjaji S, Mellouki A, Poulain L, Zeppenfeld S, Herrmann H, 2021, 'First insights into northern Africa high-altitude background aerosol chemical composition and source influences', Atmospheric Chemistry and Physics, 21, 18147-18174, doi.org/10.5194/acp-21-18147-2021.

Ezeh, G.C. Obioh I.B, Asubiojo O.I, Chiari M, Nava S, Calzolai G, Lucarelli F, Nuviadenu C.K., 2014, 'Elemental compositions of $PM_{10-2.5}$ and $PM_{2.5}$ aerosols of a Nigerian urban city using ion beam analytical techniques', Nuclear Instruments and Methods in Physics Research B, 334, 28-33, doi.org/10.1016/j.nimb.2014.04.022.

Ezeh G.C, Obioh I.B, Asubiojo O.I, Onwudiegwu C.A, Nuviadenu C. K, Ayinla S.A, 2018, 'Airborne fine particulate matter ($PM_{2.5}$) at industrial, high- and low-density residential sites in a Nigerian megacity', Toxicological and Environmental Chemistry, 100, 3, 326-333, doi.org/10.1080/02772248.2018.1485924.

Ezeh G. C, Obioh I.B, Asubiojo O.I, Abiye O.E, 2012, 'PIXE characterization of PM_{10} and $PM_{2.5}$ particulates sizes collected in Ikoyi Lagos, Nigeria', Toxicological and Environmental Chemistry, 94, 5, 884-894,doi.org/10.1080/02772248.2012.674133.

Fu P, Kawamura K, Okuzawa K, Aggarwal S.G, Wang G, Kanaya Y, Wang Z, 2008, 'Organic molecular compositions and temporal variations of summertime mountain aerosols over Mt. Tai, North China Plain', Journal of Geophysical Research: Atmospheres, 113, D19107. doi.org/10.1029/2008JD009900.

Gupta S, Gadi R, Mandal T.K, Sharma S.K, 2017, 'Seasonal variations and source profile of n-alkanes in particulate matter (PM_{10}) at a heavy traffic site, Delhi,' Environmental Monitoring and Assessment, 189: 43, doi.org/10.1007/s10661-016-5756-7.

Hildemann L.M, Markowski G.R, Cass G.R, 1991, 'Chemical Composition of Emissions from Urban Sources of Fine Organic Aerosol', Environmental Science and Technology, 25, 744-759, doi.org/10.1021/es00016a021.

Huang H, Ho K.F, Lee S.C, Tsang P.K, Ho, S.S.H, Zou C.W, Zou S.C, Cao J.J, Xu H.M, 2012, 'Characteristics of carbonaceous aerosol in $PM_{2.5}$: Pearl Delta River region, China', Atmosheric Research 104-105, 227–236, doi.org/10.1016/j.atmosres.2011.10.016.

Haque Md. M, Kawamura K, Deshmukh D.K, Fang C, Song W, Mengying B, Zhang Y-L, 2019, 'Characterization of organic aerosols from a Chinese megacity during winter: predominance of fossil fuel combustion', Atmospheric Chemistry and Physics, 19, 5147–5164, 2019, doi.org/10.5194/acp-19-5147-2019.

Iakovides M, Iakovides G, Stephanou E.G, 2021, 'Atmospheric particle-bound polycyclic aromatic hydrocarbons, n-alkanes, hopanes, steranes and trace metals: $PM_{2.5}$ source identification, individual and cumulative multi-pathway lifetime cancer risk assessment in the urban environment', Science of the Total Environment, 752, 141834, doi.org/10.1016/j.scitotenv.2020.141834.

Ibitayo O.O, 2012, 'Towards Effective Urban Transportation System in Lagos, Nigeria: Commuters' Opinions and Experiences', Transport Policy 24,141-147, doi.org/10.1016/j.tranpol.2012.07.009.

Janssen N.A.H, Hoek G, Simic-Lawson M, Fischer P, van Bree L, Brink H.T, Keuken M, Atkinson R.W, Anderson H.R, Brunekreef B, Cassee F.R, 2011, 'Black carbon as an additional indicator of the adverse health effects of airborne particles compared with PM_{10} and $PM_{2.5}$', Environmental Health Perspectives, 119, 1691-1699, doi.org/ 10.1289/ehp.1003369.

Javed W, Iakovides M, Euripides G.S, Wolfson J.M, Koutrakis P, Guo B, 2019, 'Concentrations of aliphatic and polycyclic aromatic hydrocarbons in ambient $PM_{2.5}$ and PM_{10} particulates in Doha, Qatar', Journal of the air and waste management association, 62, 2, 162-177, doi.org/10.1080/10962247.2018.1520754.

Jeng W.-L, 2006, 'Higher plant n-alkane average chain length as an indicator of petrogenic hydrocarbon contamination in marine sediments', Marine Chemistry, 102, 242-251, doi.org/10.1016/j.marchem.2006.05.001.

Kawamura K, Ishimura Y, Yamazaki K, 2003, 'Four years' observations of terrestrial lipid class compounds in marine aerosols from the western North Pacific', Global Biogeochemical Cycles, 17, 1003, doi.org/10.1029/2001GB001810.

Khedidji S, Müller K, Rabhi L, Spindler, G, Fomba K.W, Pinxteren, D.V, Yassaa N, Herrmann H, 2020, 'Chemical Characterization of Marine Aerosols in a South Mediterranean Coastal Area Located in Bou Ismaïl, Algeria', Aerosol and Air Quality Research, 20, 2448–2473, doi.org/10.4209/aaqr.2019.09.0458.

Ladji R, Yassaa N, Balducci C, Cecinato A, Meklati B.Y, 2009, 'Annual variation of particulate organic compounds in PM_{10} in the urban atmosphere of Algiers', Atmospheric Research, 92, 258-269, doi.org/10.1016/j.atmosres.2008.12.002.

Lyu Y, Xu T, Yang X, Chen J, Cheng T, Li X, 2017, 'Seasonal contributions to sizeresolved n alkanes (C8–C40) in the Shanghai atmosphere from regional anthropogenic activities and terrestrial plant waxes', Science of the Total Environment, 579, 19181928, doi.org/10.1016/j.scitotenv.2016.11.201.

Maenhaut W. Francois F, Cafmeyer J, 1994. 'The 'Gent' Stacked Filter Unit Sampler for the Collection of Atmospheric Aerosols in Two Size Fractions: Description and Instructions for Installation and Use. In Applied Research on Air Pollution using Nuclear-Related Technique', IAEA Report NAHRES, 19, 249-263, Vienna.

Mandalakis M, Tsapakis, M, Tsoga A, Stephanou, E. G, 2002, 'Gas-particle concentrations and distribution of aliphatic hydrocarbons, PAHs, PCBs and PCDD/Fs in the atmosphere of Athens (Greece)', Atmospheric Environment, 36, 25,4023–4035, doi.org/10.1016/S1352-310(02)00362-X.

Mikuška P, Křůmal K, Večeřa Z, 2015, 'Characterization of organic compounds in the $PM_{2.5}$ aerosols in winter in an industrial urban area', Atmospheric Environment, 105,97-108, doi: 10.1016/j.atmosenv.2015.01.028.

Mudge S.M, Duce C.E, 2005, 'Identifying the source, transport path and sinks of sewage derived organic matter', Environmental Pollution, 136,2, 209-220, doi: 10.1016/j.envpol.2005.01.015.

Neusüss C, Pelzing M, Plewka A, Herrmann H, 2000, 'A new analytical approach for size-resolved speciation of organic compounds in atmospheric aerosol particles: Methods and first results', Journal of Geophysical Research, 105, 4513-4527, doi.org/10.1029/1999JD901038.

Obioh I.B, Ezeh G.C, Abiye O.E, Alfa A, Ojo E.O, Ganiyu A.K, 2013, 'Atmospheric particulate matter in Nigerian megacities', Environmental Toxicology and Chemistry, 95, 3, 379-385. doi.org/10.1080/02772248. 2013.790970.

Omar N.Y.M.J, Bin Abas M.R, Rahman N.A, Tahir N.M, Rushdi A, Simoneit B.R.T, 2007, Levels and distribution of organic source tracers in air and roadside dust particles of Kuala Lumpur, Malaysia, Environmental Earth Sciences, 52, 1485-1500, doi.org/10.1007/s00254-006-0593-6.

Oseni M.O, 2016, 'Get Rid of It: To What Extent Might Improve Reliability Reduce Self-Generation in Nigeria?', Energy Policy, 93, 246-54, https://doi.org/10.1016/j.enpol.2016.03.013.

Owoade O.K, Fawole O.G, Olise F.S, Ogundele L.T, Olaniyi H.B, Almeida M.S, Ho M-D, Hopke P.K, 2013, 'Characterization and source identification of airborne particulate loadings at receptor site classes of Lagos Mega-City, Nigeria, Journal of the Air and Waste Management Association, 63, 9,1026-1035, doi: 10.1080/10962247.2013.793627.

Ozoh O.B, Okwor T.J, Adetona O, Akinkugbe, A.O, Amadi C.E, Christopher Esezobor C, Adeyeye O.O, Ojo O, Nwude V.N, Mortimer K, 2018, 'Cooking Fuels in Lagos, Nigeria: Factors Associated with Household Choice of Kerosene or Liquefied Petroleum Gas (LPG)', International Journal of Environmental Research and Public Health, 15,4, 1-13, doi: 10.3390/ijerph15040641.

Ren, L, Fu, P, He, Y, Hou, J, Chen, J, Pavuluri, C.M, Sun, Y, Wang, Z, 2016, 'Molecular distributions and compound-specific stable carbon isotopic compositions of lipids in wintertime aerosols from Beijing', Scientific Report, 6, 27481, doi.org/10.1038/srep27481.

Rogge W.F, Hildemann L.M. Mazurek M.A. Cass G.R, 1993, 'Sources of fine organic aerosol, 2, Noncatalyst and catalyst-equipped automobiles and heavy-duty diesel trucks', Environmental Science and Technology, 27, 636-651.

Rushdi A. I, El-Mubarak A. H, Lijotra L, Al-Otaibi M. T, Qurban M.A, Al-Mutlaq K.F, Simoneit B. R. T, 2017, 'Characteristics of organic compounds in aerosol particulate matter from Dhahran city, Saudi Arabia' Arabian Journal of Chemistry, 10, S3532-S3547, doi.org/10.1016/j.arabjc.2014.03.001.

Russell A. G, Brunekreef B, 2009, 'A Focus on Particulate Matter and Health', Environmental Science and Technology, 43, 4620-4625, doi.org/10.1021/es9005459

Samara C, Voutsa D, Kouras A, Eleftheriadis K, Maggos T, Saraga D, Petrakakis M, 2014, 'Organic and elemental carbon associated to PM_{10} and $PM_{2.5}$ at urban sites of northern Greece', Environmental Science and Pollution Research, 21:1769-1785, doi 10.1007/s11356-013-2052-8.

Scalan R.S, Smith J.E, 1970, 'An improved measure of the odd-even predominance in the normal alkanes of sediment extracts and petroleum', Geochimica et Cosmochimica Acta, 34, 611-620, doi.org/10.1016/0016-7037(70)90019-0.

Schauer J.J, Kleeman M.J, Cass G.R, Simoneit B.R.T, 1999, 'Measurement of emissions from air pollution sources, 2, C_1 through C_{30} organic compounds from medium duty diesel trucks', Environmental Science and Technology, 33, 1578-1587, doi.org/10.1021/es980081n.

Simoneit B.R.T, Sheng G.Y, Chen X.J, Fu J.M, Zhang J, Xu Y.P, 1991, 'Molecular marker study of extractable organic-matter in aerosols from urban areas of China', Atmospheric Environment, Part A, 25, 2111-2129, doi.org/10.1016/0960-1686(91)90088-0.

Simoneit B.R.T, Kobayashi M, Mochida M, Kawamura K, Lee M, Lim H.J, Turpin B.J, Komazaki Y, 2004, 'Composition and major sources of organic compounds of aerosol particulate matter sampled during the ACE-Asia campaign', Journal Geophysical Research, 109, doi:10.1029/2004JD004598.

Simoneit B.R.T, Elias V.O, 2000, 'Organic tracers from biomass burning in atmospheric particulate matter over the ocean', Marine Chemistry, 69, 301-312, doi.org/10.1016/S0304-4203(00)00008-6.

Simoneit B.R.T, Elias V.O, 2001, 'Detecting organic tracers from biomass burning in the atmosphere', Marine Pollution Bulletin, 42, 805-810, doi.org/10.1016/S0025-326X(01)00094-7.

Simoneit B.R.T, Cardoso J.N, Robinson N, 1991, 'An assessment of terrestrial higher molecular weight lipid compounds in aerosol particulate matter over the South Atlantic from about 30–70° S', Chemosphere, 21, 1285-1301, doi.org/10.1016/0045-6535(91)90196-K.

Sojinu S.O, Sonibare O.O, Ekundayo O, Zeng E. Y, 2012, 'Assessing anthropogenic contamination in surface sediments of Niger Delta, Nigeria with fecal sterols and n-alkanes as indicators', Science of the Total Environment 441,89-96, doi.org/10.1016/j.scitotenv.2012.09.015.

Stein A.F, Draxler R.R, Rolph G. D, B. Stunder J. B, Cohen M. D, Ngan F, 2015, 'NOAA's HYSPLIT Atmospheric Transport and Dispersion Modeling System', Bulletin of the American Meteorological Society, 96, 2059-2077, doi.org/10.1175/BAMS-D-14-00110.1.

Sun N, Li X, Ji Y, Huang H, Ye Z, Zhao Z. 2021, 'Sources of $PM_{2.5}$-Associated PAHs and n-alkanes in Changzhou China', Atmosphere, 12, 9,1127, doi.org/10.3390/atmos12091127.

Tang X.L, Bi X.H, Sheng G.Y, Tan J.H, FU J.M, 2006, 'Seasonal variation of the particle size distribution of n-alkanes and polycyclic aromatic hydrocarbons (PAHs) in urban aerosol of Guangzhou, China', Environmental Monitoring Assessment, 117, 193-213, doi.org/10.1007/s10661-006-0440-y.

Tian S, Liu Y, Wang J, Wang J, Hou L, Lv B, Wang X, Zhao X, Yang W, Geng C, Han B, Bai Z, 2020, 'Chemical compositions and source analysis of $PM_{2.5}$ during autumn and winter in a heavily polluted city in China', Atmosphere, 11, 336, doi.org/10.3390/atmos11040336.

Turpin B.J, Lim H.J, 2001, 'Species Contributions to $PM_{2.5}$ Mass Concentrations: Revisiting Common Assumptions for Estimating Organic Mass', Aerosol Science and Technology, 35, 602-610, doi.org/10.1080/02786820119445, 2001.

Van Pinxteren M, Fiedler B, Van Pinxteren D, Iinuma Y, Körtzinger A, Herrmann H, 2015, 'Chemical characterization of sub-micrometer aerosol particles in the tropical Atlantic Ocean: marine and biomass burning influences', Journal of Atmospheric Chemistry, 72, 105–125, https://doi.org/10.1007/s10874-015-9307-3.

Wang G, Huang L, Zhao X, Niu H, Dai Z, 2006, 'Aliphatic and polycyclic aromatic hydrocarbons of atmospheric aerosols in five locations of Nanjing urban area, China', Atmospheric Research, 81, 54-66. doi.org/10.1016/j.atmosres.2015.09.007.

WHO (World Health Organization) 2021, 'WHO global air quality guidelines, Particulate matter ($PM_{2.5}$ and PM_{10}), ozone, nitrogen dioxide, sulfur dioxide and carbon monoxide', Geneva.

Yang J, Lei G, Liu C, Wu Y, Hu K, Zhu J, Bao J, Lin W, Jin J, 2023, 'Characteristics of particulate-bound n-alkanes indicating sources of $PM_{2.5}$ in Beijing, China', Atmospheric Chemistry and Physics, 23, 3015-3029, 2023, doi.org/10.5194/acp-23-3015-2023.

Yttri K.E, Simpson D, Bergström R, Kiss G, Szidat S, Ceburnis D, Eckhardt S, Hueglin C, Nøjgaard J.K, Perrino C, Pisso I, Prevot A.S.H, Putaud J.-P, Spindler G, Vana M, Zhang Y.-L, Aas W, 2019,

'The EMEP Intensive Measurement Period campaign, 2008–2009: characterizing carbonaceous aerosol at nine rural sites in Europe', Atmospheric Chemistry and Physics, 19, 4211-4233, https://doi.org/10.5194/acp-19-4211-2019.

Yu J.Z, Huang X. H. H, Ho S. S. H, Bian Q, 2011, 'Nonpolar organic compounds in fine particles: quantification by thermal desorption–GC/MS and evidence for their significant oxidation in ambient aerosols in Hong Kong', Analytical and Bioanalytical Chemistry, 401, 3125-3139, doi 10.1007/s00216-011-5458-5.

Zeng L, Offor F, Zhan L, Lyu X, Liang Z, Zhang L, Wang J, Cheng H, Guo H, 2019, 'Comparison of $PM_{2.5}$ pollution between an African city and an Asian metropolis', Science of the Total Environment , 696, 134069, doi.org/10.1016/j.scitotenv.2019.134069.

Zhang N, Cao J, Li L, Ho S.S.H, Wang Q, Zhu C, Wang L, 2018, 'Characteristics and Source Identification of Polycyclic Aromatic Hydrocarbons and n-Alkanes in $PM_{2.5}$ in Xiamen', Aerosol and Air Quality Research, 18,1673-1683, doi.10.4209/aaqr.2017.11.0493.

Technical article

Assessment of carbon dioxide emission factors from power generation in Burkina Faso

Bernard Nana[1,2*], Hamadi Zalle [1], Issoufou Ouarma[1,3], Tizane Daho[1], Arsène Yonli[1], Antoine Bere[1]

[1]Laboratoire de Physique et de Chimie de l'Environnement, Ecole doctorales Sciences et Technologies, Université Joseph Ki-Zerbo, Ouagadougou, Burkina Faso
[2]Ecole Normale Supérieure, Koudougou, Burkina Faso
[3]Centre universitaire de Banfora, Université Nazi Boni, Bobo-Dioulasso, Burkina Faso
*Corresponding author: nsbernard@yahoo.fr

Received: 28 February 2023 - **Reviewed:** 5 June 2023 - **Accepted:** 21 November 2023
https://doi.org/10.17159/caj/2023/33/2.15701

Abstract

Power generation is the second largest source of greenhouse gases (GHGs), particularly carbon dioxide (CO_2), in Burkina Faso's energy sector. When preparing the National Communications on Climate Change, Burkina Faso uses the default emission factors of the Intergovernmental Panel on Climate Change (IPCC) to estimate emissions from power generation. This study presents an assessment of CO_2 emission factors from power generation for 2018 in Burkina Faso and an assessment of the contribution of renewable energy to the reduction of CO_2 emissions. The national electricity company of Burkina Faso was chosen as the scope for this study. The estimation of emission factors for combustion is based on an analysis of fuels which are characterised in terms of molecular composition, density and water content. For CO_2, the emission factor is 76 903 kg/TJ for the combustion of Heavy Fuel Oil (HFO) and 73 525 kg/TJ for that of Distillate Diesel Oil (DDO). Using these emission factors, the CO_2 emissions attributable to power generation in 2018 were estimated at almost 580 Gg. The CO_2 emission factor for thermal power generation was estimated at 0.663 kg/kWh and that of the electricity generation mix at 0.569 kg/kWh. Finally, the use of 14.25% renewable energy in electricity generation avoided 16.7% of CO_2 emissions in 2018. The emission factor of electricity production decreases with the increase in the share of renewable energy in the energy mix.

Keywords

power generation, greenhouse gases, emission factor, carbon dioxide, thermal generation.

Introduction

Increasing anthropogenic emissions of greenhouse gases (GHGs), particularly carbon dioxide (CO_2), are the main cause of global warming (IPCC 2021). In 2019, the Intergovernmental Pannel for Climate Change (IPCC) estimated global net anthropogenic GHG emissions (75% of which was attributed to CO_2 emissions) at 59 $GtCO_2$-eq (IPCC 2022). GHG emissions from anthropogenic sources come from several sectors of activity, including the energy sector. In the energy sector, GHG emissions come mainly from the use of primary fossil fuels for electricity generation and transport. According to a report by the International Energy Agency (IEA), in 2018, energy-related CO_2 emissions reached about 33.1 Gt, an increase of 1.7% compared to 2017, the majority of which comes from electricity generation (IEA, 2019).

In the face of climate change, reducing CO_2 emissions is a major priority. An important step in the search for solutions to reduce emissions is to identify the most significant sources of GHG emissions. This is done through compiling an inventory of GHG emissions on the one hand, and on the other hand, the development of programmes to reduce anthropogenic GHG emissions through the reduced consumption of fossil fuels and to increase the use of low GHG-emitting technologies such as renewable energy sources.

The most commonly used method of calculating emissions is based on emission factors (EFs), verifiable activity data and global warming potentials (GWPs). The IPCC provides three tiers for estimating GHG emissions from stationary sources in the energy sector (IPCC 2006a): Tier 1 uses the fuel combustion from national energy statistics and default emission factors. Tier 2 uses the fuel combustion from national energy statistics, together with country-specific emission factors where possible, derived from national fuel characteristics. Tier 3 uses the fuel statistics and data on combustion technologies applied together with technology emission factors; this includes the use of models and facility-level emission data where available.

GHG emission factors depend on the characteristics of the fuels including carbon content and calorific value (IPCC 2006a). According to the IPCC, CO_2 emissions can be calculated at the more detailed level provided that complete statistics on fuel consumption and the typical carbon content of these fuels are available, as combustion conditions are relatively unimportant for the EF of CO_2 compared to that of CH_4 and N_2O (IPCC 2006a). The emission factor for electricity generation depends on the country's electricity generation fleet and the fuel mix used. For fossil fuel power plants, the efficiency of conversion of fuel to electricity and the load factor both influence the GHG emission rate during combustion (Spadaro et al. 2008).

Burkina Faso, a member of the United Nations Framework Convention on Climate Change (UNFCCC) and a non-Annex I country, is committed to reducing GHG emissions. Indeed, it has prepared National Communications under the UNFCCC. These Communications show that in 2015 the energy sector was the second largest emitter of anthropogenic GHGs after the Agriculture, Forestry and Other Land Use (AFOLU) sector (PRBA 2021). Burkina Faso's electricity production was the second largest contributor to GHG emissions in the energy sector after transport in 2015 (PRBA 2021, TCN 2022). GHG emissions from the energy sector in Burkina Faso have an increasing trend from 1995 to 2015. Indeed, GHG emissions from the energy sector have increased from 3.0% of Burkina Faso's total GHG emissions in 1995 to 6.1% in 2015. (PRBA 2021). CO_2 alone accounts for more than 99% of emissions from electricity production in Burkina Faso in 2015 (PRBA 2021).

So far, Burkina Faso assesses its GHG emissions from electricity generation using the IPCC default EFs (IPCC's Tier 1 approach) due to lack of country-specific (IPCC's Tier 2 approach) or technology-specific (IPCC's Tier 3 approach) EFs. However, a rigorous estimation of the sector's contribution requires more refined EFs. It is therefore important to develop country-specific EFs for the Burkinabe power generation sector. It is in this context that this work focuses on the evaluation of the country-specific CO_2 emission factors of electricity production in Burkina Faso in 2018 and the evaluation of the contribution of renewable energies to the reduction of carbon dioxide emissions from the Burkinabe electricity production sector.

Materials and methods

Study area

Burkina Faso, a West African country, is located between 9° 20' and 15° 05' North latitude and between 5°20' West longitude and 2°03' East longitude and has a surface area of 273 187 km^2. According to the preliminary results of the Fifth General Census of Population and Housing of Burkina Faso, its population is estimated at 20.5 million (INSD 2022). Burkina Faso is characterised by a tropical Sudanese climate. Burkina Faso's electricity network is composed of three subsystems: generation, transmission and distribution. Electricity generation is the subsystem that contributes most to anthropogenic GHG and air pollutant emissions.

Table 1: *SONABEL's electricity production by source in GWh in 2018 (ME, 2018a)*

Source of Production	Thermal	Hydroelec-tricity	Solar photo-voltaic	Total electricity generation
Production Electrical	8.7	56.5	27.0	11.1

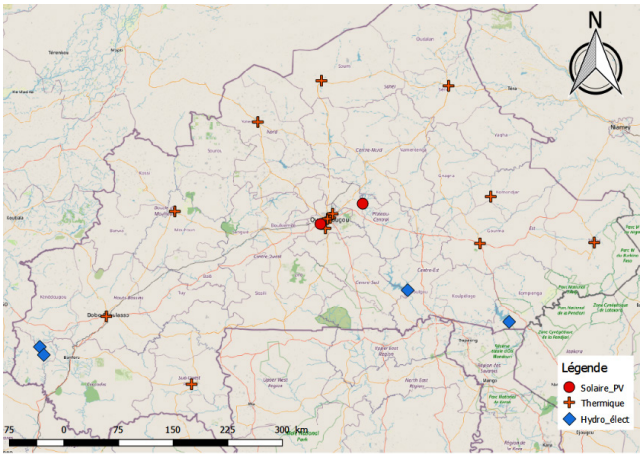


Figure 1: *Location of SONABEL's power plants in Burkina Faso in 2018*

Sources and data collection

Data was collected from the Burkina Faso Ministry of Energy (ME 2018a, 2018b). The data collected covers 2018 and provides: i) the annual electricity production of all power plants (thermal and renewable), and ii) the amount of fuel consumed by thermal power plants by fuel type. The national electricity company of Burkina Faso (called SONABEL) is the main supplier of electricity in Burkina Faso with 98% of the electrical energy delivered in 2018. Electricity production is made up of thermal, hydro and solar power (Table 1). SONABEL's power production for 2018 was composed of 85.75% thermal energy; 8.95% hydroenergy and 5.3% solar photovoltaic energy (SONABEL 2018). There were nineteen (19) functional power plants in 2018 (ME 2018b) (Figure 1).

Heavy Fuel Oil (HFO) and Distillate Diesel Oil (DDO) are the petroleum products used to run SONABEL's thermal power plant fleet. In 2018, thermal power plants consumed 186 805 t of fuel (SONABEL 2018), 86.5% of this being HFO (161 586 t) and 13.5% DDO (25 219 t). SONABHY is the only distributor of hydrocarbons in Burkina Faso.

Methodology

In this study, only the emissions related to the electricity production phase in 2018 are taken into account. The county-specific characteristics of the fuel takes into consideration the molecular structure and density of the fuel and the water content of the HFO.

Molecular constituents by characterisation and elemental composition by mass of fuels

A sample of the two fuels (see Figure 2) consumed by the thermal power plants was taken from SONABEL's Ouaga II

thermal power plant and submitted for analysis in order to determine the molecular constituents of Burkinabe DDO and HFO. To prevent diffusion of hydrocarbon components, we used black plastic film to wrap the bottles and stored them in a cool, dark place at 20°C.

Figure 2: *Photograph of HFO and DDO fuel samples*

The samples analysis was performed by a Thermo Finningan Trace 2000 Gas Chromatography Mass Spectrometry (GC-MS) system. A DB-5 MS column from Agilent (30 m, ID 0.25 mm, df 0.25 µm) was used in the chromatograph. An initial oven temperature of 40°C was maintained for 5 minutes, then the temperature was increased at a rate of 10°C/min to a final temperature of 220°C for 15 minutes. Sample were injected (split mode) in an injector heated to 180°C. The MS transfer line was heated at 220°C and the ion source was operated at 200°C, electron energy 70 eV. American Chemical Society (ACS) grade reagents were used for hydrocarbon compounds calibration (external calibration). Total ion chromatogram (TIC) was used to identify compounds in the samples then SIM mode was used for compound quantification. Data were processed by Thermo Xcalibur 3.1 software.

If $C_xH_yO_zN_uS_t$ is the chemical formula of the fuel, then the values of x, y, z, u and t are determined as the average value of the C, H, O, N and S atoms of its constituents respectively.

$$x = \sum_{i=1}^{n} p_i x_i \; ; \quad y = \sum_{i=1}^{n} p_i y_i \; ; \quad z = \sum_{i=1}^{n} p_i z_i \; ; \quad u = \sum_{i=1}^{n} p_i u_i \; ; \quad t = \sum_{i=1}^{n} p_i t_i \quad (1)$$

where p_i is the mass fraction of component i contained in the fuel and x_i, y_i, z_i, u_i and t_i represent the number of carbon (C), hydrogen (H), oxygen (O), nitrogen (N) and sulphur (S) atoms in component i respectively.

If $M_{C_xH_yO_zN_uS_t}$ is the molar mass of the fuel, then:

$$M_{C_xH_yO_zN_uS_t} = \sum_{i=1}^{n} p_i M_i \quad (2)$$

M_i is the molar mass (in g/mol) of compound i present in the fuel, i.e., M_C=12; M_H=1; M_O=16; M_N=14; M_S=32

Experimental measurement of fuel density

The density of each fuel is determined experimentally by the density meter method (Figure 3). This method is based on NF T60-101 (Guillermic 1980). Given the high variability in HFO characteristics, three samples from different origins (Cotonou, Lomé and Abidjan) were taken and submitted for analysis. For DDO, only one sample from Cotonou was submitted for analysis. The equipment used consists of a 500 mL capacity graduated cylinder; a 0.950 to 1 graduated density meter for HFO and 0.800 to 0.900 for DDO; a HANNA Checktemp digital thermometer and a Stem. A volume of the fuel was taken from the 500 mL test tube and then the thermometer and density meter were inserted. After stabilisation of the density meter (about 15 to 20 minutes and without contact with the walls of the test piece), the temperature (T) of the fuel and the measured density (d_4^T) were read.

Figure 3: *Measuring the density of each fuel*

The following equation was used to estimate the density of HFO and DDO at 15°C (Guillermic 1980):

$$d_4^{15} = d_4^T + A\,(T - 15) \quad (3)$$

where:
d_4^T is the measured density of the fuel at the temperature T;
d_4^{15} is the density at 15°C; and,
A is the density correction factor.

The table given by Guillermic (1980) for the choice of the coefficient A was used. The density of HFO at 15°C was taken as the average of all three samples.

Determination of water content

The determination of the water content was based on the NFT 60-113 standard (Guillermic 1980) (Figure 4). The water content of HFO was determined by taking the ratio of the volume of water contained in the fuel to the volume of the fuel considered. A mixture of 100 mL of xylene (non-disposable solvent) and 100 mL of HFO taken with the test tube was first poured into the

flask. Everything was stirred to obtain a homogeneous mixture. Then, we heated the mixture. The water in the sample was obtained by reflux distillation of the water-miscible solvent. After condensation, water separates from the solvent (xylene) and accumulates in the graduated recipe tube. Finally, the volume of water contained in the tube was read.

Figure 4: *The water content test protocol*

Determining the higher and lower calorific values of fuels

The higher calorific value (HCV) or gross heat (Q) of combustion of fuels consumed by thermal power stations (in cal/g) is estimated using equation (4) given by Speight (2006) which gives a fairly high accuracy of HCV with a deviation generally less than 1%.

$$HCV = Q = 12\,400 - 2100\,d^2 \quad (4)$$

Where: d is the 60/60°F specific gravity or density at 15.6°C.

The HCV in kJ/kg is obtained by multiplying the above expression by 4.185 kJ.

In this study, the density at 15°C of the fuels is used to estimate the HCV.

The lower calorific value (LCV) of fuels (in kJ/kg) is estimated using equation (5) (Sawerysyn 1993):

$$LCV = HCV - m_{H_2O}L_v \quad (5)$$

where:
LCV: Lower calorific value of fuel consumed by thermal power plants in power plants (kJ/kg)
HCV: Higher calorific value of the fuel (kJ/kg)
m_{H_2O}:Total mass of water in kg released by the combustion of 1 kg of raw fuel
L_v: the latent heat of vaporisation of water.

The total mass of water (m_{H_2O}) is estimated using equation (6) (Sawerysyn 1993).

$$m_{H_2O} = \frac{\%E}{100} + \frac{9\,\%H}{100} \quad (6)$$

where:
%E is the overall moisture content of the raw fuel; and,
%H is the hydrogen content of the fuel.

These different percentages are related to the unit mass of the raw fuel. The following information was considered: 1 kcal = 4.185 kJ and L_v = 2.51 MJ/kg at 0°C (Ibrahim et al. 2006).

Method of quantifying CO_2 emissions

The following assumptions were applied:

- The oxidation factor (OF) of carbon was assumed to be 100%;
- The results of the HFO and DDO analyses were valid for all fuels in the different thermal power plants.
- The CO_2 emission factors evaluated in this study are used to quantify CO_2 emissions for 2018.

The CO_2 emissions from combustion were estimated using the following equation (CITEPA 2020):

$$E_{CO_2} = \sum_{f=1}^{n} Q_f \times LCV_f \times EF_{CO_2,f} \quad (7)$$

where:
ECO_2 is the total CO_2 emission from fuel combustion (in kg);
Q_f is the quantity of fuel f (in kg); and,
$EFCO_{2,f}$ is the CO_2 energy emission factor for the combustion of fuel f (in kg CO_2/TJ or kgCO_2/kWh).

The CO_2 energy emission factor for the combustion of fuel (f) was calculated using the following formula (Ramphull & Surroop 2017):

$$EF_{CO_2,f} = FE_{CO_2,f} = C_E \times OF \times 3{,}667 \quad (8)$$

$$C_E = \frac{\%C}{LCV_f} \quad (9)$$

where:
%C is the carbon content in kg carbon/kg fuel;
OF is the oxidation factor;
3.667 is the ratio of the molar mass of CO_2 to that of carbon or the conversion factor of C to CO_2; and,
C_E is the carbon content per energy of the fuel in kg/kJ or kg/GJ.

Assessment of the CO_2 emission factor of electricity generation

The CO_2 emission factor of the electricity production of thermal power plants was estimated as the ratio of the CO_2 emitted in 2018 to the thermal production in the same period.

$$EF_{th} = \frac{E_{CO_2}}{E_{el,th}} \quad (10)$$

Where:
EF_{th} is the CO_2 emission factor of thermal generation in kg CO_2/kWh; and,
$E_{el,th}$ is the electricity production of thermal power plants.

The EF of the electricity generation mix was calculated by averaging the EFs of the different electricity generation sources.

$$EF_{el} = (EF_{th} \cdot \chi_{th}) + (EF_{hy} \cdot \chi_{hy}) + (EF_{PV} \cdot \chi_{PV}) \quad (11)$$

where:
EF_{el} is the emission factor of electricity generation in the electricity mix (in kg CO_2/kWh) in 2018;
EF_{th}, EF_{hy} and EF_{PV} are the CO_2 emission factors for thermal, hydro and solar PV electricity generation (in kg CO_2/kWh) respectively;
EF_{th}, EF_{hy} and EF_{PV} represent the ratio of thermal, hydoand solar PV generation in the electricity generation mix (in kWh) respectively for the period 2018.

Hydroelectricity and solar photovoltaic do not emit CO_2 during the electricity generation phase. Equation 12 can thus be simplified as folows:

$$EF_{el} = EF_{th} \cdot \chi_{th} \quad (12)$$

The EF of CO_2 from electricity generation can be estimated using the following formula:

$$EF_{él} = \frac{E_{CO_2}}{E_{el}} \quad (13)$$

Where:
E_{el} being is the total electricity production from all sources in kWh for the 2018 period.

Results and discussion

Constituents and mass composition of fuels

1,2-dimethylcyclohexane (C_8H_{18}) is the major molecular component present in HFO (72%) followed by tetradecyl cyclohexane ($C_{20}H_{40}$) (17%) (Figure 5). HFO contains mainly cycloalkanes (or naphthenes) (89%) and small amounts of organic nitrogen compounds and sulfur. No oxides are present in HFO. The indeterminates in HFO are very high and this can be explained by several factors including the source of the fuel, as there are several varieties of heavy fuel oils. For DDO, the main molecular component is decane ($C_{10}H_{22}$) (71%) followed by tetradecyl cyclohexane ($C_{20}H_{40}$) (15%) (Figure 6). DDO contains 83.5% non-cyclic alkanes or paraffins and 15% cycloalkanes or naphthenes. It is also noticeable that sulfides, nitrates and oxides are not present in DDO. The indeterminates in DDO are less significant than in HFO. Undetermined molecules (9%) in HFO are linear and branched alkanes with low percentages for each type of compounds. Thus n-alkane compounds are n-C10 (0.8%), n-C11 (0.7%), n-C13 (1.3%), n-C14 (2.2%), n-C15 (1.8%). The branched alkanes are C14 (1.2%), C15(0.5%) and C16(0.5%).

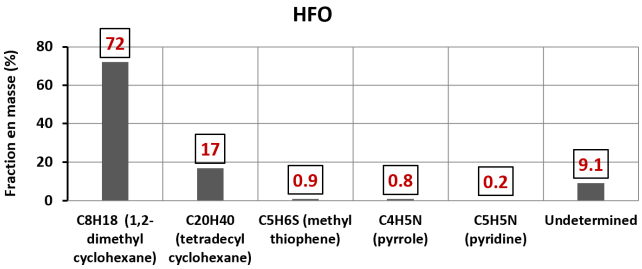


Figure 5: *Molecular composition of HFO used for thermal power generation in Burkina Faso*

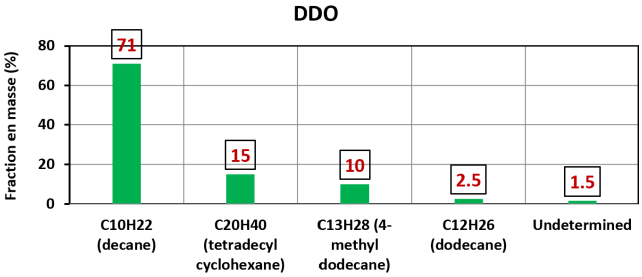


Figure 6: *Molecular composition of DDO used for thermal power generation in Burkina Faso*

Previous studies have shown that petroleum hydrocarbons mainly contain paraffins, naphthenes, olefins and aromatic hydrocarbons (Gruyer et al. 2015). Crude oil and refinery products contain other elements such as sulfides (S), nitrates (N), oxides (O) and some metals including nickel and vanadium in low or trace concentrations especially the heavier fractions (Guibet 1997). Moreover, in HFO (heavy fraction compared to DDO), N compounds are present but in smaller quantities to the S compounds. This was correlated in the sampled SONABEL HFO and DDO results (Table 2).

Table 2: *Elemental composition of HFO and DDO by mass and chemical formula of fuels*

Fuel	Carbon (C)	Hydrogen (H)	Sulfur (S)	Nitrogen (N)	Chemical formula
HFO	84.54	15.13	0.22	0.11	$C_{9.247}H_{19.864}N_{0.01}S_{0.009}$
DDO	84.85	15.15	0	0	$C_{11.7}H_{25.07}$

The HFO consumed by thermal power plants contains a low sulfur content. The estimated C, S and N content of HFO and the C content of DDO are in line with the data provided by Speight (2006) and within the limits given by Guillermic (1980). The only exception is hydrogen for both fuels with a mass fraction above 14%. Indeed, in the literature, the mass fraction of hydrogen in petroleum fuels ranges from 10 to 14%

(Guillermic 1980, Speight 2006). This observed variance may be due to two reasons: i) differences in the origin of the oil and the refining conditions, and ii) difference in the fuel characterisation methodology. The results of the work of Kornelius et al. (2022) on the analysis of liquid and gaseous fuels used in South Africa gave a carbon content of 85.93±1.58% for Heavy Fuel Oil (HFO), and 88.31±0.79% in the summer period and 87.00±0.38% in the winter period for diesel. We found a difference of less than 2% between the carbon content value estimated in this study for Burkinabe HFO and South African HFO. The carbon content of sampled South African DDO was 3.92% and 2.5%, lower than that of diesel (in summer and winter respectively) as estimated by Kornelius et al. (2022).

Density and water content of fuels

The density of HFO at 15°C is the average of the densities of the three HFO samples. It is evaluated at 0.964 kg/L (Table 3).

Table 3: *Density of HFO and DDO at measured temperature and adjusted to 15°C*

Fuel	HFO			DDO
Provenance	Cotonou	Lomé	Abidjan	Cotonou
Temperature measured (°C)	28	29	29	29
Density d_4^T (kg/L)	0.977	0.963	0.927	0.827
Correction coefficient A	$0.6x10^{-3}$	$0.6x10^{-3}$	$0.62x10^{-3}$	$0.68x10^{-3}$
Density d_4^{15} (kg/L)	0.985	0.971	0.935	0.837

According to Gruyer et al. (2015), most hydrocarbons have densities between 0.7 and 0.99 kg/L at 15°C, and the calculated density for HFO and DDO are found to be in this range. In their work, Kornelius et al. (2022) found a density of 0.994±0.12 kg/L for HFO and 0.826±0.002 kg/L and 0.825±0.002 kg/L for diesel used in South Africa in the summer and winter periods, respectively. The density of DDO is 1.3% higher than that of diesel evaluated by Kornelius et al. (2022). HFO from Cotonou and Lomé are comparable to that from South Africa. The estimated density for HFO is in line with data from Wauquier (1994) and IPCC (2006b). Indeed, the density of HFOs is higher than 0.920 kg/L at 15°C (Wauquier 1994) . For the IPCC, the density of heavy residual fuels, including those obtained by blending, always exceeds 0.90 kg/L.

In 100 mL of HFO, the volume of water measured is 0.05 mL, i.e. a water content of 0.05%. This result shows that the HFO consumed by SONABEL's thermal power stations has a very low water content. For DDO, the water content is considered to be zero because it is a cleaner fuel than HFO and even closer to diesel oil and its visual appearance shows that it contains no traces of water.

Higher and lower calorific values of fuels

The total mass of water released by burning 1 kg of HFO is 1.3622 kg and for DDO it is 1.3635 kg. The HCV/LCV ratio is approximately 1.084 for HFO and 1.081 for DDO (Table 4). In this study, the difference between HCV and LCV relative to HCV is about 7.5% for DDO and 7.8% for HFO. In general, calculations in the literature for different fuel categories show that the relative difference between HCV and LCV relative to HCV is in the order of 2-4% for coals, 5-8% for fuel oils and 0-15% for gaseous fuels (Sawerysyn 1993), and for the IPCC the LCV is about 5% lower than the HCV for coal and oil and 10% lower than the HCV for most forms of natural gas (IPCC 2006b). The estimated difference between HCV and LCV in this study is between 5 and 8%, a range given for fuel oils (Sawerysyn 1993).The LCV of HFO is slightly lower than the IPCC LCV for residual fuel oil. As shown in Table 4, the LCV of HFO is in the range of the LCV of residual fuel oils provided by the IPCC (39.8 and 41.7 TJ/Gg). Ramphull & Surroop (2017) estimated an LCV value of 40.6 MJ/kg for a viscosity of 380 cSt HFO and 41.1 MJ/kg for a viscosity of 180 cSt HFO consumed in Mauritius. For HFO with a viscosity of 180 cSt, the difference is less than 2%. The LCV of DDO (42.31 MJ/kg) is very close to that estimated for Mauritian diesel (42.3 MJ/kg) by Ramphull and Surroop (2017).The LCV of DDO is slightly lower than the IPCC LCV for diesel (with the estimated difference being 1.63%). The LCV of DDO is within the IPCC range of diesel LCV (41.4 - 43.3 MJ/kg). It is very close to that estimated by Ramphull & Surroop (2017) for diesel (42.3 MJ/kg). The calorific values evaluated for DDO are respectively close to those evaluated for diesel by Kornelius et al (2022). Indeed, the latter found the HCV to be 45.93±0.09 MJ/kg (45.85±0.043 MJ/kg) and the LCV to be 43.05±0.07 MJ/kg (42.99±0.033 MJ/kg) for diesel used in South Africa during the summer (winter) period. According to CITEPA (2022), the PCI of commercial heavy fuel oil in France is 40 MJ/kg and 42.6 MJ/kg for domestic fuel oil/gasoil.

Table 4: *Estimated HCV and LCV for Burkinabe fuels*

Fuels	HFO	DDO	Residual fuel oils (default LCV) (IPCC 2006b)	Gasoline/ Diesel (default LCV) (IPCC 2006b)
HCV (kJ/kg)	**43 726.88**	**45 737.05**	-	-
HCV (MJ/kg)	**43.73**	**45.74**	-	-
HCV (kWh/kg)	**12.15**	**12.70**	-	-
LCV (kJ/kg)	**40 307.76**	**42 314.67**	40 400	43 000
LCV (TJ/Gg ou MJ/kg)	**40.31**	**42.31**	40.4	43.0
LCV (kWh/kg)	**11.20**	**11.75**	-	-
Lower and upper limit of LCV (TJ/Gg) for 95% confidence interval	-	-	39.8 – 41.7	41.4 – 43.3

Carbon content by energy and CO_2 emission factors for fuel combustion

Figure 7 presents the results of the carbon content per energy and energy emission factors for the combustion of HFO and DDO on a net calorific basis and the IPCC default. The carbon content per energy and energy emission factors for combustion on a net calorific value basis and the IPCC default values are

also provided. Figure 7 shows the CO_2 EFs assessed for fuel combustion and the IPCC default EFs for residual fuel oils and diesel/diesel on a net calorific basis (IPCC 2006a).

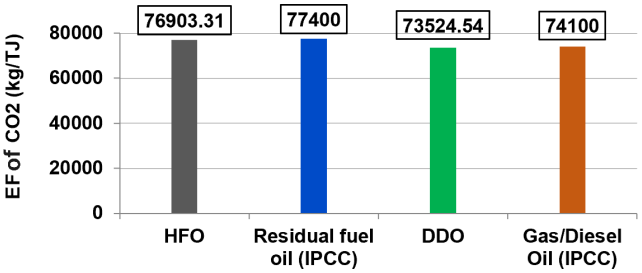


Figure 7: *Energy-related CO_2 emission factors for HFO and DDO combustion and IPCC default factors for residual fuel oils and diesel/ diesel in kg/TJ*

The CO_2 EFs estimated for HFO and DDO combustion are slightly lower than the IPCC default EFs for residual fuel oils and gasoline/diesel respectively. The calculated EFs for HFO and DDO are within the IPCC 95% confidence interval for HFO (75 500 – 78 800 kg/TJ) and DDO (72 600 – 74 800 kg/TJ) respectively. According to the IPCC, when a county-specific EF falls within the 95% confidence interval, the emission factors can be considered consistent with the IPCC default values (IPCC 2006b). The calculated country-specific emission factors are therefore in agreement with the IPCC data.

Contribution of the Burkinabe electricity sector to CO_2 emissions

Total calculated CO_2 emissions attributable to fuel combustion within Bukina Faso for the period 2018, using the developed counrty-specific emission factors, amounted to 580 Gg, i.e. 501 Gg from the combustion of HFO and 79 Gg from the combustion of DDO. This represents 86.5% for HFO and 13.5% for DDO. Using the IPCC default emission factor, SONABEL has estimated its CO_2 emissions for 2018 at 584 Gg (SONABEL 2018). The combustion of HFO emits more CO_2, for the same energy production, than that of natural gas or lighter petroleum products (petrol, diesel, domestic fuel oil) but less than that of coal or petroleum coke (Soleille 2004). The Burkinabe power generation sector is a sector that contributes significantly to CO_2 emissions through the use of fossil fuels, particularly petroleum products.

CO_2 emission factor for electricity generation

The emission factor of SONABEL's thermal electricity production is quite high, as 1 kWh of electricity produced by thermal power plants results in the release of 0.663 kg of CO_2 (Table 5). According to ADEME (2016), the emission factor of electricity production for the oil-fired power plant is 0.704 kg CO_2-e per kWh. Afilal et al. (2019) state that the electricity generated from HFO has an emission factor of 0.73 kg CO_2-eq per kWh for fuel oil. It is noted that the value estimated in this study is lower than the value provided by ADEME (2016) and Afilal et al. (2019) 5.8% and 9.2% for DDO and HFO respectively. The discrepancies are due to the fact that the EF of SONABEL's thermal production is for CO_2 only and does not include other GHGs. It also does not take into account the emissions linked to the upstream fuels.

Table 5: *CO_2 emission factor for thermal electricity generation and SONABEL's electricity generation*

SONABEL's source of electricity generation	Thermal	Generation mix (thermal and renewable)
CO_2 emission factor (in $kgCO_2$/kWh LCV)	0.663	0.569

The CO_2 emission factor of SONABEL's electricity production is quite high and highlights the largely carbon-based nature of the electricity produced in Burkina Faso by SONABEL. According to a feasibility study for the development of a regional electricity grid EF for the West Africa Power Exchange System (WAPES) in 2014, the EF of the regional electricity grid is estimated at 0.588 $kgCO_2$/kWh and is also referenced to Burkina Faso in terms of SONABEL's interconnected grid (MEEVCC 2018). This EF is 3% higher than the one estimated in our study. This discrepancy is due to the fact that the value estimated in this work does not take into account emissions due to electricity exchanges at interconnections (electricity imports). In the West African sub-region, the electricity generated EF for Côte d'Ivoire, Niger and Benin are 0,402; 0.630 and 0.684 $kgCO_2$/kWh respectively (Ritchie et al. 2022). In 2018, the global average carbon intensity of electricity generated was 0.475 $kgCO_2$/kWh (CITEPA 2019).

Assessment of the contribution of renewable energies to the reduction of CO_2 emissions in the electricity mix

Estimated CO_2 emissions avoided in 2018

In 2018, the share of renewable energy (RE) in SONABEL's electricity production was 145.5 GWh (SONABEL 2018), 14% of total production. The avoided CO_2 emissions ($E_{CO_{2av}}$) are evaluated by considering that the electricity production from RE sources (E_{ER}) is instead generated by thermal power plants and by assuming the reference CO_2 EF (FE_{th}) of 0.663 kg /kWh, estimated in this study.

$$E_{CO_{2av}} = E_{ER} \times FE_{th} \quad (14)$$

It is estimated that nearly 97 Mt of CO_2 were avoided, 16.7% of the total CO_2 emissions attributable to SONABEL's electricity production in 2018. RE in electricity production reduced the CO_2 emission factor by 14.2. The analysis of the results shows that the reduction of the concentration of anthropogenic GHGs in the atmosphere, in particular CO_2 in the electricity generation sector, can be achieved by increasing the share of RE in the Burkinabe electricity mix.

Evolution of the CO_2 emission factor of electricity generation according to the share of renewable energy

Natural resources that can be exploited for electricity production in Burkina Faso include sun (Bambara 2015), water (ECREE 2012),

biomass (Ministère des Mines et de l'Energie 2015) and wind (Landry, et al. 2011). In a context marked by the scarcity of fossil energy resources and the need to combat the harmful effects of climate change, the development of alternative energy sources is essential. In this context, one of the strategic objectives set by the Burkina Faso Ministry of Energy is to make electrical energy available and accessible to all through the development and mass use of RE in the energy supply.

In 2018, with 14.25% RE sources in the generation mix, the CO_2 emission factor of SONABEL's electricity generation was estimated at 0.569 $kgCO_2/kWh$ LCV. The emission factor ($FE_{él}$) of CO_2 for SONABEL's electricity mix, as a function of the share of RE, was obtained using equation (13) with $\chi_{th} = 1 - \chi_{ER}$. Thus:

$$FE_{el} = FE_{th} . (1 - \chi_{ER}) \quad (15)$$

Where:
χ_{ER} is the share of RE in SONABEL's electricity generation mix

A projection of the CO_2 EF of SONABEL's electricity mix according to the share of RE is presented in Figure 8.

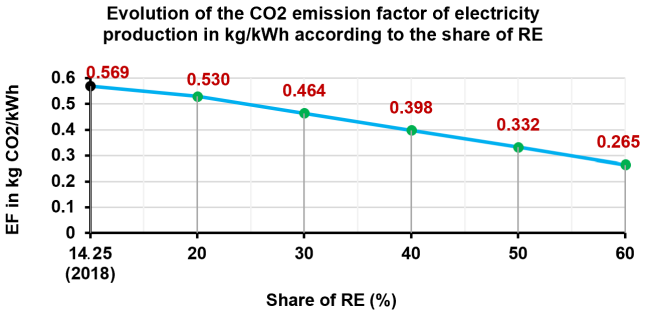


Figure 8: *Evolution of the CO_2 emission factor of electricity production as a function of the share of renewable energy*

The presence of RE (14.25%) in the electricity generation mix in 2018 enabled the CO_2 EF of the electricity generation mix to fall from 0.663 $kgCO_2/kWh$ to 0.569 $kgCO_2/kWh$, a reduction of 14.2% in 2018. As the share of renewables in the generation mix increases, the EF gradually decreases. With a RE penetration rate of 57%, the CO_2 content per kWh of the electricity generation mix will be half that of 2018. Thus, an increase of 43% in the share of renewable energies in the generation mix will result in a 50% reduction in the EF of the electricity generation mix in 2018. Burkina Faso can reduce the GHG content of its electricity if it invests more in renewable. The model proposed in this study (Figure 8) is an indicator that will enable decision-makers and public authorities to take measures to decarbonise Burkina Faso's electricity sector. Burkina Faso will be able to reduce the GHG content of its electricity on the one hand, and meet the demand for electricity on the other, by investing more in RE. Solar energy is the most abundant RE source in Burkina Faso with an average sunshine rate of 5.5 kWh/m2/day and a sunshine duration of 3000 to 3500 hours/year (MEEVCC 2018). It is the preferred RE source in the Burkinabe electricity mix for low carbon electricity.

Conclusion

The CO_2 emission factors of electricity generation in Burkina Faso and the contribution of RE to reduce CO_2 emissions from the Burkinabe power sector have been evaluated in this paper. It was found that the estimated CO_2 energy EF for HFO and DDO combustion are slightly lower than the IPCC default emission factors, but are within the 95% confidence interval given for the default EFs for residual fuel oils and diesel/gasoline respectively. The estimation of country-specific EFs took into account the characteristics of each fuel, notably the carbon content and the lower heating value. These emission factors are country-specific and are best suited to the national context for quantifying CO_2 emissions from power generation using the IPCC Tier 2 method, particularly for national GHG emission inventories under the UNFCCC. Burkina Faso's power generation sector is a major contributor to CO_2 emissions in the energy sector. Increasing the share of renewable energies in the electricity production mix will, on the one hand, will solve the energy deficit and, on the other hand, significantly reduce the sector's CO_2 emissions. As direct measurement of GHG emissions is a complex exercise, a tool for quantifying GHG emissions from the energy sector by means of emission factors could be developed in the short to medium term.

Acknowledgements

This work was conducted within the Laboratory of Environmental Physics and Chemistry (LPCE). The objectives were achieved thanks to the support and collaboration of several people. We would like to reiterate our sincere thanks to the managers of SONABEL.

Note

An earlier version of this paper was presented at the International Conference on Air Quality in Africa (ICAQ' Africa 2022) in October 2022 and was published in its Proceedings.

References

ADEME, 2016. D*ocumentation des facteurs d'émissions de la Base Carbone.*

Afilal, M. E., Abbadin, Laiche, H. & Elfarh, A., 2019. Le mix énergétique et les GES au Maroc: Évaluation des Scenarios et mesures prévues. *Revue Espace Géographique et Société Marocaine*, Volume n°26, pp. 273-290.

Bambara, B., 2015. *Atlas des énergies renouvelables du Burkina Faso, Mémoire de Master.*

BP Statistical Review of World Energy, 2022. *BP Statistical Review of World Energy*. Available at: https://www.bp.com/content/dam/bp/business-sites/en/global/corporate/pdfs/energy-economics/statistical-review/bp-stats-review-2022-full-report.pdf [Access on February 23, 2023].

CITEPA, 2019. *Emissions mondiales de CO_2: 2e année de hausse après trois ans de stabilisation*. https://www.citepa.org/fr/2019_04_a1

CITEPA, 2020. *Organisation et méthodes des inventaires nationaux des émissions atmosphériques en France*, OMINEA - 17ème édition

CITEPA, 2022. *Facteurs d'émission CO_2 et pouvoirs calorifiques inférieurs (PCI) nationaux (valeurs par défaut) par type de combustible, France.*

ECREE, 2012. A*telier Régional de la CEDEAO sur la petite hydro-électrique : Rapport de base sur des systèmes hydro-électriques existants et potentiels des pays de la CEDEA*O.

Gruyer, N., Groleau P. E, Ouellet A., Dupont F., 2015. *Hydrocarbures pétroliers : caractéristiques, devenir et criminalistique environnementale* - Études GENV22 et GENV23: Centre d'Expertise en Analyse Environnementale du Québec.

Guibet, J.-C., 1997. *Carburants et moteurs-Technologies-Energie-Environnement, Tome 1*. : Publication de l'Institut Français du Pétrole, TECHNIP.

Guillermic, A., 1980. *Le chauffage par les combustibles liquides, Caractéristiques des combustibles*, Tome 1. TECHNIP éd. s.l.:Institut Français du pétrole.

IEA (International Energy Agency), 2019. *Global Energy and CO_2 Status Report 2018.*

INSD (Institut National de la statistique et de la démographie), 2022. *Cinquième Recensement Général de la Population et de l'Habitation du Burkina Faso-Synthèse des résultats définitifs*, Burkina Faso

IPCC, 2006a. *IPCC Guidelines for National Greenhouse Gas Inventories, Volume 2: Energy, Chapter 2: Stationary Combustion.*

IPCC, 2006b. *IPCC Guidelines for National Greenhouse Gas Inventories, Volume 2: Energy, Chapter 1: Introduction.*

IPCC, 2021. *Climate Change 2021, The Physical Science Basis, Working Group I Contribution to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change.*

IPCC, 2022. *Climate Change 2022, Mitigation of Climate Change, Working Group III contribution to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change.*

Kornelius, G., Forbes, P., Fischer, T. & Govender, M., 2022. Tier 2 greenhouse gas emission factors for South African liquid and gaseous fuels. *Clean Air Journa*l, 32(2). https://doi.org/10.17159/caj/2022/32/2.15226

Landry, M., Ouedraogo, Y., Gagnon, Y. & Ouedraogo, A., 2011. *Atlas éolien du Burkina Faso*. https://doi.org/10.7135/UPO9780857288134.009

Laxalde, J., 2012. *Analyse des produits lourds du pétrole par spectroscopie vibrationnelle. Chimie théorique et/ou physique*. Thèse de Doctorat. : Université des Sciences et Technologie de Lille.

ME (Ministère de l'Energie du Burkina Faso), 2018a. *Annuaire Statistique, Burkina Faso.*

ME (Ministère de l'Energie du Burkina Faso), 2018b. *Tableau de bord, Burkina Faso*.

MEEVCC (Ministère de l'Environnement, de l'Economie Verte et du Changement Climatique), 2018. *Evaluation des Besoins technologiques pour l'atténuation - Energie et Transport* - Version finale, Burkina Faso

Ministère des Mines et de l'Energie, 2015. *Plan d'Actions National des Energies Renouvelables du Burkina Faso Période [2015-2020/2030] dans le cadre de la mise en œuvre de la Politique d'Energies Renouvelables de la CEDEAO (PERC).*

PRBA, 2021. *Premier rapport biennal actualisé du Burkina Faso sous la CCNUCC.*

Ramphull, M. & Surroop, D., 2017. Greenhouse Gas emission factor for the energy sector in Mauritius. *Journal of Environmental Chemical Engineering*, 5(6), p. 5994–6000. https://doi.org/10.1016/j.jece.2017.11.027

Ritchie, H., Roser, M., & Rosado, P. (2022). *"Energy". Published online at OurWorldInData.org. Retrieved from: 'https://ourworldindata.org/energy'* [Online Resource] [Access on February 23, 2023].

Sawerysyn, J.-P., 1993. Les pouvoirs calorifiques, *Bulletin de l'Union des Physiciens*. Volume 87, pp. 401-411.

Soleille, S., 2004. *L'industrie du raffinage et le devenir des fiouls lourds*, Rapport final.

SONABEL (Société Nationale Burkinabé d'Electricité), 2018. Rapport d'activité.

Spadaro, J. V., Langlois, L. & Hamilton, B., 2008. Emissions de gaz à effet de serre provenant des chaînes de production d'électricité, Evaluer la différence, AIEA BULLETIN 42/2/2000. pp. 19-24.

Speight, G. J., 2006. *The Chemistry and Technology of Petroleu*m. Fourth Edition. https://doi.org/10.1201/9781420008388

TCN, 2022. *Troisième Communication Nationale sous la Convention-Cadre des Nations Unies sur les Changements Climatiques (CCNUCC)*, Burkina Faso.

Wauquier, J. P., 1994. *Le raffinage du pétrole. Pétrole brut, Produits pétroliers, Schémas de fabrication, Caractéristiques des produits pétroliers utilisés à des fins énergétiques (carburants*

- combustibles) - Tome 1: Publication de l'Institut Français du Pétrole, TECHNIP.

WMO, 2020. *World Meteorological Organization (WMO) greenhouse gas bulletin N°16, The State of Greenhouse Gases in the Atmosphere Base.*

uMoya-NILU Consulting (Pty) Ltd is an air quality management consultancy with a range of service offerings. The company was established in 2007 and is a Level-2 B-BBEE contributor. uMoya-NILU has offices in Durban and Johannesburg.

SERVICE OFFERINGS

- **Atmospheric Impact Reports (AIR)**
- **Air quality management planning**
- **Emission reduction plans**
- **Ambient air quality and dust fallout monitoring**
- **Isokinetic stack emission testing**
- **Multi-point calibrations of gas analysers**
- **Emissions inventory compilation, maintenance and NAEIS reporting**
- **Leak detection and repair (LDAR)**
- **Assistance with compliance with AEL conditions for Listed Activities**

NOTEABLE RECENT PROJECTS

- **AIRs for AEL postponment applications**
- **AIRs for proejcts under the Risk Mitigation Independent Power Producer Procurement Programme (RMIPPPP)**
- **Isokinetic stack emission testing for large industrial facilities in Newcastle, Durban, Richards Bay and Port Elizabeth, amongst others**
- **Ambient air quality monitoring and network management for iron ore mines in the Northern Cape**

CONTACT IN DURBAN:
Tel: 031 262 3265
Email: mark@umoya-nilu.co.za or
bheki@umoya-nilu.co.za

Technical article

Developing and testing a $PM_{2.5}$ low-cost sensor in Ethiopia under ambient and indoor air pollution conditions

Johannes Dirk Dingemanse[1,*], **Afework Tademe**[2]

[1]Faculty of Water Supply and Environmental Engineering, Water Technology Institute, Arba Minch University, Ethiopia
[2]Faculty of Electrical and Computer Engineering, Institute of Technology, Arba Minch University, Ethiopia
* Corresponding author: johannesdirk.dingemanse@amu.edu.et

Received: 2 August 2023 - **Reviewed:** 15 September 2023 - **Accepted:** 10 November 2023
https://doi.org/10.17159/caj/2023/33/2.16488

Abstract

$PM_{2.5}$ low-cost sensors are a promising trend for low-income countries, where the $PM_{2.5}$ associated burden of disease is high and few measurement instruments are available. Commercially available Sensor Systems (SSys) are relatively affordable and easy to use. They are, however, not designed for or evaluated in contexts characterized by much biomass burning, regular power interruptions and/or low internet coverage, typical for low-income countries. Alternatively, local teams can build a sensor system with $PM_{2.5}$ sensors from Original Equipment Manufacturers (OEM). Existing African OEM projects depend on international partners and funding. This puts the affordability for local teams without funding into question. Furthermore, field comparisons of such sensors for ambient concentrations and indoor settings are rarely conducted in low-income contexts. In Arba Minch, Ethiopia, we developed a sensor system (SPSA) with the OEM Sensirion SPS30 and other components, together with an Arduino microprocessor, with LoRaWAN data transmission. We used the hardware and software in multiple configurations. The SPSA was used in 14 contexts typical for Ethiopia. During these tests we encountered problems that were easily solved by maintenance on location. On seven locations we collocated the SPSA with itself, gravimetric instruments and/or SSys. Amongst SPSA we found coefficients of determination (R^2) of at least 0.98 for three ambient and one indoor location. The accuracy in comparison with the gravimetric method was 16% under ambient and 13% under indoor circumstances. This is lower than the internationally required 25%. The R^2 in comparison to two SSys was 0.91-0.98 under ambient and 0.88-1.00 under indoor circumstances. The SPSA is a versatile sensor system that can be used in both ambient and indoor air pollution circumstances. Local development without international partners and funding resulted in local experience gaining, low costs, local ownership, and the possibility of tailoring the system to local needs regarding power and connectivity.

Amharic abstract

ጥቂት የመለኪያ መሰሪያዎች ብቻ ባሉባቸዉና $PM_{2.5}$ ጋር በተያያዘ በሽታ ከምስ። ገበያ ላይ የሚገኙ ሴንሰር ሲስተሞች (SSys) በአንጻራዊነት ሲታዩ ተመጣጣኝ እና ለመጠቀም ቀላል ናቸው። ይሁን እንጂ እነዚህ ሴንሰሮች ብዙ ባዮማስ (ተረፈ ምርት) በሚቃጠልበት፣ መደበኛ የሃይል መቆራረጥ እና/ወይም ዝቅተኛ የኢንተርኔት ሽፋንና ዝቅተኛ ገቢ ያላቸውን አገሮችን ሁኔታ ከግምት ዉስጥ በማስገባት የተነደፉ ወይም የተገመገሙ አይደሉም። ተጠቃሚዎች በአማራጭነት የአገር ዉስጥ ኦሪጂናል የፋብሪካ ዉጤቶችን (OEM) በመጠቀም የ $PM_{2.5}$ ሴንሰሮችን መስራት ይችላሉ። አሁን ላይ ያሉት የአፍሪካ የኦሪጂናል ዕቃ አምራች ፕሮጀክቶች በአለም አቀፍ አጋሮች እና የገንዘብ ድጋፍ ላይ የተመሰረቱ ናቸው። ይህም ያለ የገንዘብ ድጋፍ የሚሰሩትን የሀገር ውስጥ ተመራማሪዎችን አቅም ጥያቄ ውስጥ ይከታል። በተጨማሪም እነዚህን ሴንሰሮች በመጠቀም የቤት ዉስጥና የዉጭ አየር ብክለት መጠን ማነጻጸር በታዳጊ አገሮች ላይ እምብዛም አይካሄዱም። በዚህ ምርምር በአርባ ምንጭ OEM Sensirion SPS30ን እና ሌሎችንም እቃዎች ከአርዲኖ ማይክሮፕሮሴሰርና LoRaWAN ዳታ ማስተላለፊያ ጋር በመጠቀም ተሰርቷል። በዚህም መሰረት የተለያዩ ሃርድዌሮችንና እና ሶፍትዌሮችን በበርካታ ውቅሮች ውስጥ ተጠቅመናል። ለኢትዮጲያ እንዲመች ተደርጎ በ14 የተለያዩ ቦታዎች ላይ ጥቅም ላይ አዉለነዋል። በነዚህ ሙከራዎች ወቅት በቦታው ላይ በተደረገ ጥገና በቀላሉ የሚፈቱ ችግሮች አጋጥመውናል። በሰባት ቦታዎች ላይ SPSAን ከራሱ፣ ከግራቪሜትሪክ መሳሪያዎች እና/ወይም SSys ጋር አቀናጅተን አስቀምጠናል። በዚህም መሰረት SPSAን በመጠቀም በሶስት የዉጪ እና በአንድ የቤት ዉስጥ ሴንሰሮች ላይ የመገናኘት መጠን (R^2) 0.98 አግኝተናል። የትክክለኛነት መጠኑም ከግራቪሜትሪክ ዘዴ ጋር ሲነጻጸር በከባቢ አየር (ከቤት ዉጪ) 16% እና በቤት ውስጥ ሁኔታዎች ውስጥ 13% ነው። ይህም በአለም አቀፍ ደረጃ ከሚያስፈልገው 25% ያነሰ ነው። R^2 ከሁለት SSys ጋር ሲነጻጸር 0.91-0.98 በቤት ዉጭ እና 0.88-1.00 በቤት ውስጥ ሁኔታዎች ውስጥ ነበር። SPSA ሁለገብ ሴንሰር ስርዓት ሲሆን በሁለቱም የአካባቢ (ከቤት ዉጪ) እና የቤት ውስጥ የአየር ብክለት ሁኔታዎች ውስጥ ጥቅም ላይ ሊውል ይችላል። በዚህም መሰረት የአገር ውስጥ ልማት ያለ ዓለም አቀፍ አጋሮች እና የገንዘብ ድጋፍ የሀገር ውስጥ ልምድን በማስገኘት ፣ ዝቅተኛ ወጭ በማስወጣት ፣ የአካባቢ ባለቤትነት በማስገኘት እና ስርዓቱን ከኃይል እና ተያያዥነት ጋር ለአካባቢያዊ ፍላጎቶች የማበጀት እድል አስገኝቷል።

Keywords

ambient air pollution, indoor air pollution, $PM_{2.5}$, low-cost sensor, Sensirion SPS30, measurement network, Arduino

Introduction

Air pollution is amongst the top risk factors for the global disease burden (Babatola, 2018; Shaddick et al., 2018), placed second for data of 2019 (IHME, 2021). This burden is relatively higher in low-income countries, due to sources like open waste burning and cooking with biomass fuel (World Health Organization, 2021, 2022). Paradoxically, resources for measurements are lowest in those countries. A primary indicator for air pollution in indoor and ambient situations is particulate matter with a diameter smaller than 2.5 µm ($PM_{2.5}$) (World Health Organization, 2021). The reference method for monitoring $PM_{2.5}$ is filter-based gravimetry. This method typically assesses concentrations on a 24-hour average level (EPA, 2006; European Commission, 2010), and is associated with high operating costs (Sousan, Regmi and Park, 2021). Various continuous monitors (monitoring concentrations at hour- or even second level) are recognized as equivalent to the reference method, such as the Beta Attenuation Monitor (BAM), the Tapered Element Oscillating Microbalance (TEOM), and Palas Fidas. These are expensive as well, with costs of $11 500 – $30 000 per monitor (Mooney, Willis and Stevenson, 2006), and technically still need calibration to the gravimetric reference method. In high-income countries the application of such monitors is widespread, and measured data is usually openly available. For African countries, however, only few have some coverage (Subramanian and Garland, 2021).

A promising trend is the development of low-cost sensors (LCS). Karagulian et al. (2019) distinguish two types of LCSs: only a sensor as developed by original equipment manufacturers (OEM) and sensor systems (SSys), which combine an OEM with encasing, sampling system, power system, hardware, software, and data acquisition methods. SSys cost between $200 and $500, whereas self-built systems with OEMs for measuring $PM_{2.5}$ can be created for $50. Using SSys does not require skills in programming and electronics, but their internet and power requirements are challenging (Sewor, Obeng and Amegah, 2021). Moreover, SSys are still expensive for African researchers (Subramanian and Garland, 2021), and data ownership of data collected through the network functionality lies with the producer. These issues plea for a focus on OEM. Creating an own system based on an OEM is cheaper, and the users can decide to combine the OEM with alternative energy sources (power bank, solar panel) and network connectivity (LoRaWAN, GSM), according to their context. This makes OEMs more compatible to local climates (Subramanian and Garland, 2021). Working with OEMs requires programming and electronics proficiency. When a local team develops a setup with an OEM, there is local experience gaining, local ownership of instruments and data, and local decision making. Apart from making a system with an OEM locally, it is important to evaluate it under the conditions where it will be used. Parameters such as particle density, particle hygroscopicity, refraction index, and particle composition strongly affect the operational principle of $PM_{2.5}$ LCS (light scattering). All these factors vary from site to site and with seasonality (Karagulian et al., 2019). Open biomass burning, cooking with biomass fuel and older vehicles are air pollution sources typical for a low-income country. The data quality of an OEM should be evaluated under such circumstances.

Three examples of OEM projects in the African continent are https://sensors.africa/air (OEM SDS011), Ngom et al. (2018) (OEM HK-A5) and Airqo.net (Bainomugisha, Ssematimba and Okure, 2023; OEM Plantower PMS 5003). Sensors.Africa and AirQo have international partners and funding. This raises the question whether local development without funding is possible. Furthermore, we only found field tests of data quality in a low-income country context for AirQo. Adong et al. (2022) compared the AirQo to a BAM at two ambient locations in Kampala, Uganda. We did not find any field comparisons under indoor air pollution circumstances for these OEM projects.

Dingemanse (2022) presented the evaluation of three LCS under field conditions in Arba Minch, Ethiopia. One of these LCS was a locally developed system with the OEM Sensirion SPS30 and an Arduino microprocessor. In this article, we present this system and show the field evaluation in more detail. We refer to this system as SPSA, named after the sensor (SPS) and the microprocessor, being Arduino (A). The SPSA has a power back-up and can transmit its data with LoRaWAN technology. This makes the setup compatible with a context that often lacks stable power supply and WiFi. We conducted extensive field comparisons in Ethiopia, at three ambient and four indoor high concentration locations. We collocated the SPSA with itself, with a gravimetric measurement instrument, and with two SSys: IQAir Airvisual Pro (the SSys of Airvisual, which is the largest real-time air quality databank (Wernecke et al., 2021)) and PATS (designed for indoor air pollution measurements). We developed the system and conducted the field comparisons without funding.

The next part of the article includes a description of the system we used and evaluated, followed by an evaluation of its operational reliability and an evaluation of the data quality. This article ends with a discussion and conclusion on the implications of the operational reliability, data quality and field evaluations in general. While we included all parts of a working sensor system with network aspect, our aim is not to present a finished product. Rather, it is a showcase of the potential of Do-It-Yourself development with OEM within Ethiopia. Our unique contributions are the presentation of a sensor system developed locally without international partners or funding, and field tests of this system across both ambient and indoor circumstances in Ethiopia. These are the first field tests of the OEM SPS30 in a low-income country. We hope it inspires the reader to conduct similar projects, including field comparisons, in low-income country contexts. All software and data analysis code and raw data, used in this study, is shared on an OSF repository: https://doi.org/10.17605/OSF.IO/DXEZ8.

Set-up description

Hardware

Components and connections

Core components of the system are a $PM_{2.5}$ sensor, SD card module and Real Time Clock (RTC), connected to a microprocessor. Optional components include a Relative Humidity sensor,

LoRaWAN communication shield and LED control light. Table 1 gives an overview of the components.

Table 1: *SPSA components.*

No.	Component	Description
1	Arduino Mega 2560	Microprocessor
2	SPS30 Sensirion	$PM_{2.5}$ sensor
3	DS3231	Real Time Clock
4	Micro SD card reader	Module for SD card, for offline data storage
5	BME280	Relative humidity and temperature sensor
6	Dragino LoRa shield	Module for transmission of data over Long Range (LoRa) network, 868 MHz
7	LED	Green LED for instrument status

The Dragino LoRa Shield is a premade component that fits directly on the pin configuration of the microprocessor. Other components of the system are connected to the main circuit board via shielded connecting wires. The wires are kept short, less than 10 centimeters, to avoid electromagnetic interference and crosstalk. Power supply connections are shared to avoid wiring complexity. Also, the SDA and SCL connection points are shared between the SPS30 and DS3231. Figure 1 shows a sketch of the circuit.

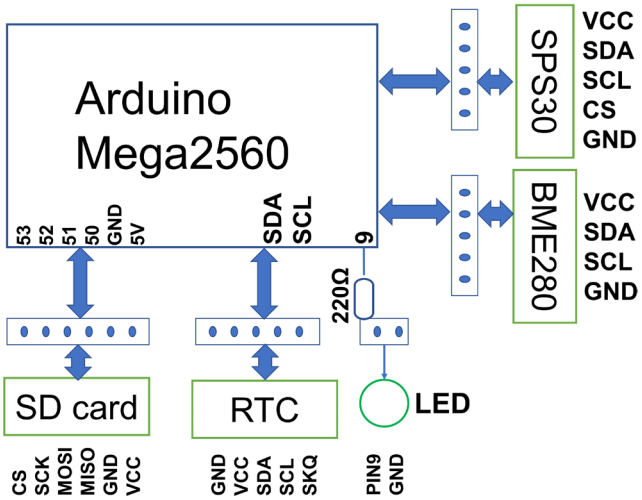


Figure 1: *Sketch of the SPSA circuit. LoRaWAN is not included in the sketch; the Dragino LoRa shield is premade to fit directly on the normal Arduino Uno or Mega pin layout.*

Configurations

The SPSA hardware was combined with a power bank in plastic boxes (lunch boxes or plastic cups). The boxes were chosen based on availability on the local market and the intended use. Three configurations have been used:

- C1: A larger box for stationary sampling, including LoRa and BME280
- C2: A smaller box for stationary sampling, without LoRa and BME280
- C3: A cup-box for personal sampling, without LoRa and BME280.

Figure 2 shows photos of the three configurations.

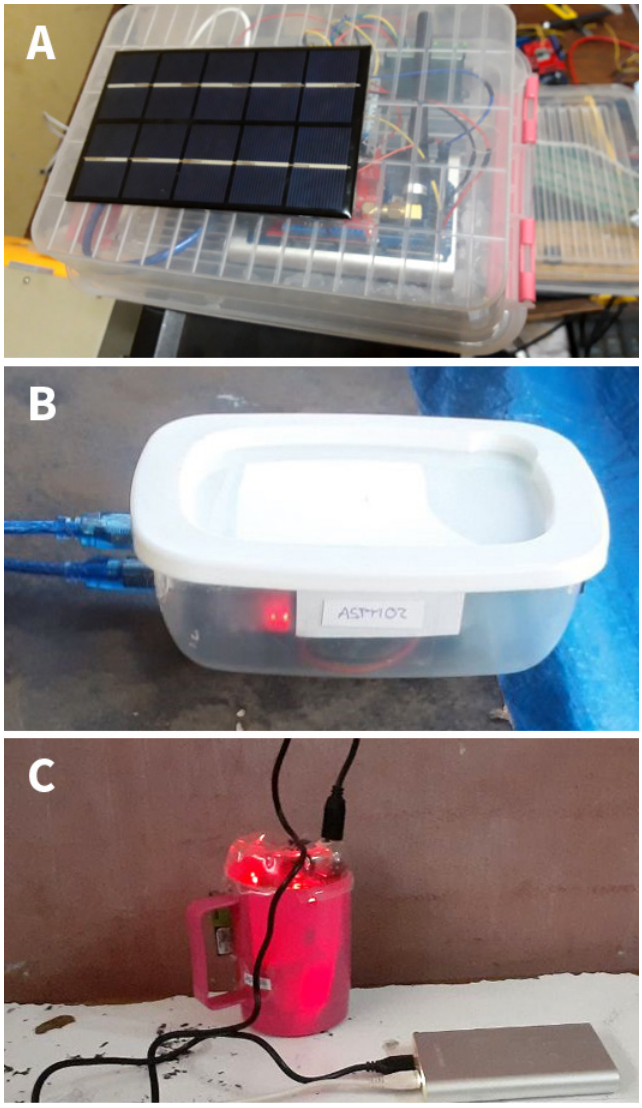


Figure 2: *Configurations of the SPSA. (A) Stationary sampling box with LoRa and BME280. (B) Stationary sampling box without LoRa and BME280. (C) Personal sampling box without LoRa and BME280.*

The air sampling and ventilation openings of the SPS30 are at the frontside of the sensor. In the plastic boxes a hole the size of the SPS30 was cut, and the SPS30 was placed through the hole for approximately 5 millimeters. In this way, the sensor was exposed to the ambient air while keeping most of the sensor safe inside the box. Also, because the hole was exactly sized to the SPS30, movement of the sensor within the SPSA was not possible. All other components were kept at their place with glue. For configuration C1, the BME280 air and humidity sensor was placed outside the box. A hole was cut inside the box for exactly the size of the BME280 wires. Configuration C3's casing was a cup, so that it could be fixed to a belt.

Power consumption

Based on their respective data sheets, the power consumption of the SPSA components is as follows:

1. Arduino Mega 2560: 5 V x 50 mA = 250 mW.

2. SPS30: 5 V x 80 mA = 400 mW.
3. DS3231: 5V with a maximum of 0.65 mA = 3.26 mW. A rechargeable battery is included as back-up.
4. Micro SD card with adapter: maximum standby current of 1 mA x 5 V = 5 mW.
5. BME280: 0.0036 mA at 3.6 V = 0.013 mW.
6. LoRaWAN: 5 V x 32 mA = 160mW during transmission, and 5 V x 1 mA = 5 mW during sleep mode. This transmission is used once every five minutes. We assume a transmission time of maximum 20 seconds. This results in per hour 20 * 12 = 240 seconds of transmission mode, and remaining time sleep mode.
7. LED: 40 mA x 5 V = 200 mW when in use. A blink time of 0.2 seconds is used per minute, leading to a usage time of 0.2 * 60 = 12 seconds per hour.

Table 2 gives an overview of the SPSA hourly power consumption.

Table 2: *Power consumption calculation for the SPSA.*

No.	Component	Consumption (mW)	Operation time %	Average hourly consumption mW
1	Arduino Mega 2560	250	100	250
2	SPS30 Sensirion	400	100	400
3	DS3231	3.25	100	3.25
4	Micro SD card reader	5	100	5
5	BME280	0.013	100	0.013
6	Dragino LoRa shield	Transfer mode: 160 Sleep mode: 5	7 93	10.7 47
7	LED	ON mode 200 OFF mode 0	0.3	0.7
	Total			674

Most commercial power banks have a capacity of 6-10 Ah at 5 V. In case of connection to an irregular power grid, a 6 Ah power bank can provide back-up power during power interruptions for 6 Ah / (0.674 W / 5 V) = 44 hours. In combination with a 1.5 W solar panel, the power bank can be charged during daytime and used during nighttime. The battery capacity for 12 hours, including a maximum depth of discharge of 20%, should then be (0.674 W * 12 h) * 1.2 / 5 V = 1.94 Ah.

Introduction of a sleep mode for the SPS30 can decrease power consumption. Under certain conditions, it is possible to achieve the same monitoring quality through point measurements taken every 15 minutes. This would reduce the power consumption of the SPS30 sensor with at least 90%. As a result, the total system consumption would drop to 314 mW from its current level of 674 mW.

Software

Figure 3 shows a flowchart of the SPSA operation process.

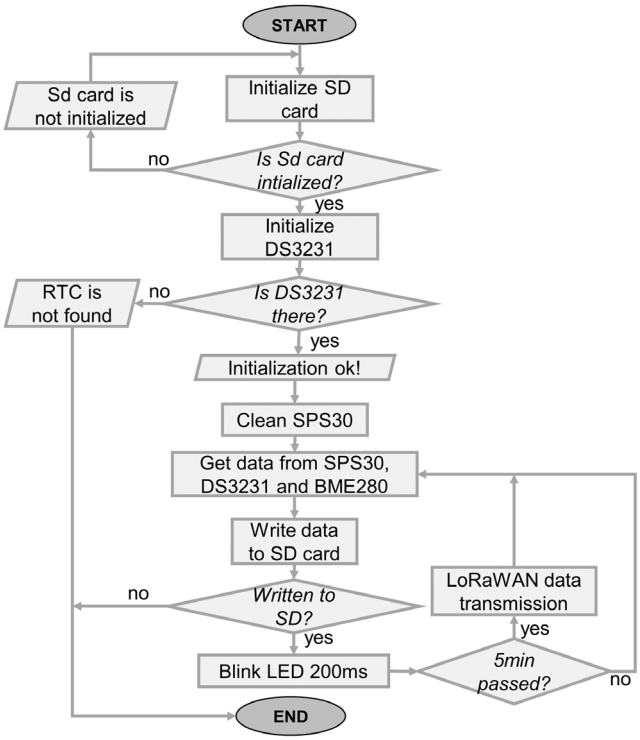


Figure 3: *Flowchart of software on the SPSA.*

SPSA software

Three versions of software have been used:

- For C1, 1-minute average
- For C2, 1-minute average (C2_1)
- For C2 and C3, 10-second point measurement (C2_2 and C3).

All three software scripts are included in the supplementary materials and available at the OSF repository (https://doi.org/10.17605/OSF.IO/DXEZ8).

LoRa network software

With LoRa functionality, data is sent to a gateway. The gateway is registered at The Things Network (TTN; https://www.thethingsnetwork.org/). If the receiving gateway is connected to the internet, the data is stored on the TTN server for seven days. Within those seven days, data can be retrieved from the server. For proof of concept, we created Python code for the following tasks:

- Retrieving data from TTN
- Creating a graph of the data
- Uploading a graph to a website.

Example code is uploaded to the OSF repository (https://doi.org/10.17605/OSF.IO/DXEZ8). If this code runs on an online server, data sent over LoRa can be turned into online visualizations. This creates a real-time measurement network. This data transmission method can also be used to check the SPSA, such as the general operation, the real-time clock, and the measured values.

Operational reliability evaluation

This section discusses to what extent the SPSA, our setup, is operationally reliable in the context of Arba Minch and Addis Ababa. The evaluation methodology section describes how the SPSA is used under different circumstances. The operational reliability section describes the findings on operational reliability based on this use.

Evaluation methodology

The SPSA is used under various circumstances: indoor and outdoor, stationary, and mobile and in high concentration and low concentration environments. For stationary measurements, the instruments were placed between 1.5–2.5 meters above the ground. Mobile measurements have been taken by placing the instrument in the frontside of a public transport tricycle, or by letting students walk around with the instrument on a belt (personal sampling). Most measurements have taken place in Arba Minch, but two set-ups have been installed in Addis Ababa. A full overview of measurement circumstances is given in Table 3.

Measurements at locations A1-A3 and K1-K4 were conducted for an evaluation of the SPSA data quality. These locations are selected for their distinct ambient and indoor concentration levels. Regarding these locations, more information is given under the measurement locations section of the data quality evaluation. Other measurements were part of student measurement projects. Locations and durations were chosen by the students.

In situations 4, 8, 9 and 14, the instruments were used with a power bank that was charged before each use. In all other situations a continuous power supply was maintained by connecting the instruments to the grid via the power bank.

In situations 1, 2, 10 and 11 the SPSA was deployed with LoRa functionality. Two gateways were installed in Arba Minch. A full network structure including database was not yet built. Instead, the focus was on 'proof of concept': for a certain time having the gateways up and testing the reception at different distances.

Operational reliability

This section describes the experiences related to the operational reliability of the SPSA which were gained while conducting the measurements as described in the evaluation methodology section.

Instrument

Single SPSAs functioned well in all measurement locations, except for the following three issues:

- In the first period of field measurements (No. 1, 2), one type of SD cards (brand FFFAS, 16 GB) malfunctioned. After it happened in multiple SPSAs, other brands of SD cards were used.
- In some cases (No. 1, 2, 3 7, 8, 14), the battery of the Real Time Clock died, resulting in a reset of the time. In most cases, data could be restored by comparing the (wrong) time on the instrument with the actual time, and changing the time based on that difference. Validation of this was done based on comparison with collocated sensors and occurrence of daily concentration peaks.
- In two cases (No. 1 and 2), an SPS30 sensor was giving extremely high values without an air pollution source nearby. The sensors in question were replaced.

Table 3: *Details on measurement locations of the SPSA, including per location the SPSA type, number of SPSAs (#) and the amount of data collected at that location (data quantity)*

No.	Measurements		Measurement location		SPSA		Data quantity
	Type	Method	ID	Description	Type	#	
1	Ambient	Stationary	A1	Arba Minch, quiet neighborhood	C1	3	9889 hours
2	Ambient	Stationary	A2	Arba Minch, center	C1	2	3874 hours
3	Ambient	Stationary	A3	Addis Ababa, Tikur Anbessa area	C2_2	2	12 642 hours
4	Ambient	Stationary	A4	Quiet road, entrance to university	C2_1	1	321 hours
5	Ambient	Stationary	A5	Arba Minch, busy traffic square	C2_1	1	643 hours
6	Ambient	Stationary	A6	Arba Minch, busy traffic square	C2_1	1	687 hours
7	Ambient	Stationary	A7	Arba Minch bus-station	C2_1	1	258 hours
8	Ambient	Mobile	A8	Arba Minch public transport tricycle	C2_2	1	265 hours
9	Ambient	Mobile	A9	Arba Minch public transport tricycle	C2_2	1	259 hours
10	Indoor	Stationary	K1	Next to wood cooking kitchen, in same room charcoal burning	C1	1	993 hours
11	Indoor	Stationary	K2	In kitchen, above wood cooking location	C1	1	96 hours
12	Indoor	Stationary	K3	Wide kitchen, various wood cooking places	C2_2	2	457 hours
13	Indoor	Stationary	K4	Wide kitchen, various wood cooking places	C2_2	2	377 hours
14	Ambient & indoor	Mobile	P1	Student personal sampling	C3	3	647 hours

Despite the above-mentioned problems, many hours of data have been collected. This shows that the SPSA is operationally reliable. The setup was able to collect data under ambient (1-7), on-road mobile (8,9), indoor (10-13) and personal (14) circumstances. It could be used by staff (No. 1-3, 10-13) and students (No. 4-9, 14). Regular check-up is advisable, considering the above-mentioned problems. However, we found that also without check-up some systems could operate without errors for a long period. In Addis Ababa (No. 3) the instruments were installed in July 2022. For logistical reasons there was no check-up from the end of July 2022 until May 2023.

Network

The network potential of a group of SPSAs works well with LoRa. The transmission over LoRa is largely dependent on the proper placement of the gateway. Messages have been received over an 8 km distance. This reception is greatly influenced by obstruction of line-of-sight. One gateway received significantly less messages from the direction where a large building was obstructing line of sight to the SPSA. For about two weeks a full 'network' was running: SPSAs with LoRa functionality were deployed, gateways were up and running, and the Python scripts, as listed in the LoRa network software section, ran on an hourly interval at a Virtual Private Server. During those weeks, an hourly updated real-time graph showed live $PM_{2.5}$ concentrations on various locations in Arba Minch.

Data quality evaluation

This section discusses the field testing we conducted with the SPSA for the purpose of data quality evaluation. The evaluation methodology section discusses the methodology we use for evaluating data quality. The concentration patterns section displays what concentrations were measured (to validate that measurements were taken under various conditions). The remaining sections show the data quality metrics for the three comparison types we used: within, gravimetric and SSys.

Evaluation methodology

Measurement locations

Table 3 listed 14 locations where the SPSA has been used. For data quality evaluation, three ambient locations (A1-A3) and four indoor locations (K1-K4) were selected. Only at these locations, the SPSA was collocated with itself and/or other instruments. For the three ambient locations, sources of $PM_{2.5}$ are traffic and neighborhood biomass burning. The three ambient locations represented three distinct concentration levels. Two locations were in Arba Minch, and one location in Addis Ababa, sub-city Lideta. According to latest projections, population of Arba Minch town, Addis Ababa, and its sub-city Lideta are 210 255, 3 945 000 and 290 466, respectively (Ethiopian Statistics Service, 2023). The population in Addis Ababa is higher than in Arba Minch, and the number of vehicles per person is also higher. As per 2020, Addis Ababa had registered 630 440 vehicles, while the region SNNP (of which Arba Minch is a part), with a population estimation of 13 044 044, had registered 118 424 vehicles (Abiye, 2020). Location **A1** was in front of a house in a quiet neighborhood in Arba Minch (latitude 6.01589 N, longitude 37.55480 E). At the compound of this house, cooking only is done with electric stoves. The nearest road to this location is approximately 120 meters, and this road is only used by local traffic. Location **A2** was at the entrance of a hotel compound in the center of Arba Minch (latitude 6.03311 N, longitude 37.55783 E). The instrument was approximately 15 meters from the nearest road. This road is the main road through Arba Minch. It is used by traffic driving through Arba Minch, and by stopping and starting traffic visiting businesses. Location **A3** was in Addis Ababa, sub-city Lideta, at the city monitoring station of the Ethiopian Meteorology Institute (latitude 9.01888 N, longitude 38.74728 E). This location is approximately 10 meters from the nearest road.

The four indoor locations were selected for their use of biomass fuel, to represent (extremely) high and variable concentrations. All four locations were in Arba Minch. Location **K1** was in a small local restaurant, in a room connected to the kitchen. In that room, charcoal fire was used for coffee preparation. In the kitchen, food was prepared with biomass fuel. Location **K2** was in the kitchen of a student restaurant. In this kitchen, food was prepared with biomass fuel. Locations **K3** and **K4** were in a large kitchen with more than six biomass fuel cooking locations. These two locations were on opposite sides of the kitchen, and cooking fires closest to the respective locations were used at different times. For this reason, these locations are considered separate.

Measurement comparison types

We evaluate data quality by comparing measurement results of an SPSA with instruments that are collocated. Three types of measurements have been conducted for data quality evaluation. The **first** type compares measurement results of SPSAs amongst each other (*within* comparison). We placed multiple SPSAs in the same location.

The **second** type compares SPSA with a gravimetry instrument (*gravimetric* comparison). Reference measurement methods for $PM_{2.5}$ are based on gravimetry. As gravimetric instrument, we used the Ultrasonic Personal Aerosol Sampler (UPAS), as this instrument was the only available gravimetric instrument in Arba Minch, Ethiopia. The UPAS is designed for measuring medium to high concentrations. Over ranges of 20–1000 µg/m^3, Volckens et al. (2017) found strong correlations with the EPA federal reference method. Afshar-Mohajer et al. (2021, p. 131) found that 'the UPAS may be a suitable alternative for [Respiratory Dust] mass sampling' for ranges of 100–500 µg/m^3 in occupational settings. We conducted gravimetric analysis of the filters with a Mettler AE240 Dual Range balance. This balance has a readability of 10 µg and a reproducibility of ±20 µg (IET, no date).

For Addis Ababa (3), there were no gravimetric measurements taken by us. At another location (Jacros area, Addis Ababa, distance to location A3 8 km, latitude 9.01163 N, longitude 38.82151 E), the US Embassy operates a BAM. A BAM is

Table 4: *Overview of measurements used for the data evaluation of the SPSA. Individual SPSAs are labeled Spx, where x refers to the setup ID number. IQAV and PATS are coded Iqx and Pax, respectively.*

Location	SPSA	Gravimetry	SSys	Period[1]
A1	Sp1, Sp2		Iq2	April 2021 – April 2022
A2	Sp3, Sp5	UPAS (3 samples)	Iq1	June 2021 – October 2021
A3	Sp11, Sp12	Indicative: BAM, gravimetry	Iq7, Iq10	July 2022 – April 2023
K1	Sp2, Iq5		Iq5	June 2021 – August 2021
K2	Sp4, Iq3	UPAS (3 samples)	Iq3	1–5 October 2021
K3	Sp6, Sp7	UPAS (2 instruments, both 4 samples)	Pa1, Pa3, Pa4	7–18 June 2022
K4	Sp8, Sp9	UPAS (4 samples)	Pa2, Pa5, Pa6	7–18 June 2022

[1] Not all instruments were available during the whole period. Therefore, the relationship between the number of data pairs in evaluations and the period duration is not linear.

Table 5: *Data quality metrics.*

Metric	Source	Requirement	Scope[1]	Calculation method
Slope (S) and coefficient of determination (R^2)	Often used	R^2>0.75 (R^2>0.9 'very good')	All	Calculation resulting from Ordinary Least Squares (OLS) regression without intercept.
Coefficient of Variation (CV)	EPA, NIOSH	<10%	Within	$CV = \frac{1}{n}\sum\frac{\sigma_i}{\mu_i}$, where σi is the standard deviation and μi is the mean of measurements of identical LCS during time period i, and n is the number of time periods.
Between-sampler uncertainty (u_{bs})	DEM	<2.5 µg/m³	Within, ambient, daily	$u_{bs} = \sqrt{\frac{\sum(y_{i,1}-y_{i,2})^2}{2n}}$, where $y_{i,1}$ and $y_{i,2}$ are the results of parallel measurements for time period i, and n is the number of time periods.
Pearson correlation coefficient (r)	EPA	>0.97	Gravimetric	
Bias (B)	NIOSH	Correction if >10%	Gravimetric	$B = \frac{1}{n}\sum(\frac{x_i}{y_i} - 1)$, where x_i is the concentration of the LCS and y_i the concentration of the reference instrument for time period i, and n is the number of time periods. Corrected data xnew based on the old data x_{old} was calculated with $x_{new} = \frac{x_{old}}{B+1}$
Accuracy (Ac)	NIOSH	<25%	Gravimetric	The upper value of the confidence interval at 90% of all $\frac{x_i}{y_i}$, where x_i is the concentration of the LCS and y_i the concentration of the reference instrument for time period i. We used the names $Ac_{\_BC}$ and $Ac_{\_AC}$ to distinguish between accuracy before and after bias correction, respectively.

[1] Not all metrics apply in all situations. Where applicable, the table lists the scope of application with respect to type of comparison (within, gravimetric, SSys), time-averaging period (10-minute, hourly, daily, filter-duration or monthly) and/or context (ambient, indoor).

Table 6: *Data quality evaluation summary.*

Evaluation	Time periods[1]	Metric	Locations
Within (SPSA versus SPSA)	1,2,3 1,2,3 3	S & R^2 CV u_{bs}	A1-A3, K3, K4 A1-A3, K3, K4 A1-A3
Gravimetric (SPSA versus gravimetric method) *Indicative* comparison	4 4 4 4 3, 5	r S R^2 B & Ac S & R^2	A2, K2- K4 A2, K2-K4 A2, K2-K4 A2, K2-K4 A3
SSys (SPSA versus IQAV or PATS)	2,3	IS & R^2	A1-A3, K1-K4

[1] We conducted data quality evaluations at time periods 10-minute (1), hour (2), day (3), filter-duration (4) and month (5).

recognized as equivalent to the gravimetric reference method. Data of this BAM is available online (AirNow Department of State, 2023). We used measurements of this BAM for *indicative* comparison: indicative because the BAM and SPSA are not at the same location. Location A3 is within the city ring roads, while the location with the BAM is closer to the outskirts of Addis Ababa. A second source of indicative comparison is a gravimetry measurement project at the same location as our measurements (A3). During November 2015 – November 2016, Tefera et al. (2020) collected 69 $PM_{2.5}$ samples. Their measurement method is the official $PM_{2.5}$ reference method (gravimetry), and their measurements are at the same location, but their measurements are not during the same time period. We used these measurement results as indicative comparison with respect to the long-term concentration level.

The **third** type compares measurements of the SPSA with two SSys (*SSys* comparison): Airvisual Pro (IQAV) and UCB-PATS+ (PATS). Like the SPS30 in our system, both IQAV and PATS estimate the $PM_{2.5}$ concentration based on scattering of infrared light (Pillarisetti et al., 2017; Zamora, Rice and Koehler, 2020; Sousan, Regmi and Park, 2021). The PATS is designed for personal sampling and (high) indoor concentrations, but not for low ambient concentrations (lower detection limit is 10 µg/m³). Hence, we only used the PATS at indoor locations. We used the IQAV both in ambient and indoor situations. The IQAV is not meant for very high concentrations (>5000 µg/m³), because the highest reported value of the IQAV is set to 4488 µg/m³.

Table 4 shows an overview of all measurements used for data quality evaluation.

Data corrections

In their guidelines for Demonstrating Equivalence of Ambient Air Monitoring Methods (DEM) the European Commission (2010) allows for removal of up to 2.5% percent of outliers based on Grubb's outlier test at 99% level. We removed outliers with this method for Sp3 and Sp5 at location A2, and for Sp11, Sp12, Iq7 and Iq10 at location A3.

The Sp11 and Sp12 comparison is only done from October 2022, because up to that time there was a real time clock (RTC) issue for Sp11. In contrast, the Sp12 / IQAV comparison could only be done with measurements until October 2022, because after October 2022 both Iq7 and Iq10 malfunctioned.

At locations K3 and K4, there was data loss during the gravimetric measurements. We did not use LCS results with more than 15% data loss during measurements with the gravimetric method in the gravimetric comparison.

Data analysis

Metrics for expressing equivalence of instruments that are reported most often, are Slope (S) and a coefficient of determination (R^2) originating from Ordinary Least Squares (OLS) regression. Official guidelines for testing the equivalence of $PM_{2.5}$ measurement methods have been made by the Environmental Protection Agency of the United States of America (EPA) (EPA, 2006), the National Institute for Occupational Safety and health (NIOSH) (NIOSH, 2012), and by the European Commission in the Guide to the Demonstration of Equivalence of Ambient Air Monitoring Methods (DEM) (European Commission, 2010). Evaluation metrics are taken from these sources and summarized in Table 5.

The DEM also prescribes expanded uncertainty (WCM) for ambient reference comparison. That metric applies to reference comparison at 24-hour level. With the gravimetric method available in our study (the UPAS) it was not possible to conduct 24-hour comparison measurements under ambient circumstances. Although Dingemanse (2022) reported this metric, we did not include it in our study.

For data-analysis, we used five time-averaging periods: 10-minute, hour, day, filter-duration, and month. Hour-averages are most used in monitoring networks, while daily averages represent the short-term WHO $PM_{2.5}$ guideline value. Filter-duration is used in the gravimetric comparison, where the duration varied according to the time needed to get sufficient filter load. We only used monthly averages for the indicative comparison. For the SPSA inter-comparison, we also present 10-minute averages. Table 6 shows a summary of the data quality evaluation.

Concentration patterns

Figure 4 shows the average and 95-percentile concentrations per 10-minute period on a day, for locations A1, A2 and A3. This is not meant to compare locations, since they are not from the same time. It is merely to show the order of magnitude of measured concentrations.

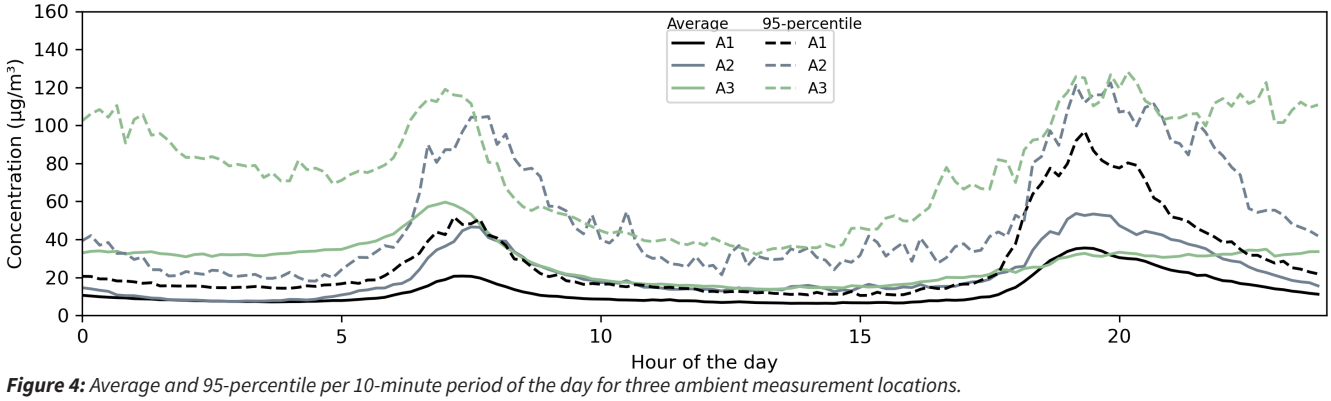


Figure 4: *Average and 95-percentile per 10-minute period of the day for three ambient measurement locations.*

Table 7: *Number of data pairs (N) and data quality metrics for two collocated SPSAs. The results are ordered according to location and time averaging period.*

Averaging period	Parameter	Location A1		Location A2	Location A3	Location K3	Location K4
		Sp1/Sp2	Sp1/Sp4	Sp3/Sp5	Sp11/Sp12	Sp6/Sp7	Sp8/Sp9
Daily	N Slope (R^2) CV[1] [%] u_{bs} [µg/m³]	5 0.94 (1.00) 5.1 0.63	66 1.04 (1.00) 0.34	84 1.05 (1.00) 3.3 0.73	205 1.10 (1.00) 7.0 1.65	11 1.08 (1.00) 5.3	11 1.13 (1.00) 6.6
Hourly	N Slope (R^2) CV[1] [%]	101 0.94 (0.99) 4.8	1539 1.03 (1.00)	1600 1.05 (1.00) 3.6	4909 1.09 (0.99) 8.7	218 1.07 (1.00) 6.2	140 1.14 (0.78) 6.6
10-min	N Slope (R^2) CV[1] [%]	599 0.93(0.98) 5.4	9211 1.02 (0.99)	9436 1.05 (1.00) 4.0	29 365 1.09 (0.99) 9.1	1294 1.07 (1.00) 6.3	788 1.10 (0.87) 5.7

[1] *CV is calculated for any number of collocated instruments, instead of a single pair.*

Average concentrations at locations A1, A2 and A3 are 12.6, 20.6 and 27.8 µg/m³, respectively. 10-minute average concentrations ranged up to 120 µg/m³, in Addis Ababa (A3) this happened both in the morning and in the evening.

Figure 5 shows box plots of 10-minute, hourly and daily averaged concentrations in the four indoor locations K1-K4.

Averages in the four indoor locations ranged from 300 to 2 000 µg/m³, with the highest 10-minute concentrations well over 10 000 µg/m³.

Collocated SPSA (*within* comparison)

Figure 6 shows data pairs for all collocated SPSA measurements, on different time averaging periods. Table 7 shows the data quality metrics.

Both Figure 6 and Table 7 show that multiple SPSA under the same circumstances show an extremely similar signal across all concentration ranges. At location K4, for the hourly and 10-minute averages the R^2 is 0.78–0.87. In all other cases the R^2 is at least 0.98. The CV is lower than the required 10%, and the ubs is lower than the required 2.5 µg/m³.

Gravimetric comparison

SPSA vs UPAS

Figure 7 shows the UPAS gravimetry and SPSA measurement results. Table 8 shows the data quality metrics.

Accuracy is sufficient in all cases (<25%). For ambient circumstances, the Pearson correlation is too low (<0.9). This most probably has to do with the fact that there have only been limited gravimetric measurements in ambient conditions (n=6). Both slope and bias show that under ambient conditions a calibration of a factor 2 is required. For indoor / high exposure conditions this is not required (|B| < 10%). This observation implies that the SPS30 sensor responds differently to aerosols primarily originating from biomass burning compared to those originating from a variety of ambient sources.

Table 8: *Number of data pairs (N) and data quality metrics for SPSA compared to UPAS gravimetric measurements.*

Location	SPSA	N	r	S, R^2	A_{c_BC}	Bias	A_{c_AC}
A2	Sp3,5	6	0.70	1.93, 0.99	53	-0.47	16
K2-4	Sp6-9	21	0.99	0.95, 0.98	15	-0.08	13

Addis Ababa indicative comparison

Figure 8 shows daily and monthly averaged concentrations for Sp12 in comparison with a BAM at another location in Addis Ababa. Despite these not being on the same location, the correlation is good (R^2>0.9). This indicates that the SPSA follows the city-wide concentration trend.

On the same location, between November 2015 and November 2016 the average concentration for measurements with the gravimetric reference methods was 53.8 µg/m³ (Tefera et al., 2020). The average for 10 months of measurements with the Sp12 was 28.7 µg/m³. This indicates a factor two difference with the gravimetric method. This is similar to the calibration factor found between SPSA and gravimetry under ambient circumstances in Arba Minch (see the SPSA vs UPAS section).

Comparison with SSys

Table 9 shows the slope and R^2 for SPSA versus the two SSys, IQAV and PATS. It is shown for all hourly and daily averaged data pairs. For some locations there are multiple SPSA. At those locations, the SPSA which has the most data pairs with the collocated SSys is selected.

Correlations are high to both IQAV in ambient (R^2 0.91–0.98) and PATS in indoor (R^2 0.88–1.00) circumstances. For the IQAV we see similar slopes under similar circumstances. For the PATS there is a rather large variability in witnessed slopes. The IQAV under indoor circumstances gives estimations that are too low because all concentrations higher than 4488 µg/m³ are reported as 4,488 µg/m³. Table 10 shows correlation results between SPSA and IQAV for only <4489 µg/m³ data. For that data, slopes are substantially closer to 1, and R^2s are on average higher.

Table 9: *Data pairs comparison between the SPSA and SSys.*

Location	X	Y	Hourly		Daily	
			N	S (R^2)	N	S (R^2)
A1	Sp1	Iq2	7164	1.08 (0.93)	316	1.12 (0.96)
A2	Sp5	Iq1	1015	1.19 (0.91)	85	1.15 (0.97)
A3	Sp12	Iq7 Iq10	1516 1507	0.74 (0.96) 0.71 (0.95)	67 67	0.79 (0.98) 0.73 (0.96)
K1	Sp2	Iq5	942	0.10 (0.53)	48	0.16 (0.71)
K2	Sp4	Iq3	94	0.17 (0.93)	5	0.19 (0.98)
K3	Sp6	Iq3 Iq4 Iq5 Pa1 Pa3 Pa4	239 239 239 92 92 92	0.36 (0.91) 0.41 (0.92) 0.46 (0.92) 0.59 (0.98) 1.25 (0.88) 0.53 (0.99)	11 11 11 6 6 6	0.46 (0.96) 0.50 (0.97) 0.56 (0.96) 0.59 (0.99) 0.94 (0.97) 0.52 (1.00)
K4	Sp8	Iq6 Iq7 Iq8 Pa2 Pa5 Pa6	132 133 139 90 90 90	0.49 (0.74) 0.62 (0.83) 0.52 (0.90) 0.66 (0.93) 0.57 (0.93) 0.67 (0.90)	11 11 11 6 6 6	0.52 (0.99) 0.65 (1.00) 0.58 (0.99) 0.67 (1.00) 0.56 (1.00) 0.70 (0.98)

Table 10: *Data pairs comparison between the SPSA (only <4489 µg/m³) and IQAV.*

Location	X	Y	Hourly		Daily	
			N	S (R^2)	N	S (R^2)
K1	Sp2	Iq5	942	0.92 (0.90)	48	0.95 (0.99)
K2	Sp4	Iq3	94	0.42 (0.95)	5	0.46 (0.99)
K3	Sp6	Iq3 Iq4 Iq5	239 239 239	0.50 (0.94) 0.58 (0.95) 0.64 (0.92)	11 11 11	0.58 (0.99) 0.63 (0.99) 0.70 (0.99)
K4	Sp8	Iq6 Iq7 Iq8	132 133 139	0.58 (0.75) 0.74 (0.83) 0.62 (0.89)	11 11 11	0.60 (1.00) 0.75 (0.99) 0.66 (1.00)

Discussion

The implications of our showcase involve two areas: the operational reliability and the data quality in contexts as encountered in Ethiopia. In the operational reliability and data quality sections, we discuss these two areas subsequently. We discuss field testing in general in the field testing section.

Operational reliability

Regarding operational reliability it was found that not all systems were without errors. However, the problems were solved by replacing materials such as a sensor or a real time clock battery. A malfunctioning SSys requires shipment to the producer for repair. Our system simply needs replacement of the broken part, which can be done on location. However, to get reliable repair services, knowledge of both air quality and electronics, or a close collaboration with electronics staff is necessary.

The LoRa network functionality worked well if the sensors were placed outside. A network of LoRa will more likely become operational if additional setups, measuring other environmental data like soil and water quality, are used in the study area. As an alternative to LoRa, in bigger cities with stable internet connection, WiFi is an option. Most SSys use this to turn their systems into a network. Alternatively, GSM communication can be used. More generally, with an OEM-based self-built system the data transmission method can be chosen according to the local context. A SSys with WiFi instead forces the user to provide good internet connections or face data loss.

For upscaling the SPSA to a whole network of SPSAs, a higher level of organization is needed. Production, installation, maintenance, data collection and data distribution of many more systems are not likely run by a single initiative taker, such as was the case in this showcase. There are some examples of organized LCS application in the African continent. Awokola et al. (2020) describe the use of the SSys PurpleAir at 13 different sites across 7 African countries. Afriqair.org (Giordano and Jaramillo, 2021) is another African initiative that uses existing SSys (PurpleAir and MetOne). Working with a SSys, however, results in dependency on foreign funding, foreign expertise, specifications that are not necessarily designed for the local context, and it raises questions on data ownership. Local development with OEM sensors, instead keeps the costs lower, gives local staff the opportunity to gain expertise, gives the possibility for systems tailor-made for the local context and gives clear local ownership. We found three African initiatives that create their own sensor system with OEMs: https://sensors.africa/air (OEM SDS011), Airqo.net (Bainomugisha, Ssematimba and Okure, 2023; OEM Plantower PMS 5003), and Ngom et al. (2018) (OEM HK-A5). Sensors.Africa is based on the German initiative Luftdaten, which instead of a SSys has made their system build with OEM completely public and therefore useable by others. They are currently providing measurements in 11 cities across 7 African countries. AirQo.net is modelled after the SSys PurpleAir, since like the PurpleAir they use two Plantower PMS 5003 sensors in their system. They provide their sensor as African-made SSys, and have sensors installed in eight African countries. Both sensors.Africa and AirQo use WiFi to turn their systems into a network of data collection. Ngom et al. (2018) presented separate work of a sensor system used in Senegalese cities, with LoRa data transmission.

Data quality

Regarding the data quality, this case shows high concentration correlations between collocated instruments. This is especially the case for Arba Minch, where abundant measurements were conducted. Less evidence is gathered for Addis Ababa. The strong correlations between SPSAs show high instrument reliability. This implies calibration can happen in retrospect or with other studies. For ambient conditions, the results suggest a strong equivalence with the gravimetric method if calibration is conducted, but more evidence is required. For indoor circumstances, the strong correlation with the gravimetric reference method, in combination with the low bias, shows that the SPSA can be used without calibration. The strong correlations between measured concentrations of collocated

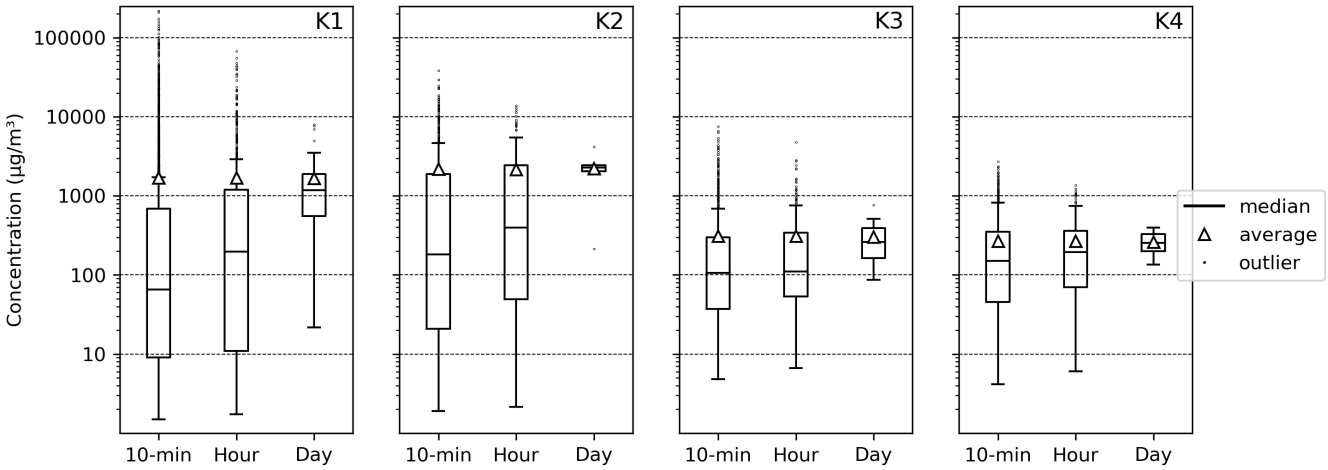


Figure 5: *Box plots of time averaged data for four kitchen locations (K1-K4). The boxes extend from the first quartile (Q1) to the third quartile (Q3) of the data. The whiskers extend from the boxes by 1.5 times the inter-quartile range (Q3-Q1).*

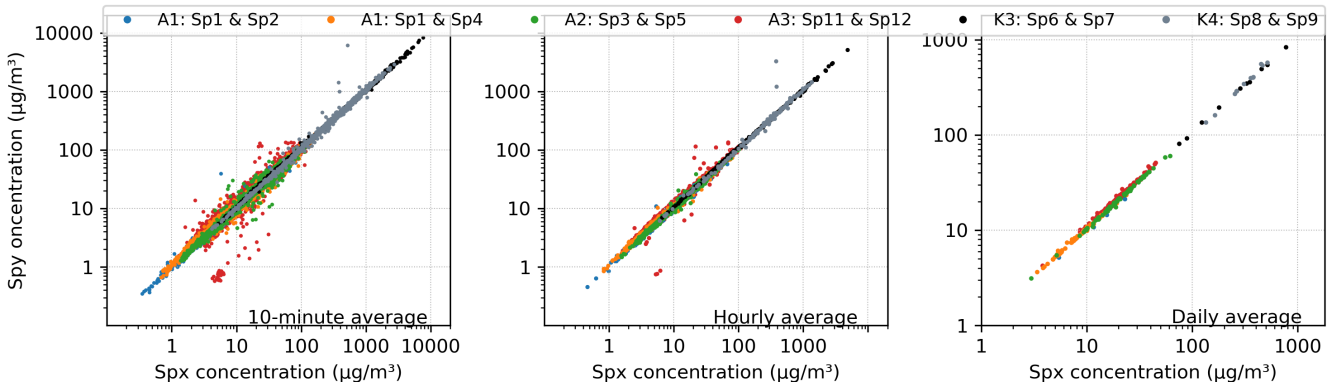


Figure 6: *Scatter plots for all data pairs of collocated SPSA, (A) 10-minute averages, (B) hourly averages and (C) daily averages.*

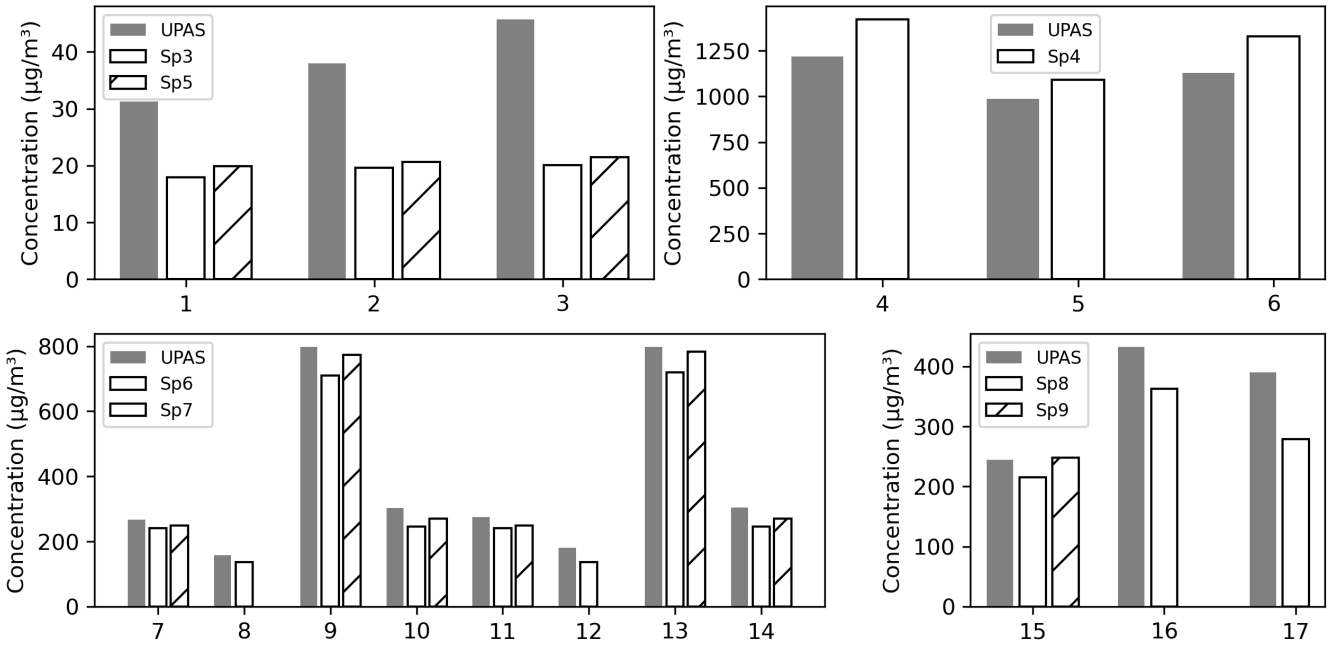


Figure 7: *UPAS and SPSA measurement results for 17 samples across locations A2 (A), K2 (B), K3 (C) and K4 (D).*

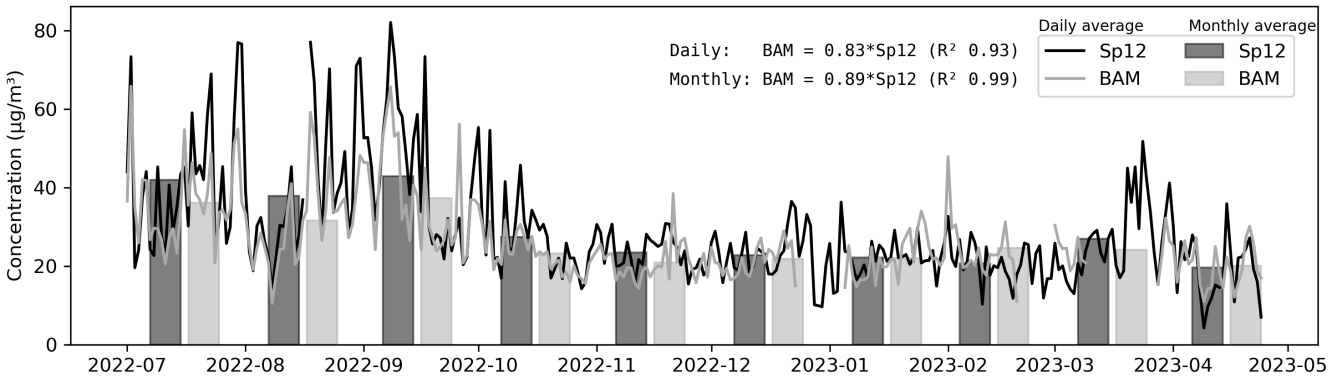


Figure 8: *Daily and monthly averaged concentration for one SPSA and a BAM, at distinct locations in Addis Ababa.*

SPSA and SSys imply that SSys can be used as unofficial calibration instruments in local and remote contexts that have no resources for calibration with a gravimetric method. The large difference in required calibration between ambient and indoor circumstances is most likely due to particle differences (Karagulian et al., 2019). For the SPS30, Sousan et al. (2021) found under laboratory conditions highly linear results (r = 0.99) with a reference instrument for three aerosol types. The slopes for each of the aerosol types, however, were different, ranging from 0.75 to 2.0. This underlines the importance of testing an LCS at the location of intended use. We further discuss this under the field testing section.

Of all OEMs created, we have only used the SPS30 in our setup. Other studies find that of the different OEMs, SPS30 performs well. In their laboratory study, Sousan et al. (2021) compared the SPS30 along with other LCS to a reference instrument. They conclude that the 'SPS30 and OPC-N3 (…) demonstrated the best performance, with high correlation and lowest bias values, for all aerosol types and PM metrics in environmental and occupational settings. (…) In contrast, AirBeam2 and PMS A003 exhibited low accuracy for all aerosol types and PM metrics in both settings.' (Sousan, Regmi and Park, 2021, pp. 23–24). The AirBeam2 uses instruments and calibration like the PurpleAir. The PMS A003 is a Plantower instrument, used both in the PurpleAir and AirQo. A common problem for a $PM_{2.5}$ LCS is influence by relative humidity (RH). However, compared to OEMs Plantower and Honeywell, SPS30 has the lowest reaction to RH (Hassani et al., 2023). According to Budde et al. (2018), the SDS011 (used in sensors.Africa) has a significant variance amongst similar sensors and is strongly influenced by relative humidity. Like Sousan et al. (2021), we observed a robust performance of the SPS30 across different concentration ranges. The sensor showed a small variance amongst collocated identical sensors. The sensor is versatile for both indoor and ambient situations, across all concentration ranges. This was less the case for the two SSys used as comparison in our study (IQAV and PATS). A low influence of RH also follows from the fact that over a period of 10 months the SPSA followed the city-wide concentration trend in Addis Ababa, across both dry and wet seasons.

Field testing

It is important to evaluate LCSs in the same context as where they will be used to monitor air quality. For both SSys and OEMs, field validation in low-income country contexts is limited. Karagulian et al. (2019) conducted a review of 112 different LCS (64 independent studies, 31 OEMs and 81 SSys). From all 64 studies, only one concerned a $PM_{2.5}$ sensor which is evaluated in a low-income country: the SSys PATS inside kitchens in Guatamala (Pillarisetti et al., 2017). Most projects using SSys and OEMs in Africa do not mention field tests at the location of the study. Awokola et al. (2020) present the data of measurements with a PurpleAir 'as is' and refer for field tests to studies conducted at regulatory sites in the USA (Malings et al., 2020). AfricaAir, using the SSys PurpleAir and Met-One, likewise refer to these studies. For OEMs, Adong et al. (2022) compared the AirQo (OEM Plantower) to a BAM at two ambient locations in Kampala city in Uganda, with average concentrations of 37.5 and 45.1 µg/m^3. Wernecke et al. (2021) announced tests of LCS and reference-grade instruments collocated at one ambient location, for half a year in 2021. We were not able to find any published results of this yet. For indoor contexts, we did not find any field validation of OEMs in a low-income country. Also, none of the sparse ambient field validations make mention of comparison with the gravimetric method. In other words, thus far the field validation of LCS in African contexts has been poor, both for indoor and ambient studies circumstances. Our study is the first to compare the SPS30 under indoor field circumstances to a gravimetric method. Our ambient field testing is also a first of its kind in this size (multiple ambient locations across multiple cities). Still also in our study the ambient testing is limited. We have only used the gravimetric method three times at one ambient location. Currently, data collection for the SPSA is ongoing at ambient locations in Addis Ababa and Adama, with 20 collocated gravimetric samples planned. We used the UPAS for both ambient and indoor measurements for comparison, but the UPAS is designed for indoor concentrations. Using a gravimetric instrument made for ambient monitoring (such as a Low Volume Sampler) would make the reliability of the data quality testing stronger. At this moment, such an instrument is not available in Ethiopia.

In any case, a large advantage of using OEMs instead of SSys is that, when a better sensor enters the market, one can decide

to self-assemble the newer sensor in the system. For example, sensors.Africa at this moment uses the SDS011. They are likely to have the infrastructure to switch to an OEM with lower variance and less reaction to RH, like the SPS30 seems to have.

Conclusion

Low-cost sensors are a promising direction for $PM_{2.5}$ monitoring in Ethiopia, and self-developing the sensor system with an OEM even more so. While low-cost sensor projects have started in the African continent, most are externally funded and field validation at the location of deployment is still very poor. Our results from Arba Minch and Addis Ababa show the potential for a cheap, flexible, and reliable $PM_{2.5}$ measurement instrument, across both ambient and indoor pollution circumstances, based on the OEM Sensirion SPS30. To use it in larger quantities (such as a network), or for others to use it, either a more official approach is required (with the danger of losing ownership), or good cooperation between experts of different professions. Challenges of working with an OEM are the required expertise and lower ease of use compared to a SSys. These challenges should be seen as opportunities for increasing local skill, local ownership, instruments tailormade for local use, and for achieving the highest data quality for the lowest costs.

Acknowledgements

We thank Meseret Tesfaye for support in constructing SPSA set-ups, and Mekuanint Hailemichael for arranging the installation and the network connection of a LoRa gateway. Measurements at the locations have been possible because of the owners of the restaurants and hotels allowing measurements at their premises, and undergraduate students of Water Supply and Environmental Engineering (G3 and G4 2021/2022, G3 2022/2023) assisting in those measurements. We also thank Tofikk Redi of the Ethiopian Meteorology Institute (EMI) for assisting in the placement of SPSA and IQAV instruments at the measurement location of EMI in Addis Ababa. Finally, we thank Dr. Zelalem Abera Angello for making an Amharic translation of the abstract.

Author contributions

Johannes Dirk Dingemanse: conceptualisation; methodology; data collection; sample analysis; data analysis; validation; data curation; writing – the initial draft; writing – revisions; student supervision; project leadership; project management. Afework Tademe: methodology; data collection; validation; writing – the initial draft.

Data availability

The data generated and/or analysed for the current study (measurement data and Python scripts) is available in the OSF repository https://doi.org/10.17605/OSF.IO/DXEZ8. For locations A1, A2 and K1-K4, data quality evaluations were first shown in Dingemanse (2022). Processed data files for those locations are taken from that study. Preprocessing and raw data for those locations are in the OSF repository https://doi.org/10.17605/OSF.IO/YTV79.

References

Abiye, Y. (2020) *Registered Vehicle In Ethiopia Reaches 1.2 Mln | The Reporter | Latest Ethiopian News Today*. Available at: https://www.thereporterethiopia.com/10186/ (Accessed: 25 September 2023).

Adong, P., Bainomugisha, E., Okure, D. and Sserunjogi, R. (2022) 'Applying machine learning for large scale field calibration of low-cost $PM_{2.5}$ and PM_{10} air pollution sensors', *Applied AI Letters*, 3(3), p. e76. Available at: https://doi.org/10.1002/ail2.76.

Afshar-Mohajer, N., Foos, R., Ramachandran, G. and Volckens, J. (2021) 'Field Evaluation of the Ultrasonic Personal Aerosol Sampler (UPAS) for Respirable Dust Exposure in a Taconite Mine', *Annals of Work Exposures and Health*, 65(1), pp. 127–135. Available at: https://doi.org/10.1093/annweh/wxaa094.

AirNow Department of State (2023) 'Addis Ababa Jacros $PM_{2.5}$ concentration'. Available at: https://www.airnow.gov/international/us-embassies-and-consulates/#Ethiopia$Addis_Ababa_Jacros.

Awokola, B.I., Okello, G., Mortimer, K.J., Jewell, C.P., Erhart, A. and Semple, S. (2020) 'Measuring air quality for advocacy in Africa (MA3): Feasibility and practicality of longitudinal ambient $PM_{2.5}$ measurement using low-cost sensors', *International Journal of Environmental Research and Public Health*, 17(19), p. 7243. Available at: https://doi.org/10.3390/ijerph17197243.

Babatola, S.S. (2018) 'Global burden of diseases attributable to air pollution', *Journal of Public Health in Africa*, 9(3), p. 813. Available at: https://doi.org/10.4081/jphia.2018.813.

Bainomugisha, E., Ssematimba, J. and Okure, D. (2023) 'Design Considerations for a Distributed Low-Cost Air Quality Sensing System for Urban Environments in Low-Resource Settings', *Atmosphere*, 14(2), p. 354.

Budde, M., Schwarz, A., Müller, T., Laquai, B., Streibl, N., Schindler, G., Riedel, T., Dittler, A. and Beigl, M. (2018) 'Potential and Limitations of the Low-Cost SDS011 Particle Sensor for Monitoring Urban Air Quality'. Available at: https://doi.org/10.14644/dust.2018.002.

Dingemanse, J.D. (2022) 'Evaluation of Three Low-Cost Particulate Matter ($PM_{2.5}$) Sensors for Ambient and High Exposure Conditions in Arba Minch, Ethiopia', *Ethiopian Journal of Water Science and Technology*, 4, pp. 33–61. Available at: https://doi.org/10.59122/134080D.

EPA (2006) *General requirements of an equivalent method determination (Subchapter C)*. 40 CFR Parts 53. Available at: https://www.ecfr.gov/current/title-40/chapter-I/subchapter-C/part-53/subpart-C.

Ethiopian Statistics Service (2023) *Population projection: Population Size by Sex Zone and Wereda July 2023*. Addis Ababa. Available at: https://www.statsethiopia.gov.et/wp-content/uploads/2023/08/Population-of-Zones-and-Weredas-Projected-as-of-July-2023.pdf.

European Commission (2010) *Guide to the demonstration of equivalence of ambient air monitoring methods*. European Commission. Available at: https://www.academia.edu/65705447/Guide_to_the_Demonstration_of_Equivalence_of_Ambient_Air_Monitoring_Methods.

Giordano, M.R. and Jaramillo, P. (2021) 'Commentary: AfriqAir's Mission Towards Cleaner Air for Africa and a Call to Action', *Clean Air Journal*, 31(1). Available at: https://doi.org/10.17159/caj/2021/31/1.11173.

Hassani, A., Castell, N., Watne, Å.K. and Schneider, P. (2023) 'Citizen-operated mobile low-cost sensors for urban $PM_{2.5}$ monitoring: field calibration, uncertainty estimation, and application', *Sustainable Cities and Society*, 95, p. 104607. Available at: https://doi.org/10.1016/j.scs.2023.104607.

IET (no date) 'Mettler Analytical Balance AE240 Dual Range Balance Data Sheet'. IET - Used Lab Equipment. Available at: https://www.ietltd.com/pdf_datasheets/Mettler%20AE240%20Data%20Sheet.pdf.

IHME (2021) *Global Burden of Disease Collaborative Network. Global Burden of Disease Study 2019 (GBD 2019)*. Seattle: Institute for Health Metrics and Evaluation (IHME). Available at: https://doi.org/10.6069/1D4Y-YQ37.

Karagulian, F., Barbiere, M., Kotsev, A., Spinelle, L., Gerboles, M., Lagler, F., Redon, N., Crunaire, S. and Borowiak, A. (2019) 'Review of the Performance of Low-Cost Sensors for Air Quality Monitoring', *Atmosphere*, 10(9), p. 506. Available at: https://doi.org/10.3390/atmos10090506.

Malings, C., Tanzer, R., Hauryliuk, A., Saha, P.K., Robinson, A.L., Presto, A.A. and Subramanian, R. (2020) 'Fine particle mass monitoring with low-cost sensors: Corrections and long-term performance evaluation', *Aerosol Science and Technology*, 54(2), pp. 160–174. Available at: https://doi.org/10.1080/02786826.2019.1623863.

Mooney, D., Willis, P. and Stevenson, K. (2006) *A guide for local authorities purchasing air quality monitoring equipment.* AEAT/ENV/R/2088 Issue 2. Oxfordshire, United Kingdom: AEA Technology PLC. Available at: https://uk-air.defra.gov.uk/library/reports?report_id=386.

Ngom, B., Seye, M.R., Diallo, M., Gueye, B. and Drame, M.S. (2018) 'A Hybrid Measurement Kit for Real-time Air Quality Monitoring Across Senegal Cities', in 2018 1st International Conference on Smart Cities and Communities (SCCIC). *2018 1st International Conference on Smart Cities and Communities (SCCIC)*, pp. 1–6. Available at: https://doi.org/10.1109/SCCIC.2018.8584551.

NIOSH (2012) *Components for Evaluation of Direct-Reading Monitors for Gases and Vapor*s. 2021–162. Cincinnati, OH: National Institute for Occupational Safety and Health. Available at: https://stacks.cdc.gov/view/cdc/11956.

Pillarisetti, A., Allen, T., Ruiz-Mercado, I., Edwards, R., Chowdhury, Z., Garland, C., Hill, L.D., Johnson, M., Litton, C.D., Lam, N.L., Pennise, D. and Smith, K.R. (2017) 'Small, Smart, Fast, and Cheap: Microchip-Based Sensors to Estimate Air Pollution Exposures in Rural Households', *Sensors*, 17(8), p. 1879. Available at: https://doi.org/10.3390/s17081879.

Sewor, C., Obeng, A.A. and Amegah, A.K. (2021) 'Commentary: The Ghana Urban Air Quality Project (GHAir): Bridging air pollution data gaps in Ghana', *Clean Air Journal*, 31(1). Available at: https://doi.org/10.17159/caj/2021/31/1.11172.

Shaddick, G., Thomas, M.L., Amini, H., Broday, D., Cohen, A., Frostad, J., Green, A., Gumy, S., Liu, Y., Martin, R.V., Pruss-Ustun, A., Simpson, D., van Donkelaar, A. and Brauer, M. (2018) 'Data integration for the assessment of population exposure to ambient air pollution for Global Burden of Disease assessment', *Environmental Science and Technology*, 52(16), pp. 9069–9078. Available at: https://doi.org/10.1021/acs.est.8b02864.

Sousan, S., Regmi, S. and Park, Y.M. (2021) 'Laboratory Evaluation of Low-Cost Optical Particle Counters for Environmental and Occupational Exposures', *Sensors*, 21(12), p. 4146. Available at: https://doi.org/10.3390/s21124146.

Subramanian, R. and Garland, R. (2021) 'Editorial: The powerful potential of low-cost sensors for air quality research in Africa', *Clean Air Journal*, 31(1). Available at: https://doi.org/10.17159/caj/2021/31/1.11274.

Tefera, W., Kumie, A., Berhane, K., Gilliland, F., Lai, A., Sricharoenvech, P., Samet, J., Patz, J. and Schauer, J.J. (2020) 'Chemical Characterization and Seasonality of Ambient Particles ($PM_{2.5}$) in the City Centre of Addis Ababa', *International Journal of Environmental Research and Public Health*, 17(19), p. 6998. Available at: https://doi.org/10.3390/ijerph17196998.

Volckens, J., Quinn, C., Leith, D., Mehaffy, J., Henry, C.S. and Miller-Lionberg, D. (2017) 'Development and evaluation of an ultrasonic personal aerosol sampler', *Indoor Air*, 27(2), pp. 409–416. Available at: https://doi.org/10.1111/ina.12318.

Wernecke, B., Wright, C.Y., Vande Hey, J., Piketh, S., Burger, R., Kunene, Z., Panchal, R., Millar, D., Oosthuizen, D.N., Batini, C., John, C., Guyatt, A.L., Packer, R.J., Tobin, M.D., Hansell, A., Gulliver, J., Cordell, R.L., Micklesfield, L.K., Ramsay, M., Gayenga, J., Gomez-Olive, F.X., Ngobeni, K., Baloyi, V. and Language, B. (2021) 'Commentary: Opportunities for the application of low-cost sensors in epidemiological studies to advance evidence of air pollution impacts on human health', *Clean Air Journal*, 31(1). Available at: https://doi.org/10.17159/caj/2021/31/1.11219.

World Health Organization (2021) *Ambient (outdoor) air pollution, WHO Newsroom*. Available at: https://www.who.int/news-room/fact-sheets/detail/ambient-(outdoor)-air-quality-and-health (Accessed: 5 September 2022).

World Health Organization (2022) *Household air pollution and health, WHO Newsroom*. Available at: https://www.who.int/news-room/fact-sheets/detail/household-air-pollution-and-health (Accessed: 5 September 2022).

Zamora, M.L., Rice, J. and Koehler, K. (2020) 'One year evaluation of three low-cost $PM_{2.5}$ monitors', *Atmospheric Environment*, 235, p. 117615. Available at: https://doi.org/10.1016/j.atmosenv.2020.117615.

Thank you to our Sponsors

The organising committee would like to extend a heartfelt thank you to the following sponsors for their contributions to making the 2023 National Association for Clean Air Conference a great success:

- Platinum and Gold Sponsors -

- Silver Sponsors -

- Bronze Sponsors -