Quantification and Characterisation of Particulates from Australian Coal Mines: Towards Improved Emissions Estimation

Quantification and Characterisation of Particulates from Australian Coal Mines: Towards Improved Emissions Estimation

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STATEMENT OF ORIGINALITY

This work has not previously been submitted for a degree or diploma at any university. To the best of my knowledge and belief, this thesis contains no material previously published or written by another person, except where due reference is made in the thesis itself.

Signed: 

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Date: 22 July 2019

ABSTRACT

The community health effects associated with exposure to particulate pollution are well documented. Furthermore, the World Health Organisation has identified that no safe limit exists for community exposure to particulate pollution. In this context, policy and regulations associated with particulate exposure continue to reduce the allowable exposure thresholds both in Australia and overseas. This has consequences for industries such as mining, which is currently the primary source of industrial particulate emissions in Australia.

Atmospheric dispersion modelling of particulates is a key tool used to inform environmental policy decisions. The accuracy of dispersion modelling has been found to be critically reliant on the quality of the input data, with the emission data inputs being a key variable. Particulate emission estimation methods currently adopted in Australia are of varying quality and only a limited sub-set of emission estimation methods are soundly based on local empirical data. The use of estimation methods developed for other regions has been shown to introduce significant uncertainty to emission estimation and pollution inventories. In the absence of appropriate local data, a number of the currently adopted particulate emission estimation techniques in Australia rely on methods based on data for other regions such as the United States. In addition, there is an almost complete absence of empirical data relating to PM_{10} and $PM_{2.5}$ emissions from specific mining activities in Australia and overseas, and this a significant gap in our current understanding.

The objective of this research programme is to characterise and define the appropriate emission rates for particulates for a range of activities at open cut coal mines in Australia. This objective was achieved by addressing a series of focus questions as follows:

i. What are the key sources of emissions in Australian open cut coal mines?

ii. What emission rates are currently used to define particulate emissions from mining activities at open cut coal mines, and how appropriate are they?

iii. Is there significant variation in emission rates within mines due to local features of the mine or operations?

iv. What are the appropriate emission rates to adopt for mines in different regions of Australia?

To respond to these focus questions, the research has involved collection of source emission samples from a range of activities at Australian open cut coal mines. Three key approaches have been adopted. Firstly, PM_{10} and $PM_{2.5}$ sampling at the boundary of mines and specific locations downwind of the mine has been completed. The samples were subsequently subject to chemical and visual analysis to identify particulate characteristics. Secondly, sampling of emission rates from erodible surfaces at coal mines has been completed using a portable wind tunnel methodology. Finally, sampling of a range of particulate sources in coal mines has been completed using a downwind sampling transect technique and subsequently applying Gaussian plume calculation techniques to determine the emission rate at the source.

Practical components of the research have been completed in accordance with published peer reviewed methodologies and standard procedures. The guidance of regulatory agencies such as the United States Environmental Protection Agency in relation to developing a high-quality emission dataset has also been considered in developing the methodologies adopted for the field data collection and analysis. A range of quality assurance measures have been adopted. For field work and laboratory analysis these include pre and post calibration of field instrumentation, provision of field and laboratory blanks, temperature and humidity controlled analytical environments and selection of appropriate instrumentation and

methodologies. All data analysis has been subject to a quality assurance review of calculation methodologies and the overall results, to confirm the accuracy and validity of the datasets.

The results of these research activities provide new knowledge in relation to a range of aspects of particulate emissions from Australian coal mines. The new knowledge can be summarised as follows:

- At the boundary of a mine the percentage of $PM_{2.5}$ particulates is close to 50 % lower than in the typical urban environment. However, these particulates have potential to be transported over long distances, hence the mine can remain a significant source of $PM_{2.5}$ particulates in the region.
- The size fractionation of particulate emissions from different regions can vary significantly, hence development of regional emission datasets is necessary.
- New region and activity specific TSP and $PM_{2.5}$ emission rates have been determined for Australian open cut coal mines
- Emission estimation equations have been determined for calculation of TSP emission rates for wind speeds in the range 5 m s^{-1} to 12 m s^{-1}. This will allow estimation of region-specific emission rates based on local meteorological conditions.
- Typical haul route watering rates at Australian mines achieve a control efficiency of 27 %. The water application rate is well below the rate defined in the currently adopted emission estimation methodologies for achieving 50 % particulate emission control.
- The US EPA moisture ratio approach for estimating haul route watering control efficiency is not representative of Australian conditions.
- The research has confirmed the significance of local meteorological conditions for accurate estimation of mining particulate emissions for specific regions.
- Surface crusting reduces particulate emissions from surfaces at wind speeds up to 6.7 m s^{-1}.

Overall, the research makes an original contribution to the current research relating to particulate emissions from open cut coal mines. The research expands on our current understanding of particulate sources in open cut coal mines with respect to size fractions, chemical and physical composition, and emission rates for a range of mining activities. The accuracy and applicability of the currently adopted haul route watering control efficiency calculation has been explored and found to significantly over estimate actual control efficiencies in practice for Australian coal mines.

The research also validates the currently adopted open cut coal mining emissions estimation methods and provides hitherto unavailable empirical data relating to emission rates for particulates from a range of open cut coal mine sources. This provides a sound empirical basis for improving the accuracy of particulate emission estimation techniques for Australian open cut coal mines.

TABLE OF CONTENTS

TABLES

FIGURES

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The Australian Coal Association Research Programme (ACARP) was the primary source of funding for the three research studies completed during the course of this research programme. The financial and practical support offered by ACARP throughout the research is gratefully acknowledged. The support and assistance of the open cut coal mines in Queensland and New South Wales that provided suitable sampling sites and local personnel to assist with access, safety and information about mining operations is also acknowledged. Without access to the mines and support from local mine personnel, this research would not have been possible.

In completing the research presented in this thesis, assistance was provided with the field work elements by mine sites and graduate technical support personnel. Some assistance in terms of data collation, data entry and analysis was also provided, under my supervision and direction.

Technical support acknowledgements are as follows:

- Craig Beyers for assistance with design, manufacture and validation of the wind tunnel, technical support during the field work and data analysis phase of the research programme and described in Chapter 5.
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Finally, the ongoing support and guidance provided throughout the PhD programme by my Joint Principal Supervisors, Professor Igor Agranovski and Dr Shannon Rutherford has been invaluable, and is gratefully acknowledged.

PUBLISHED PAPERS INCLUDED IN THIS THESIS

This thesis includes published papers as described in Chapters 4, 5 and 6, with full copies of the papers presented in Appendices A, B and C. These papers have been co-authored with my supervisors and have been peer reviewed prior to publication in international academic journals. My contribution to each of the published papers is detailed in the introduction to the relevant chapter. The bibliographic details for these publications are as follows:

- Chapter 4 and Appendix A: Characterisation of Particulate Emissions from Australian Open Cut Coal Mines: Towards Improved Emissions Estimates. Richardson C, Rutherford S, Agranovski A. *Journal of Air and Waste Management Association*, 68 (2018), 6, 598 – 607.
- Chapter 5 and Appendix B: Wind Speed Dependent Particulate Emission Rates for Open Surfaces in Open Cut Black Coal Mines. Richardson C, Rutherford S, Agranovski I. *Journal of Environmental Management*, 232 (2019) 537 – 544.
- Chapter 6 and Appendix C: Open Cut Black Coal Mines: Empirical Verification of $PM_{2\,5}$ Emission Estimation Techniques. Richardson C, Rutherford S, Agranovski I. *Journal of Atmospheric Research* 216 (2019) 151 - 159.

Each paper acknowledges those that contributed to the research but did not contribute as authors to the papers.

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ABBREVIATIONS AND DEFINITIONS

Term	Definition
Aerodynamic diameter	The diameter of a sphere, with a density of 1 g cm^{-3} which settles in still air at the same velocity as the particle in question.
Emission rate	Mass of a substance released over a given time period, usually defined as g/s or kg/hour. May also be defined as mass of substance released per unit activity, e.g. kg/tonne processed or kg/km travelled
g/s	Gram per second
Geometric diameter	An equivalent mean particle diameter for irregular shaped particles.
NPI	National Pollutant Inventory
Particulates	A collective name for fine solid or liquid particles added to the atmosphere by processes at the earth's surface. Particulate matter includes dust, smoke, soot, pollen and soil particles.
PM_1	PM_1 – particles which pass through a size-selective inlet with a 50 % efficiency cut-off at 1 µm aerodynamic diameter.
PM_{10}	PM_{10} – particles which pass through a size-selective inlet with a 50 % efficiency cut-off at 10 µm aerodynamic diameter. PM_{10} corresponds to the "thoracic convention" as defined in ISO 7708:1995, Clause 6.
$PM_{2\,5}$	$PM_{2\,5}$ – particles which pass through a size-selective inlet with a 50 % efficiency cut-off at 2.5 µm aerodynamic diameter. $PM_{2\,5}$ corresponds to the "high-risk respirable convention" as defined in ISO 7708:1995, 7.1.
PM	Particulate Matter.
Suspended Particulate Matter	Particulates with insufficient mass to settle out of the atmosphere under the force of gravity.
µg/m^3	Micrograms per cubic metre.

1 INTRODUCTION

1.1 THESIS OVERVIEW

1.1.1 Background

Airborne particulate matter arises from a range of sources, natural and anthropogenic. The relationship between particulates and potential health impacts is well documented (Laden et al, 2000) and most health related criteria and goals are currently based on exposure to a specific mass concentration of particulates for given size fractions. Whilst particulate size and mass concentration are key metrics, the chemical composition of particulates is also an important characteristic in determining potential community health impacts.

Pollution emissions inventories are important tools in guiding regulatory policies and determining the most effective strategies for managing emissions that may result in health impacts (Huertas, Huertas et al, 2012, Weng et al., 2012). Emissions inventories are developed using emission estimation tools, such as monitoring and calculation techniques. The accuracy and suitability of these estimation techniques is fundamental to the development of emissions inventories for use in air quality decision making frameworks.

In Australia, open cut coal mines are currently the most significant national source of industrial particulate emissions. The National Pollutant Inventory (NPI) for Australia (Environment Australia, 2012 & 2015) identifies that approximately 25 % of emissions of particulates from industrial sources arise from coal mining (Environment Australia, 2017). The significance of mining as a source of particulates in Australia is unsurprising, given that mineral exports generated 32 % of total export income for Australia in 2014 (DFAT, 2017). The need for source specific data to assist in the overall management of particulate emissions has been identified (Weng et al, 2012).

The particulate emission techniques adopted in Australia (Environment Australia, 2012b) are based on data from the USA from the 1980's, and data from Australian sampling of specific mine sources completed in the 1990's. Source specific $PM_{2.5}$ sampling data is very limited for open cut coal mines, hence this size fraction is estimated on the basis of assumed particulate size distributions for different activities (US EPA, 1998a). It has been demonstrated that application of the USA emissions estimation methods to other regions may result in significant over estimates of emissions (Chaulya, 2006), and the lack of source specific emissions data for $PM_{2.5}$ is a significant gap in the current understanding of particulate emissions from open cut coal mines.

In broader terms, the published literature has also identified the need for appropriate testing procedures to predict the properties of mineral dust from mining operations and mineral fractionation according to particle size. Patra (2016) reviewed available data relating to emissions and the human health impact of particulate matter from surface mining operations. The need for further research to both determine emission rates of particulate matter (PM) generated due to mining activities, and to characterise the physical and chemical properties of PM to allow the assessment of potential impacts in the surrounding atmosphere and on the health of mine workers in the mines was identified. Patra also identified that the health hazards and exposure to PM deserve to be investigated as thoroughly as for PM generated from mining operations to contribute to safer workplaces and healthier environments at and in the surroundings of surface mines. Ghose (2007) has identified that, despite the detrimental impacts of coal mining emissions and the move to opencast mines, well guided research is required to determine the appropriate emission rates to consider in planning studies for new mines. While Ghose evaluates the available research in the context of mines in India, this work has yet to be completed for Australian mines.

Recently, the National Pollutant Inventory (NPI) emissions estimation methods commonly adopted in Australia have been criticised for inaccuracy and for providing inconsistent data for major sources of emissions (Cooper, Green et al, 2017). In particular, Cooper criticises the current NPI emission estimation method for estimation of particulate releases from unsurfaced roads, identifying the methods as outdated and using inaccurate values and formulas to estimate emissions paved/unpaved roads. Given the significance of surface mines as a source of particulate emissions in Australia, the accuracy of the emissions estimation techniques adopted for proposed mines and for existing mines is of particular importance.

1.1.2 Identified Gaps in Knowledge

As documented in the literature review presented in Chapter 2, the issues associated with the current emissions estimation datasets for open cut coal mining in Australia can be summarised as follows:

- the Australian empirical data relating to TSP emissions from open cut coal mines is over 25 years old. These data are representative of the mine operating techniques and practices at the time. Data representing current operating practices is not currently available;
- there is no empirical data for size fractionated emissions from specific mining activities in Australia or the US to verify the currently adopted particulate emissions estimates for PM_{10} and $PM_{2.5}$;
- the quality of the currently adopted emissions estimation techniques for mining is relatively poor, and inaccuracies and inconsistent emission data has been identified in the Australian National Pollutant Inventory emissions estimates;
- there are no emissions data for coal mines in Queensland, the largest source of particulates in Australia today.

To address these deficiencies in the existing commonly used datasets, a research program was designed to address the key question of:

> *What are the appropriate emission rates to define particulate emissions from sources at open cut coal mines in different locations in Australia?*

1.1.3 Research Programme

The identified gaps in our current knowledge have been addressed through a research programme that provides the empirical data and analysis for addressing the following specific focus questions:

i. What are the key sources of particulate emissions in Australian open cut coal mines?
ii. What emission rates are currently used to define particulate emissions from mining activities at open cut coal mines, and how appropriate are they?
iii. Are there significant variation in emission rates within mines due to local features of the mine or operations?
iv. What are the appropriate emission rates to adopt for mines in different regions of Australia?

The research has involved collection of source emission samples from a range of activities at Australian open cut coal mines.

1.2 THESIS STRUCTURE

The background, results and findings of the research programme are presented in this thesis. A review of the current and historic literature relevant to coal mining particulates is presented in Chapter 2. The methodologies adopted in characterising coal mining emissions are identified and discussed in Chapter 3, and specific details relating to sampling procedures and techniques are presented.

The results of the three research components that form the basis of the overall findings and conclusions of this research programme are presented in Chapters 4, 5 and 6. A research paper has been prepared and published in an international peer reviewed journal for each of these research elements, and these papers are presented in Appendices A, B and C.

The results of the three research elements have been further analysed to address in more detail particulate emission control efficiencies for haul routes in open cut coal mines. The results and findings of this additional analysis are presented in Chapter 7.

Finally, the key findings, overall conclusions and opportunities for additional research are presented in Chapter 8.

2 LITERATURE REVIEW

2.1 HISTORICAL OVERVIEW

Particulates have been a source of health impacts on humans since early civilisation. With the adoption of fire to cook food and provide a source of heat in early human societies, exposure to smoke resulted in a range of health impacts. Evidence from studies of human skeletons from the medieval period indicates that enlarged nasal cavities are most likely attributed to inhalation of domestic and industrial smoke causing sinusitis and other breathing related health complaints (Roberts, 2009).

In 1257 the wife of Henry III, Queen Eleanor, was so affected by smoke from coal fires during a visit to Nottingham that she left the city in search of cleaner air (Luard, 1866). As a result of incidents of this type, particularly among the wealthier members of the population (Brimblecombe, 1976), an early air pollution prevention statute was adopted in London in 1307 seeking to limit the use of sea coal due to concerns about the health and visual effects of smoke emission (Public Record Office, 1307).

During the Middle Ages, numerous observers documented community concerns about smoke from coal fires (Evelyn, 1661) and the connection between health, mortality and smoke was identified and documented (Gaunt, 1662). In late Victorian times, the notorious London smogs heralded a new era of pollution related health impacts. The Smoke Nuisance Abatement (Metropolis) Acts 1853, 1856, and the Public Health (London) Act 1891 sought to control sources of smoke. The on-going smoke related pollution issues and the association with excess deaths in major cities eventually culminated in the Clean Air Acts in 1955 in the United States (Folinsbee, 1992) and in 1956 in the UK (Brimblecombe, 1978).

In the late 20th and early 21st century, the potential health impacts of particulates became an increasing focus of air pollution research (Khafaie, 2016; Schwartz, Slater et al, 1993). The potential relationship between fine particulate exposure and cardio-pulmonary health was documented, and this was the subject of a keynote study based on a series of cohort analyses in major cities in the United States (Dockery, Pope et al, 1993). In 2005 the World Health Organisation (WHO) identified that there was insufficient evidence to suggest a threshold below which no adverse health effects would be anticipated as a result of exposure to particulates (WHO, 2005).

Specific characteristics of particulates – size, morphology and chemical composition – and the relationship of these characteristics with epidemiological outcomes have also been the subject of extensive investigation (Anderson, Thundiyil et al, 2012; Pope, Burnett et al, 2004; Pope & Dockery, 2006). In 2014, the International Agency for Research on Cancer (IARC) identified that particulates in diesel engine exhaust were a Group 1 carcinogen, further supporting the need for more stringent air quality goals for particulates.

More recently, the overall economic cost of air pollution related premature deaths and morbidity has been costed in excess of 1.4 trillion dollars (US) per annum for the European region (WHO, 2013; WHO, 2015). In the context of the latest research and the demonstrated human and economic cost of air pollution, regulatory interest and the interest of the broader community in the issue of particulate exposure and potential health impacts has increased substantially (Bickerstaff & Walker, 2001; Higginbotham, Freeman et al, 2010).

2.2 AIR QUALITY GOALS

Early air pollution legislation focused on management of sources such as smoke. The 1956 Clean Air Act in the UK identified smoke free zones where specific fuel types (coal in particular) were not permitted to

be used for domestic heating purposes (Greater London Authority, 2002; UK DEFRA, 2013). Controls on industrial smoke were also imposed. Monitoring networks were implemented in cities in the UK that provided for sampling of both smoke (using a smoke stain reflectance technique) and sulphur dioxide concentrations (National Physical Laboratory, 2007) and a European wide air quality goal for smoke was adopted (European Union, 1980). As scientific knowledge and monitoring techniques improved, specific methodologies were developed for quantification of deposited particulates and suspended particulates. Legislation was subsequently enacted relating to control of particulates based on mass concentrations in the atmosphere (European Union, 1999).

In the late 1980s, ambient air quality goals for particulates with an aerodynamic diameter of less than 10 microns (PM_{10}) were developed. This was based on research studies that demonstrated the risk associated with smaller particulates penetrating the lungs, and a resultant association with epidemiological effects. Continued research identified that a smaller size fraction, $PM_{2.5}$, posed a more significant human health risk (Ostro, Broadwin et al, 2006). As a result, $PM_{2.5}$ goals and criteria were promulgated in Europe in 2008 (European Union, 2008) and other jurisdictions thereafter.

In Australia, national air quality goals were first promulgated in June 1998 (National Environmental Protection Council, 1998), and these included an air quality goal for PM_{10} particulates. Subsequent amendments to the NEPM Ambient Air Quality provided a guide value for $PM_{2.5}$ (National Environmental Protection Council, 2003) and, in 2015, a national air quality goal for $PM_{2.5}$ was adopted (National Environmental Protection Council, 2015).

Because of the increasing recognition of the health risk associated with PM_{10} and $PM_{2.5}$, compliance with ambient air quality goals has now become a common regulatory requirement. Furthermore, in many cases, compliance goals for suspended particulates are defined in environmental licenses granted to industry, including mining operations (Department of Environment and Conservation (NSW), 2004; Department of Environment and Heritage Protection, 2016; SNL Metals and Mining, 2015). Further research into the dose response relationships associated with exposure to ultrafine particulates, PM_1 and below, is a current focus of interest and has been the subject of numerous research studies (Morawska, Moore et al, 2004). Ultrafine particulates are primarily associated with anthropogenic emissions and combustion processes (Squizzato, Masiol et al, 2016) hence are of lesser significance in the context of mining activities, where the primary source of particulates relates to mechanical processes.

2.3 SOURCES OF PARTICULATES

The particulates that are present in the environment arise from a range of natural and manmade sources. Natural sources include volcanic activity, dust storms, forest and bush fires. Volcanic events can result in releases of large quantities of highly concentrated smoke, that have potential to affect weather conditions throughout the lower atmosphere. An example of this phenomenon is the eruption of the Mount St Helens volcano in 1980. This resulted in an extensive ash plume that cloaked a significant portion of the northern hemisphere for a number of days (Crabtree & Kitchen, 1984; NASA, 1980). Similarly, in Australia, there are regular occurrences of bush fire related particulate concentrations that result in exceedances of the national ambient air quality goals for particulates (Johnston, Hanigan et al, 2011).

Manmade sources of particulate exposure were historically associated with domestic cooking and heating and localized industrial activities (Brimblecombe, 1976). With the advent of industrialization, there has been a significant increase in the number of particulate sources. The wide spread use of motorised transportation has resulted in particulate emissions from this activity becoming one of the most significant sources of particulates in the developed world (Penner, 1996).

Mineral dust is recognized as one of the major components of the global aerosol mix (Radhi, Box et al, 2010). Differences in the size fractionation of particulates from different regions of the world are recognised and, for example, soil derived particulate matter in Australia has been shown to have different physical properties to the coarser grained dust that arises naturally in Africa (Kiefert, McTainsh et al, 1996). This research has further shown that, in undeveloped rural Australia, the size distribution in atmospheric aerosols comprises approximately 50 % $PM_{2.5}$, and around one third PM_1 (with biomass burning the primary source of PM_1). The coarser size distribution indicates that mineral dust may pose a lesser health risk than combustion related particulates, which occur predominantly in the PM_1 fraction. This is because the smaller particulate fractions are likely to penetrate deep into the lungs. As a consequence, the particulates are less likely to be exhaled - and may pass into the blood stream. The WHO has noted that there is some evidence, albeit limited, that exposure to PM_{10} from dust storms is less toxic than particles from combustion sources (WHO, 2000b). This suggests that mechanically derived particulates may pose a lesser health risk than those derived from combustion related sources. This conclusion is further supported by more recent research that identifies the significance of the mutagenic and cytotoxic effects of ultrafine particles due to combustion related chemical composition (Landkocz, Ledoux et al, 2017). Nevertheless, emissions of relatively inert mineral particulates generated from crustal material by mechanical processes remains a significant issue, due to the quantities of material arising from industrial processes such as mining.

2.4 PARTICULATE CHARACTERISTICS

The diverse characteristics exhibited by particulates, in terms of size, morphology and chemical composition, introduces challenges in determining the causal relationship between public health risk and particulate toxicity.

Airborne particulate matter in the environment arises from a range of sources, natural and anthropogenic. Typically, combustion related particulates are the primary source in developed areas and arise from stationary combustion sources (e.g., power stations and domestic space heating) and mobile (vehicle) emissions (Laden, Neas et al, 2000). In less developed regions, the primary sources of particulate emissions are associated with natural sources which include breakdown of crustal material, sea spray and biomass burning (Sorek-Hamer, Broday et al, 2017). With the exception of biomass burning in rural areas, emission from natural sources such as breakdown of crustal materials most commonly fall in the PM coarse size fraction of $PM_{2.5-10}$ ("<123_EISDoc_Mine Air Quality Report.pdf>," ; Deshmukh, Deb et al, 2012; Kaufman, Tanré et al, 2002; Senate Community Affairs Committee Secretariat, 2013)

There are a number of particulate characteristics that are significant from a health impact perspective. The size of atmospheric particulates is one of the key metrics in determining the potential health impact of exposure to particulates (Senate Community Affairs Committee Secretariat, 2013). The United States Environmental Protection Agency (US EPA), in 1996, summarized the size classifications of significance for particulates (US EPA, 1996). Based on this schema, the largest particles, > 30 µm, settle out of the atmosphere under the force of gravity and do not remain suspended in the atmosphere. Particles with a mean diameter of < 30 µm will tend to remain suspended in the atmosphere for extended periods of time. The exception to this is where particles are removed from the atmosphere due to phenomenon such as rainfall, impingement on surfaces or agglomeration that results in increased mass (hence are removed via gravity). The coarse size fraction is generally produced by mechanical breakup of larger solid particles, for example agriculture, wind-blown dust and vehicles travelling on unpaved roads. Insect parts and pollen are also classified in the coarse size range, and salt from sea spray is fairly coarse in nature. Typically, there is a lower size limit of 1 µm for coarse particulates, below which further break up by mechanical means is limited (WHO (2000b).

Suspended particles can be inhaled into the human respiratory system. The deposition of particles in the human respiratory tract has been documented for over a century, and the rate of deposition depends upon the size, shape and density of the particles (Heyder, 1986). The coarser suspended particulates (>10 µm) will generally be captured in the nasal passages.

The relationship between particle size and potential health impacts has been documented since the latter part of the 19th Century (Heyder, 1986). Given the wealth of research relating to this causal relationship, most health-related criteria and goals are currently based on exposure to a specific mass density of particulates for given size fractions.

Whilst particulate size and mass concentration are key metrics, there are a range of additional characteristics that are relevant from a health exposure perspective. The chemical composition of particulates is an important characteristic. Where particulates are comprised of a substance that, in itself, is a known health risk, then the potential health impacts may be more significant than the simple mass concentration/size fraction relationship would indicate. This is because fine particulates, $PM_{2.5}$ and smaller, which penetrate deep into the alveolar system, can act as a carrier of other substances that may pass into the blood stream (Landkocz, Ledoux et al, 2017). Diesel particulates are a specific known risk, and the World Health Organisation has classified diesel particulates as a possible Class II carcinogen (IARC, 2014; Radhi, Box et al, 2010). Similarly, some forms of silica are also known carcinogens (IARC, 1997; Radhi, Box et al, 2011; WHO, 2000a), hence lower particulate exposure thresholds are adopted where a composition specific risk has been identified. Combustion derived particulates have a specific chemical signature, hence air quality goals for PAH and other components such as organic carbon may be adopted (Squizzato, Masiol et al, 2016).

In some cases, the shape or morphology of particulates is related to specific health outcomes. Exposure to fibers, such as asbestos, can result in cancer and this is related to the fiber acting as an irritant as opposed to the chemical composition of the particulates causing the adverse response. Specific exposure thresholds to a range of types of fibres, including asbestos and glass fibres, have been defined for the occupational environment (Safe Work Australia, 2013), and may also be relevant to the general public.

Whilst mass concentration is a well-established metric for assessing the health impact of fine and coarse particulates (> $PM_{2.5}$), in the realm of ultrafine and nano particles alternate approaches may be relevant. Whilst there is a growing body of research relating to the use of particle number for quantification of the health risk of ultrafine particulates (Kumar, Morawska et al, 2014), exposure thresholds have yet to be formally adopted based on particulate count (Sioutas, Delfino et al, 2005). More recent research has identified that the surface area of spherical nanoparticles could be the most relevant dose-response metric for assessing the pulmonary health impacts of fine particulate matter (Schmid & Stoeger, 2016). However, a recent review of the possible causal relationships has concluded that there is currently no compelling evidence to identify specific health risk metrics for ultra fines from a public health risk perspective, relative to the metrics that are currently adopted for particulate exposure as a whole (Heal, Kumar et al, 2012).

Overall, the current epidemiological evidence indicates that mass concentration exposure thresholds correlate well with a range of adverse health outcomes. These can be further sub-divided based on particle size and chemical composition where dose response relationships have been established. On this basis, the adoption of mass concentration exposure criteria is likely to continue for the near future, until such time as additional evidence is available to support alternate approaches based on different measurement metrics (National Environmental Protection Council, 2011). This in turn is relevant to the development of emission inventories and estimation techniques. The descriptors for the emission source must be consistent with the adopted benchmarks or policy goals, if these data are to be used in analysis that provides a basis for policy decisions.

2.5 KEY INDUSTRIAL SOURCES OF PARTICULATES

Pollution emissions inventories are important tools in guiding regulatory policies, and for determining the most effective strategies for managing emissions across industry sectors. The use of pollutant inventories as a means of informing and successfully addressing pollution challenges has been recognised (Huertas, Camacho et al, 2012; Weng, Mudd et al, 2012).

In the developed world, a number of national governments and regional jurisdictions have introduced legislation requiring the estimation and reporting of pollution emissions to air. These include the United States, Europe and Australia. Review of national and regional inventories provides an insight into the significance of particulate emissions, and an indication of the significance of sources of particulate pollution in the developed world.

The National Pollutant Inventory for Australia identifies the total annual emissions of particulates from industrial sources from 2010 – 2015 as shown in Table 2-1. These data demonstrate that, on a national basis, $PM_{2.5}$ mass emissions are estimated to constitute less than 4% of PM_{10}.

Table 2-1: Total Estimated Particulate Emissions, Australia (tonnes)

Year	Total PM_{10}	Total $PM_{2.5}$
2010/2011	1,300,000	31,000
2011/2012	1,400,000	34,000
2012/2013	1,500,000	32,000
2013/2014	1,600,000	32,000
2014/2015	1,600,000	31,000

Source: (Environment Australia, 2017)

Table 2-2 provides a breakdown of the industrial emissions of PM_{10} particulates by industry type, for the reporting year 2014/2015, for the industry groups estimated as emitting 10,000 tonnes or more. Table 2-3 presents the same information for $PM_{2.5}$, for industry groups with estimated emissions in excess of 1,000 tonnes per year. The data presented in Table 2-2 demonstrates that, for PM_{10}, mining emissions are 53 % of overall emissions, and coal mining alone accounts for 25 % of all estimated emissions. The pattern of emissions is somewhat different for $PM_{2.5}$, with combustion related sources more significant than mining sources.

The data in Table 2-3 confirm that electricity generation is estimated to produce the highest overall emissions of $PM_{2.5}$ for the 2014/15 reporting year. Coal mining was the second most significant source and was estimated to result in 24 % of all $PM_{2.5}$ emissions, with metal ore and coal mining combined responsible for 43 % of emissions.

Table 2-2: Estimated PM_{10} Particulate Emissions, All Sources, Australia 2014/2015 (tonnes)

Industry Group	Estimated Emissions of PM_{10}
Metal ore mining	450,000
Coal mining	400,000
Burning/wildfires (natural)	240,000
Windblown dust	190,000
Paved/unpaved roads	160,000
Electricity generation	24,000
Solid fuel burning (domestic)	20,000
Motor vehicles	12,000
Other	104,000
Total (All Sources)	1,600,000

Source: (Environment Australia, 2017)

Table 2-3: Estimated $PM_{2\,5}$ Particulate Emissions, by Industry, Australia 2014/2015 (tonnes)

Industry Group	Estimated Emissions of $PM_{2\,5}$
Electricity Generation	8,900
Coal mining	7,300
Metal ore mining	5,900
Sugar and confectionary manufacturing	2,900
Basic non-ferrous metal manufacture	1,200
Other	4,800
Total	31,000

Source: (Environment Australia, 2017)

Table 2-4 and Table 2-5 present a breakdown of the 10 Australian facilities reporting the highest emissions of PM_{10} and $PM_{2\,5}$ particulates for the 2014/2015 year. The % contribution of each facility to the overall annual PM_{10} emissions of 1,600,000 tonnes and $PM_{2\,5}$ emissions of 31,000 tonnes in 2014/15 is also identified. Queensland coal mines (highlighted in bold) feature as top 10 emitters for both PM_{10} and $PM_{2\,5}$, hence are key sources of emissions. Coal mines in other parts of Australia, such as the Hunter Valley, are not identified in the top 10 emission sources.

Table 2-4: Estimated PM_{10} Particulate Emissions by Facility, Australia, 2014/2015 (tonnes)

Facility Name, Location	Mine Type	PM_{10}	% of Total Annual Emissions
Wheelarra Hill, Jimblebar, Newman, Western Australia	Iron	43,702	2.7
Mining Area C, Newman, Western Australia	Iron	29,622	1.9
Peak Downs Mine, Moranbah, Queensland	**Coal**	**28,584**	**1.8**
Blackwater Mine, Blackwater, Queensland	**Coal**	**26,989**	**1.7**
Goonyella Riverside Broadmeadow Mine, Moranbah, Queensland	**Coal**	**21,634**	**1.4**
Christmas Creek Operations, Newman, Western Australia	Iron	20,265	1.3
Saraji Mine, Dysart, Queensland	**Coal**	**19,696**	**1.2**
Solomon Operations, Tom Price, Western Australia	Iron	19,658	1.2
Dawson, Moura, Queensland	**Coal**	**18,014**	**1.1**
Cloudbreak Operations, Mulga Downs, Western Australia	Iron	14,643	0.9

Source: (Environment Australia, 2017)

Table 2-5: Estimated $PM_{2\,5}$ Particulate Emissions by Facility, Australia, 2014/2015 (tonnes)

Facility Name, Location	$PM_{2\,5}$	% of Total Annual Emissions
Energy Australia, Yallourn, Victoria	1,951.33	6.3
GDF SUEZ, Hazelwood, Morwell, Victoria	877.95	2.8
Loy Yang B Power Station, Traralgon, Victoria	777.71	2.5
AGL Loy Yang, Traralgon, Victoria	592.05	1.9
The Maryborough Sugar Factory Ltd, Maryborough, Queensland	547.00	1.8
Tarong Power Station, Nanango, Queensland	498.23	1.6
Blackwater Mine, Blackwater, Queensland (Coal)	**479.84**	**1.5**
Eraring Power Station, Eraring, New South Wales	442.86	1.4
BSL, Gladstone, Queensland	429.00	1.4
Peak Downs Mine, Moranbah, Queensland (Coal)	**418.11**	**1.3**
Saraji Mine, Dysart, Queensland (Coal)	**402.93**	**1.3**

Source: (Environment Australia, 2017)

In terms of global contributions of mining source, in 2014 China, the United States, India, Australia and Indonesia were ranked as the top five coal producing countries in the world (Mining Technology, 2017). Of these countries, a national pollutant emission inventory database is only available for the United States and Australia. China, India and Indonesia do not currently compile these data in a systematic way, hence comparison with emissions inventory data for Australia is problematic.

PM_{10} and $PM_{2\,5}$ emissions estimates for the United States from 2010 to 2016 are presented in Table 2-6 and Table 2-7 (US EPA, 2016b). Mining activity is not identified as an individual source category by the US EPA. Mining related emissions are included in the 'Other industrial processes' source category. Review of the emissions estimates for the 'Other' sources indicates estimated PM_{10} emissions of 777 Mtonnes in 2016, equivalent to just 3 % of overall estimated PM_{10} emissions, and 4 % for $PM_{2\,5}$. The 'Other industrial processes' category includes a range of sources in addition to mining, hence this represents the maximum estimated mining contribution to particulate emissions overall in the United States. This highlights that that the US emissions estimates are more than ten times lower than PM_{10} mining emissions estimated for Australia (53 % of total), as a proportion of overall emissions.

Table 2-6: United States Emissions Estimates, PM_{10} (Mtonnes)

Source Category	**2010**	**2011**	**2012**	**2013**	**2014**	**2015**	**2016**
Fuel combustion - Electricity	323	280	263	246	229	229	229
Fuel combustion - Industrial	246	275	291	308	325	325	325
Fuel combustion - Other	412	424	405	387	368	368	368
Chemical & assoc. product manuf.	23	22	21	21	20	20	20
Metals processing	69	63	62	60	58	58	58
Petroleum & related industries	33	35	35	36	36	36	36
Other industrial processes	870	766	770	773	777	777	777
Solvent utilisation	4	4	4	4	4	4	4
Storage and transport	52	52	57	62	67	67	67
Waste disposal and recycling	207	192	230	268	305	305	305
Highway vehicles	279	371	351	330	310	298	287
Off highway	228	223	216	208	200	181	163
Miscellaneous	18,076	18,015	19,369	20,723	21,803	21,803	21,803
Wildfires	1,178	1,326	1,326	1,326	1,030	1,030	1,030
Total	20,823	20,723	22,074	23,425	24,502	24,472	24,442

Source: US EPA 2016

Table 2-7: United States Emissions Estimates, $PM_{2\,5}$ (Mtonnes)

Source Category	2010	2011	2012	2013	2014	2015	2016
Fuel combustion – Electricity	240	205	196	187	177	177	177
Fuel combustion - Industrial	197	223	229	235	240	240	240
Fuel combustion – Other	403	415	397	380	363	363	363
Chemical & assoc. product manuf.	18	17	16	15	14	14	14
Metals processing	53	48	47	46	45	45	45
Petroleum & related industries	27	29	30	31	32	32	32
Other industrial processes	298	277	287	297	307	307	307
Solvent utilisation	4	4	4	4	4	4	4
Storage and transport	22	21	20	19	18	18	18
Waste disposal and recycling	178	165	194	223	253	253	253
Highway vehicles	199	198	185	172	159	146	133
Off highway	213	210	201	191	182	166	151
Miscellaneous	4,111	4,288	4,416	4,545	4,425	4,425	4,425
Wildfires	999	1,125	1,125	1,125	873	873	873
Total	6,963	7,225	7,348	7,470	7,093	7,064	7,035

Source: US EPA 2016

The significance of mining as a source of particulates in Australia is unsurprising, given that mineral exports generated 32 % of total export income for Australia in 2014 (DFAT, 2017). For this reason, the need for source specific data to assist in the overall management of these sources of emissions has been identified (Weng et al, 2012).

Based on forward projections, the future significance of coal mining in Australia is likely to continue. The Australian Bureau for Agricultural and Resource Economics and Science (ABARES) estimates for future energy production in Australia (Syed, 2010) are summarised in Table 2-8. This confirms that 2.4 % annual growth in black coal production is expected in the period from 2007/8 to 2029/30.

Table 2-8: Estimated Growth in Energy Production (Syed, 2010)

Energy Source	2007-08 (Petajoules)	2029-30 (Petajoules)	Average Annual Growth Rate (%) 2007-08 to 2029-30
Black coal	8,696	13,423	2.0
Brown Coal	610	452	-1.4
Oil	945	425	-3.6
LPG	103	243	4.0
LNG	2,040	8,505	6.7

Source: Syed, 2010

2.6 AUSTRALIAN OPEN CUT COAL MINES AS A SOURCE OF PARTICULATES

Mining involves extraction of minerals from the ground. Where reserves are located close to the surface, extraction using above ground 'open cut' techniques is practicable. This involves removing soil and other materials, or 'over burden', that do not contain the resource of interest in order to expose the material of interest. Where deeper deposits occur, extraction involves underground mining. Surface mining has many advantages, including recovery of a higher proportion of the resource (85 % +) compared to 40 – 70 % for underground mines in the case of coal (Committee on Coal Research Technology and Resource Assessments, 2007).

The initial preparation of a coal mine for operation involves removal of topsoil and subsoil with large scrapers. The topsoil is carried by the scrapers to cover a previously mined and regraded area as part of the reclamation process or is placed in temporary stockpiles. The exposed overburden, the earth that is between the topsoil and the coal seam, is leveled, drilled, and blasted. Then the overburden material is removed down to the coal seam, usually by a dragline or a shovel and truck operation. The uncovered coal seam is then drilled and blasted. A shovel or front-end loader loads the broken coal into haul trucks, and it is taken out of the pit along graded haul roads to the tipple, or truck dump. Raw coal may be dumped onto a temporary storage pile and later re-handled by a front-end loader or bulldozer.

At the raw coal stockpile, coal is dumped into a hopper that feeds the primary crusher, then is conveyed through additional coal preparation equipment such as secondary crushers, screens and coal wash plant prior to depositing in a coal product storage area. If the mine has open storage piles, the crushed coal passes through a coal stacker onto the pile. The piles, usually worked by bulldozers, may be subject to wind erosion.

During mine reclamation, which typically occurs throughout the life of the mine, overburden spoil piles are smoothed and contoured by bulldozers. Topsoil is placed on the graded spoils, and the land is prepared for revegetation by furrowing, mulching, etc. From the time an area is disturbed until the new vegetation emerges, all disturbed areas are subject to wind erosion.

Open cut mines emit more particulate emissions to the ambient air than an underground mine. This is because the workings are open to the environment, as opposed to enclosed with ventilation extracts. However, as open cut mines provide for recovery of a significantly higher proportion of the resource, are safer for workers and are more cost effective, the proportion of resources mined using this method has increased over time (Mudd, 2007). The proportion of overburden to mineral resource extracted varies for different reserves and is typically in the range 1:1 to 6:1 overburden to deposit (Mudd, 2009). The data presented by Mudd show that over time the over burden ratio tends to increase. This is because the more economic reserves, which generally have lower proportions of overburden, tend to be extracted first. This trend is likely to increase in the future. This confirms that overburden material is the most significant source of particulate emissions at open cut mines, rather than the material being mined, due to the ratio of waste to resource that is extracted.

Because of the dominance of overburden as a source of mining related particulates, the local geological characteristics are a key factor in determining the nature and composition of mechanically generated particulate emissions from mining. Soil particles have a range of sizes from grit and pebbles to fine clays that may have a particle diameter of less than 1 micron. Soil particles of less than 2 mm are generally divided into three major size groups: sand, silt and clay. Table 2-9 shows a standard size classification. Particle size analysis is used in soil science to evaluate soil texture and allows determination of the particle size distribution for a given soil type.

Table 2-9: Soil Particle Sizes (Standards Australia, 1993)

Soil Particle Sizes	
Term	**Size Range**
Boulders	> 200 mm
Cobbles	63 – 200 mm
Coarse Gravel	20 – 63 mm
Medium Gravel	6 – 20 mm
Fine Gravel	2.36 – 6 mm
Coarse Sand	0.6 – 2.36 mm
Medium Sand	200 – 600 µm
Fine Sand	75 – 200 µm
Silt	2 – 75 µm
Clay	<2 µm

Particle size distributions are of relevance when estimating emissions from activities that process soils or soil type materials, and where vehicles and machinery are used on unmade roads. This is particularly relevant to open cut coal mining, as the majority of potential sources of PM_{10} and $PM_{2.5}$ emissions involve soil extraction and movement of overburden on unsurfaced haul roads. The types of material associated with different operations within a coal mine are shown in Table 2-10.

Table 2-10: Coal Mining Operation Emission Factors and Materials (US EPA, 1995b)

Operation	**Material**
Blasting	Coal or overburden
Truck loading	Coal
Bulldozing	Coal or overburden
Dragline	Overburden
Vehicle traffic	Road surface material
Grading	Road surface material
Wind erosion of stockpiles	Coal

Table 2-10 identifies that many operations that generate particulates at a coal mine are not coal related. For the remainder, particulate emissions relate to processing of soil and rock material (over burden) or movement on open haul roads. For these sources the silt fraction of the material or surface is a key determinant in the emission estimation equation (US EPA, 1995b).

2.7 ASSESSMENT OF THE ENVIRONMENTAL RISK OF PARTICULATE RELEASES FROM SURFACE MINING

The broader health issues associated with environmental exposure to particulate matter were discussed in Sections 2.1 to 2.4. The specific impacts associated with mining related emissions have also been documented, particularly in an occupational exposure context. Whilst the particulate size and composition of emissions from mining have been identified as posing a lesser risk to the community than combustion related particulates (Senate Community Affairs Committee Secretariat, 2013; WHO, 2004), a broader correlation between community health effects and exposure to mining particulates has been identified (Hendryx, 2015). In this context, Hendryx argues that prudent steps should be taken to protect public health impacts from mining through more effective regulatory control.

Determining the environmental risk of particulates from surface mining has been considered in an extensive body of research in the published literature (Department of Environment and Conservation (NSW), 2004; Ghose, 2007a; Hendryx, 2015), and is also routinely considered in environmental assessment studies completed for new mines and proposed expansions to existing mines. Pollution emissions inventories are important tools in guiding regulatory policies, and for determining the most effective strategies for managing emissions across industry sectors. Having established the actual or predicted impact on the atmospheric environment of a specific source or project, a mitigation strategy can be developed (Ghose, 2007a).

In order to determine the potential environmental risk of surface mining releases of particulates the following steps are necessary:

i. Quantify the rate of release of particulate emissions,

ii. Characterise the particulates being released, and

iii. Determine the transport and environmental fate of these emissions.

Quantification of the rate of release of particulates can be completed by direct measurement or by emission estimation techniques. Due to the cost associated with direct measurement techniques, emission estimation techniques are generally adopted. Environmental monitoring is commonly completed to determine the off-site impacts of mining particulates; however these types of study do not permit identification of emission rates for specific sources at a surface mine. For this reason, environmental monitoring of particulates is of limited value for the establishment of emission data. Monitoring studies can be used for validating the results of atmospheric modelling. This is particularly useful where an existing mine is applying for an approval for expansion, as the modelling can be benchmarked for the existing operations using the environmental modelling data, and then adjusted to account for the additional emissions expected to be associated with the new sources of particulates.

Clearly, adoption of representative particulate emissions data is fundamental to accurate environmental risk assessment and provides the foundation for analysis of the environmental fate of these releases. The availability of suitable emission data to characterise emissions from surface coal mines is discussed in more detail in Chapter 2.8.

Having identified the appropriate emissions data, modelling approaches are then generally adopted to determine the predicted environmental risk (Ghose, 2007b; Huertas, Huertas et al, 2014; Huertas, Huertas, & Díaz, 2012). Atmospheric dispersion modelling of the estimated emissions is the approach most commonly adopted to allow prediction of expected concentrations of particulates external to the surface mine. Dispersion modelling may also provide information about concentrations within the surface mine, to inform occupational exposure risk assessment.

Atmospheric dispersion modelling techniques are well developed and Gaussian, Puff and Lagrangian models approved by the United States Environmental Protection agency are commonly adopted for the prediction of particulate emissions (Huertas, Huertas et al, 2014). Assuming that state-of-the-art air quality models adequately describe plume dispersion, then the precision and accuracy of their results directly depend on the suitability of their geographical, meteorological and emission input data. A number of researchers have commented on the significance of emissions data inputs to atmospheric dispersion modelling as a key source of error or modelling uncertainty. In an analysis of the performance of the atmospheric dispersion models AERMOD and CALPUFF to an open pit quarry situation, Tartakovsky identified that due to severe uncertainties in the model inputs, it is necessary to complete extensive model iterations when assessing model performance (Tartakovsky, Broday et al, 2013).

Further, Holnicki & Nahorski (2015) have identified that, when basing policy decisions on atmospheric dispersion modelling predictions, it is critical that the uncertainty associated with the methodology is understood. In particular, air pollution emission data from industry, traffic and municipal sources was identified as a primary cause of modelling uncertainty in urban areas, with a potential for negative environmental and health consequences. With reference to the opencast mining industry, Chakraborty demonstrated that the average accuracy between measured and calculated emission rates varies significantly – ranging from 77 to 80 % (Chakraborty, Ahmad et al, 2002). Specific issues associated with the imprecision of emissions inventories used for atmospheric dispersion modelling have been identified (Holnicki & Nahorski, 2015). The paucity of emissions concentration data for accurate source characterisation to allow modelling of downwind hazards was identified by Beiringer et al in 2017 (Bieringer, Young et al, 2017), and this is a significant gap in our current knowledge.

2.8 COAL MINE PARTICULATES EMISSION ESTIMATION METHODS

2.8.1 Currently Adopted Methods

In Australia, the National Pollutant Inventory Guide (Environment Australia, 2015) identifies the following methods for determining emissions for the purposes of meeting national emission reporting requirements:

- mass balance calculations;
- engineering calculations;
- sampling or direct measurements;
- emission factors; and
- alternative (approved) techniques.

Of these, emission factors are generally adopted where insufficient data is available to allow the application of mass balance or engineering calculations, or where direct sampling methods for a specific site are not practicable. Environment Australia publishes emission estimations manuals for a range of industries and these are used in calculating estimated emissions to atmosphere, land and water for a range of reportable substances. These reportable substances include TSP and PM_{10} from mining. Environment Australia publishes emission estimation methods for coal mining in the Australian National Pollutant Inventory (NPI) Emission Estimation Manual for Mining (Environment Australia, 2012b). The NPI manual for Mining is the recommended approach for calculation of overall particulate emissions for the National Pollutant Inventory data and is the basis for the data presented earlier in Table 2-1 to Table 2-5.

Particulate emission estimation is also an important input to dispersion modelling studies prepared for proposed new and expanded mines. Review of Environmental Impact Studies (EIS) prepared for proposed

coal mines in Australia provides an indication of the emission estimation approaches currently adopted. A summary of the sources of data adopted for the estimation of particulate emissions rates of particulates for Environmental Impact Studies (EIS) prepared for surface mining operations in New South Wales, Australia for the period 2014 – 2016 is presented in Table 2-11.

Table 2-11 indicates that the NPI emission estimation manual is identified by a number of authors as one of the methods adopted for estimation of particulate emissions. The Australian NPI particulate emission estimation techniques rely on data from a study completed in the Hunter Valley in 1988 (NERDCC, 1988), supplemented by data from the US EPA AP42 (US EPA, 1995b) inventory where Australian derived emissions data are not available. The US EPA AP42 data in Section 11.9.1 – Western Surface Coal Mining is based on research completed in the United States from 1978 – 1994. The last full edition of AP42 was published in 1995 and the document is commonly cited as the 1995 edition, despite subsequent amendments. The additional emission estimation methods that are referenced in Table 2-11 are based on Australian empirical studies that were subsequently adopted in the 2012 NPI emission manual.

Table 2-11: Sources of Particulate Emission Factors Used in NSW Open Cut Coal Mining EIS

Date	Author	Mine	Source of Particulate Emission Factors
12/12/2012	PAE Holmes	Watermark Coal	NPI 2012, SPCC 1983, US EPA AP42 1985
30/11/2012	PAE Holmes	Wallarah 2 Coal	NPI 2012, NERDDC 1988, US EPA AP42 1995
26/2/13	PAE Holmes	Chain Valley Colliery	NERDDC 1988, US EPA AP42 1985 & 1995
19/5/13	PEL	Bulga Coal	SPCC 1986, US EPA AP42 1985 & 2006
12/7/13	Todoroski Air Sciences	Bengalla Mine	NPI 2012, SPCC 1983, US EPA AP42 1984
29/7/13	PEL	Western Coal Services	SPCC 1986, US EPA AP42 1985
20/3/14	SLR	Northern Coal	NPI 2012, MRI 2006, US EPA AP42
12/6/14	Todoroski Air Sciences	Warkworth Mine	NPI 2012, SPCC 1983 & 1986, US EPA AP42 1985
12/6/14	Todoroski Air Sciences	Mount Thorley	NPI 2012, SPCC 1983 & 1986, US EPA AP42 1985
29/10/14	PEL	Mount Owen	NERDCC 1988, US EPA AP42 1985
1/4/15	PEL	Drayton South	US EPA 1985, SPCC 1986
1/7/15	PEL	Bylong Coal	US EPA AP42 1995
20/11/15	Todoroski Air Sciences	Wilpinjong	US EPA AP42 1985, SPCC 1983 & 1986, NPI 2012

To provide an indication of the differences between the emission factors adopted in the NPI 2012 manual as compared to the original Hunter Valley research (NERDCC, 1988) and the US EPA AP42 data, the total suspended particulate emission rates cited in these documents are presented in Table 2-12. The NPI emission factors that are based on measurements completed in the Hunter Valley are highlighted in bold

font. The emission factors relate to non-controlled emissions (i.e., without controls such as watering) except where identified.

Review of Table 2-12 confirms that few of the NPI emission rates are, in fact, based on data collected in Australia. A total of 5 of the default emission factors (highlighted in bold font) are based on the research completed in the Hunter Valley in the 1980s. A number of the NPI default emission factors are based on the US EPA AP42 emissions equations, with typical Hunter Valley (NSW) silt content and moisture contents applied to provide a correction for Australian conditions. The use of local data is expected to provide an acceptable description of emissions at the Hunter Valley mines represented by these corrections. However, the US EPA AP42 identifies that there is significant variability between, and in some cases within, mines due to the changes in soil and overburden particle size distributions. As the corrections adopted in the NPI are based on data for the Hunter Valley only, these corrections are not necessarily applicable to other regions of Australia based on the guidance in AP42. For example, the US EPA AP42 emission factors for drilling are 0.1 kg/hole for drilling of coal in a loamy/sandy/clay/clay loamy soil, and a default factor of 0.59 kg/hole for drilling of overburden. The NPI emissions estimation manual identifies the appropriate emission factor for Australian mines as 0.1 kg/hole, despite the wide variability demonstrated in the US dataset. No correction for local conditions is provided in the US EPA AP42 emission estimation for drilling of blast holes and none has been applied in the current NPI emission estimation method for this activity. Overall, this example indicates that the emissions for drilling in Australian mines could significantly underestimate particulate emissions. However, no local data is currently available to verify the suitability of these emissions.

A number of studies have investigated the applicability of the US EPA AP42 (US EPA 1995) emissions estimation methods to regions outside the United States. Adoption of the US EPA emission rates in India has been identified as inappropriate, due to the differences in mining site practices, geological and climatic conditions (Chaula et al 2001). Even where the emissions estimation methods are applied in mines in the country for which they were developed, the uncertainties can be large. Application of the AP42 equations to various mines in the US was found to result in underestimation by a factor of up to 13, through to overestimation by a factor of 1.5 (Huertas, Huertas and Diaz, 2012). Therefore, it is concluded that the current NPI emission estimation factors for mining particulates may not be applicable to all regions in Australia and should be used with caution as the majority are based on data for mines in the US. In addition to issues related to applicability of the default NPI emission equations to other regions, the stated emission rates for truck loading and shoveling in the NPI 2012 edition also appear to be erroneously based on controlled, rather than uncontrolled, emission rates. Based on review of the NERDDC report, the NPI emission rate for truck loading and shoveling are equivalent to the controlled rates reported in the research, despite the fact they are adopted in the NPI as uncontrolled emission rates. This may lead to underestimation of emissions from these sources, where the emission equations from the current edition of the NPI manual are adopted. This indicates that there may be errors in the currently adopted Australian emission factors for particulate emissions from mining.

Table 2-12: Comparison of NPI, Hunter Valley(NERDCC) and US Mining Default TSP Emission Rates

Activity	NPI 2012	NERDCC 1986	US EPA AP42 1995
Truck and shovel loading of coal (kg/t)	**0.029**	0.029[a]	0.007[b]
Truck and shovel loading of overburden (kg/t)	**0.025**	0.025[a]	0.018
Dragline handling overburden (kg/m^3)	**0.060**	0.039[a] – 0.072	0.010[c]
Coal and overburden transport by off-highway trucks (kg/veh/km)	4.23	1.8[a] – 3.5	4.23[d]
Coal transport by off-highway trucks (kg/veh/km)	4.23	2.0[a] – 3.2	4.23[d]
Overburden transport by off-highway trucks (kg/veh/km)	4.23	1.6[a] – 2.8	4.23[d]
Trucks dumping overburden (kg/t)	**0.012**	0.012[a] – 0.02	0.001
Trucks dumping coal into ROM hopper (kg/t)	0.010	0.01[a] – 0.02	0.033
Bulldozers on coal kg/hr/veh	102	-	32.5[e]
Bulldozers on material other than coal kg/hr/veh	17	-	4.1[e]
Drilling kg/hole	0.1	-	0.1 - 0.59[f]
Blasting kg/blast	0.59	-	3.24[g]
Scrapers – travel mode kg/veh/km	2.08	-	2.08[h]
Scrapers – removing topsoil kg/veh/km	0.029	-	0.029 - 0.22[i]
Graders kg/veh/km	0.19	-	0.19
Loading Stockpiles kg/t	0.004	-	0.004 – 0.02[j]
Unloading from stockpiles kg/t	0.03	0.06[k]	n/a
Loading to trains kg/t	0.0004	-	0.014 – 0.018
Miscellaneous transfers kg/t/transfer point	0.00032	-	0.00032[k]
Wind erosion kg/ha/hr	**0.4**[l]	n/a	0.85[m]

[a] Normal dust control measures applied, based on typical controls adopted in the Hunter Valley.
[b] Assuming coal with a moisture content of 8 %.
[c] Assuming a 12 m drop height and 2 % moisture content.
[d] From AP 42 S13.2.2 (2006) wheel generated dust from unpaved roads, silt content of 10 %, vehicle gross mass of 48 t, k value of 1.38.
[e] Silt content 7 %, moisture content 2 %.
[f] 0.1 = coal, loamy/sandy/clayey/clay loamy soil. 0.59 = overburden.
[g] 930 m^2 area blasted, 2 % moisture content.
[h] 10 % silt content, gross vehicle mass of 48 t.
[i] 0.029 default, 0.22 for loamy/loamy to sandy soil.
[j] 0.004 - dump truck unloading coal, loamy/sandy/clayey/clay loamy soil. 0.02 - scraper unloading topsoil, loamy/loamy to sandy soil.
[k] Front end loaders loading topsoil, uncontrolled.
[l] 3.6 m/s mean wind speed; k value of 0.74
[m] NPI data is for active coal stockpiles, AP 42 data represents seeded land, stripped overburden, graded overburden.

2.8.2 Particle Size Fractions

One of the key constraints associated with adopting the emissions data presented in Table 2-12 for current emission estimation studies relates to the fact that both the US EPA and the Australian emission sampling data was restricted to total suspended particulates. When these studies were completed over 30 years ago, suspended particulate exposure criteria related to exposure to all size fractions of suspended particulate, and size specific health based particulate goals were not in force. Since that time, particularly in the last two decades, the focus of particulate health goals has been toward specific size fractions – primarily PM_{10} and $PM_{2\,5}$. In the absence of source emission data for these size fractions for inclusion in the US EPA AP42 and Australian NPI emission manuals, assumed particle size distributions are applied to the estimated TSP emission based on a measurement data relating to particle size fractionation for NSW coal mines. In Australia, at the time of the TSP emission factor studies, particle size distributions were determined for a range of activities by collecting particulate samples on filters using high volume sampling techniques. The particulates were subsequently extracted from the filters using ethanol in an ultrasonic bath, and the particulate sizes were determined using a Malvern 2600 laser particulate counter. The assumed particle size distributions adopted for size fractionation of the TSP data is presented in Table 2-13, along with the NPI and AP42 recommended size fractionation ratios. The size distributions determined in this study (State Pollution Control Commission NSW, 1986) are the basis for the size fractionation adopted in the 2012 NPI emission estimation manual for mining.

Table 2-13: Particle Size Fraction in TSP

Activity	**Hunter Valley (State Pollution Control Commission NSW, 1986)**			**NPI 2012**	**US EPA AP42**	
	0 – 2.5 µm	2.5 – 15 µm	15 – 30 µm	PM_{10}: TSP ratio	PM_{10}:TSP ratio	$PM_{2\,5}$:TSP ratio
Overburden and Coal Haul	0.06	0.53	0.41	0.3	n/a	n/a
Dragline handling	0.07	0.50	0.43	0.43	n/a	n/a
Overburden drilling	0.09	0.62	0.29	0.52	0.75	0.017
Dumping	0.04	0.44	0.53	0.42	n/a	n/a
Coal dumping	0.04	0.49	0.47	0.42	n/a	n/a
Shovel loading coal to trucks	0.05	0.58	0.37	0.48	0.75	0.019
Average	0.06	0.52	0.43	0.43	0.75	0.018

Comparison of the datasets presented in Table 2-13 indicates that the US EPA PM_{10}:TSP fraction of 0.75 is high compared to the NPI 2012 ratio. The average NPI ratio of 0.43 is equivalent to the $PM_{15} – PM_{30}$ fraction as measured in the 1986 Hunter Valley study, hence may itself be an underestimate. The US EPA notes, in AP42 Section 11.9.6, that while efforts have been made to revise the emission factors for surface coal mines to ensure that they do not overestimate emissions from western surface coal mines, this has proved impractical due to resource and technical limitations. The US EPA identify that there is a tendency

for overprediction of particulate matter impact for PM_{10} from coal mining, for as yet undetermined causes. For this reason, the AP42 emission equations for PM_{10} are not approved for use in regulatory decision making in the United States. This notwithstanding, the US EPA identifies that as there are no better alternative data currently available, the emission equations for PM_{10} are provided, and that users should accordingly use these factors with caution and an awareness of their likely limitations (US EPA, 1998a). Overall, whilst the US EPA PM_{10} fraction has been identified as high, recent research has demonstrated that laser particle sizing techniques such as the Malvern, as used in the 1986 study (State Pollution Control Commission NSW, 1986) underestimate fine particulates in the < PM_{10} size fraction, and overestimate coarse particulates in the > PM_{10} (Vdović, Obhođaš et al, 2010).Furthermore, as the Malvern method results in estimation of the geometric mean particle diameter, comparison with aerodynamic diameters may result in average underestimation by a factor of 1.4 (Reid et al, 2003). These confounding factors mean that it is not clear whether the US EPA size fractionation is in fact erroneously high, or whether the size fractionation relied upon in the NPI 2012 emission estimation manual represents an underestimation of the finer fractions.

In terms of regional differences, it is noted in the Hunter Valley study that there can be significant differences in particle size fraction distributions between mines due to geological differences, and even within mine sites in cases where the mine geology is not homogenous (State Pollution Control Commission NSW, 1986).

The use of Total Suspended Particulate (TSP) data for the purposes of estimating PM_{10} emissions, based on assumed particle size distributions, has also been identified in the literature as a significant area of uncertainty. Huertas et al (2012a) identified that there are no studies currently related to PM_{10} emissions for open cut coal mining and, given that PM_{10} is more harmful to health than TSP, there is a need for standardized emissions estimation methodologies for PM_{10} emissions. Huertas also identified that the particle size distribution varies significantly for different activities at surface mines (Huertas, Camacho et al, 2012) and that further research is required to adapt existing published emission factors to local mining conditions.

2.8.3 Emission Estimation Factor Accuracy

The overall accuracy of emission inventories is an important consideration. Recently, the Australian National Pollutant Inventory (NPI) emissions database has been criticised for inaccuracy and, furthermore, for providing inconsistent data for major sources of emissions (Cooper, Green et al, 2017). In particular, Cooper criticises the current NPI emission estimation method for particulates from unsurfaced roads, identifying the methods as outdated, and as using inaccurate values and formulas used to estimate emissions paved/unpaved roads.

Given the significance of surface mines as a source of particulate emissions in Australia, the accuracy of the emissions estimation techniques adopted for proposed mines and for existing mines via the National Pollutant Inventory is of particular importance. The NPI is one of the sources of data used to inform regulatory decisions about environmental management, and to identify those industry groups and activities that pose the greatest risk to community health. Environmental impact statements assess the risk of impacts and identify a range of mitigation solutions that are typically defined in approvals for the operational phase of the activity. These mitigation requirements impose significant costs on the mine operator and it is essential that the EIS correctly identifies the sources posing the greatest risk to the environment so that any mitigation expenditure achieves the greatest benefit to the community. Significant community risks can arise if the emissions estimations are underestimates. Conversely significant community economic disbenefit can arise if the emissions are over-estimated, or if mitigation measures are not directed at the most significant source. In the latter example, environmental monitoring could demonstrate on-going

environmental non-compliances despite the implementation of mitigation, and additional mitigation expenditure may be required as the original expenditure was not targeted at the primary source.

Average emissions differ significantly from source to source. The extent of variability that exists, even among similar individual sources, can be large depending on the process, control system, and pollutant. Even when the major process variables are accounted for, the emission factors developed may be the result of averaging source tests that differ by factors of five or more (US EPA, 1995b). This feature is exhibited in the dataset that formed the basis for many of the PM_{10} emission factors adopted in the current Australian National Pollutant Inventory Mining Handbook (Environment Australia, 2012a). Review of the report (NERDCC, 1988) that provided these data confirms that the typical standard deviation for the dataset was up to 2.9 times the average emission factor. This illustrates the variability that is present amongst emissions of this type for currently adopted emission factors.

As noted previously, emission factors published by the United States form the basis for many emission estimation methodologies used worldwide; furthermore, they form the basis for many of the emission factors used in the Australian National Pollutant Inventory emission estimation methodologies. Historically, US EPA quality ratings of emissions test data and test reports were largely subjective. This is because each test program adopted varying approaches and methodologies (i.e. the US EPA identify that no two facilities, their operation or the tests conducted at those facilities, are exactly alike). Due to this variability, the EPA applied letter-grade quality ratings (A through E) for test reports based upon review of the process operations tested, test methods and sample procedures, process information available, and the overall analysis and calculations. The approach to emissions estimation methodology and quality ratings process adopted by the US EPA is illustrated in Figure 2-1. The increasing cost associated with the development of high-quality emission factors is also highlighted.

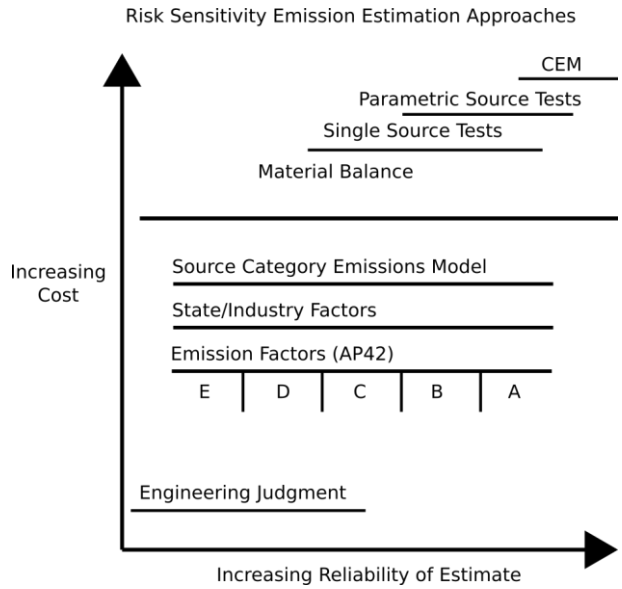


Figure 2-1: US EPA Emission Factor Quality Ratings (US EPA, 1995b)

The Australian NPI follows a similar quality rating scheme, providing for ratings from A to E, with an additional category ‘U’ denoting the rating is unclassified.

The NPI emission factor ratings are as follows:

A Excellent
B Above average
C Average
D Below average
E Poor
U Unrated

The data quality ratings attributed to the US EPA AP42 and NPI particulate emissions estimates for mining activities are variable. The quality ratings for TSP emission estimation are presented in Table 2-14. Data highlighted in bold indicates that the default emission factors were based on sampling completed in the Hunter Valley.

Table 2-14: Comparison of NPI and US Mining Default TSP Emission Factor Quality Ratings

Activity	**NPI 2012**	**US EPA AP42 1995**
Truck and shovel loading of coal (kg/t)	**U**	B
Truck and shovel loading of overburden (kg/t)	**U**	B
Dragline handling overburden (kg/m^3)	**B**	B
Coal and overburden transport by off-highway trucks (kg/veh/km)	B	B
Coal transport by off-highway trucks (kg/veh/km)	B	B
Overburden transport by off-highway trucks (kg/veh/km)	B	B
Trucks dumping overburden (kg/t)	**U**	E
Trucks dumping coal into ROM hopper (kg/t)	Not stated	E
Bulldozers on coal kg/hr/veh	B	C
Bulldozers on material other than coal kg/hr/veh	B	B
Drilling kg/hole	C	C
Blasting kg/blast	C	C
Scrapers – travel mode kg/veh/km	A	E
Scrapers – removing topsoil kg/veh/km	E	E
Graders kg/veh/km	B	C
Loading Stockpiles kg/t	U	E
Unloading from stockpiles kg/t	U	E
Loading to trains kg/t	U	E
Miscellaneous transfers kg/t/transfer point	U	E
Wind erosion kg/ha/hr	**U**	C

Source: (Environment Australia, 2012b; US EPA, 1995b)

Table 2-14 identifies that even where the emission factors are identified as relying on local data, the resultant emission factor may still be classed as 'U'. This indicates a high uncertainty regarding the accuracy of the emission factors.

Whilst the Australian NPI emission estimation manual provides single quality rating factors for TSP and PM_{10}, the US EPA AP42 provides specific emission factor quality ratings for the assumed size fractionated emission rates for PM_{15}, PM_{10} and $PM_{2.5}$. The quality ratings for the emission estimations for these size fractions, as defined in AP42, are presented in Table 2-15.

Table 2-15: Comparison of US EPA Emission Factor Quality Ratings – Size Fractionated Emission Estimates (US EPA, 1995b)

Activity	**TSP**	**PM_{15}**	**PM_{10}**	**$PM_{2.5}$**
Truck and shovel loading of coal (kg/t)	B	B	C	C
Truck and shovel loading of overburden (kg/t)	B	n/a	n/a	n/a
Dragline handling overburden (kg/m^3)	B	C	D	D
Coal and overburden transport by off-highway trucks (kg/veh/km)	B	B	B	B
Coal transport by off-highway trucks (kg/veh/km)	B	B	B	B
Overburden transport by off-highway trucks (kg/veh/km)	B	B	B	B
Trucks dumping overburden (kg/t)	E	n/a	n/a	n/a
Trucks dumping coal into ROM hopper (kg/t)	E	n/a	n/a	n/a
Bulldozers on coal kg/hr/veh	C	C	D	D
Bulldozers on material other than coal kg/hr/veh	B	C	D	D
Drilling kg/hole	C	n/a	n/a	n/a
Blasting kg/blast	C	-	D	D
Scrapers – travel mode kg/veh/km	E	n/a	n/a	n/a
Scrapers – removing topsoil kg/veh/km	E	n/a	n/a	n/a
Graders kg/veh/km	C	C	D	D
Loading Stockpiles kg/t	E	n/a	n/a	n/a
Unloading from stockpiles kg/t	E	n/a	n/a	n/a
Loading to trains kg/t	E	n/a	n/a	n/a
Miscellaneous transfers kg/t/transfer point	E	n/a	n/a	n/a
Wind erosion kg/ha/hr	C	n/a	n/a	n/a

The comparison of the quality ratings by size fraction demonstrates that few TSP emission factors have a high (A or B) rating and many are unclassified. For the finer particle fractions, in almost all cases the quality rating is less than the TSP factor. This is because the ratings for the size fractions are based on the

TSP data, with an adjustment applied for size fractionation, hence the rating must always be lower than the TSP emission rate. Were size specific emissions data available, then higher quality ratings would be expected to apply.

Review of the quality data confirms that the NPI quality ratings are in some cases higher than the quality designation applied to the original data source. For example, in the case of graders and scrapers the NPI emission rates are based on AP 42 data rated as C, however the NPI attributes a quality rating of B to the emission estimation method in the NPI manual. With respect to application of the AP 42 data for US western surface coal mines to other regions, the US EPA identifies that the quality ratings should be reduced by one category where local data on silt contents and moisture content is not available, as the accuracy of emissions estimates is likely to be lower. Applying this to an Australian situation, using the EIS studies identified in Table 2-11 as an example, local particle size data is referenced in only one of these EIS studies. This indicates that the NPI 2012 emission data quality ratings should be further reduced by one category based on the typical practices adopted in applying these methods currently in Australia. Therefore, the currently adopted quality ratings for the NPI designate a higher quality rating than is appropriate in many cases.

2.9 SUITABILITY OF CURRENT EMISSION ESTIMATION METHODS

2.9.1 Existing Approaches

It is apparent from this analysis of the currently adopted particulate emission rates for open cut coal mines that these methods are based on studies that are over 25 years old, when regulatory policies related to TSP, and PM_{10} and $PM_{2.5}$ environmental regulations were not in force. While mining techniques have not changed significantly over this time period, some changes have occurred and differences in emission rates may arise as a result. In addition, over this time period, the proportion of overburden to mined coal has increased and this may alter the relative proportions of particulates in the PM_{10} and $PM_{2.5}$ size fractions relative to the overall TSP emissions.

Community health goals are currently focused almost exclusively on the PM_{10} and $PM_{2.5}$ size fractions and these fractions are not represented in the currently available emissions datasets. To provide an indication of the expected emissions of particulates from open cut coal mines, size fractionation data for a range of mining activities is applied to the TSP emission rates derived from the empirical datasets. Whilst this approach may be valid. Furthermore, the empirical data for Australia relates only to the Hunter Valley mines, whereas the NPI emissions data for 2010 – 2015 indicate that the largest particulate emissions sources nationally are coal mines in Queensland. Given the regional differences in particulate emissions that may occur due to varying geology, the applicability of the NSW emission data to the modern mining sector in Queensland has not been verified.

In terms of the particle size distribution data defined in US EPA AP42, the inherent uncertainty associated with the PM_{10} emission estimation equations is noted. Specifically, the US EPA identify that PM_{10} estimates completed on the basis of the size fractionations presented in AP42 have considerable uncertainty and are not approved for use in regulatory decision making or approvals. This notwithstanding, these emission equations remain a primary source of data relating to PM_{10} emissions from open cut mining,

The data quality for the resultant emission factors is average to poor, and in application is downgraded further as few practitioners are adopting site specific vales for key variables in the emissions equations. Furthermore, it has been shown that two of the default emission factors adopted in the current NPI emission estimation manual for mining are incorrect, as they relate to controlled rather than uncontrolled emission rates.

Given the significance of mining as a source of fine particulate emissions in Australia, the availability of high quality, well documented particulate emission estimation methods that are applicable to the key mining regions is a fundamental requirement. These data are a necessary input to the environmental decision-making framework in Australia. The need to adopt activity specific particulate emission techniques to ensure the appropriateness of emissions estimates adopted in national pollutant inventories such as the Australian NPI has been highlighted (Weng, Mudd et al, 2012). Weng et al also identify the limitations in our ability to interpret national pollutant inventory data due to the lack of clarity in the underlying causes of the total mass emissions.

In broader terms, the published literature has also identified the need for appropriate testing procedures to predict the properties of mineral dust from mining operations and mineral fractionation according to particle size (Noble, Parbhakar-Fox et al, 2017). In 2016 Patra reviewed available data relating to the emission and human health impact of particulate matter from surface mining operations. The need for further research to both determine emission rates of PM generated due to mining activities, and to characterise the physical and chemical properties of PM to allow the assessment of potential impacts in the surrounding atmosphere and on the health of mine workers in the mines was identified (Patra, Gautam et al, 2016). Patra also identified that the health hazards and exposure to PM deserve to be investigated as thoroughly as for PM generated from mining operations to contribute to safer workplaces and healthier environments at and in the surroundings of surface mines. Ghose has identified that, despite the detrimental impacts of coal mining emissions and the move to opencast mines, well guided research is required to determine the appropriate emission rates to consider in planning studies for new mines (Ghose, 2007a). While Ghose evaluates the available research in the context of mines in India, this work has yet to be completed for Australian mines.

2.10 Identified Gaps in Current Emission Estimation Datasets for Mining

Based on this literature review, the issues associated with the current emissions estimation datasets for open cut coal mining in Australia can be summarised as follows:

- the Australian empirical data relating to TSP emissions from open cut coal mines is over 25 years old. These data are representative of the mine operating techniques and practices at the time. Data representing current operating practices is not currently available;
- there are no empirical data for size fractionated emissions from mining activities in Australia or the US to verify the currently adopted particulate emissions estimates for PM_{10} and $PM_{2\,5}$;
- the quality of the currently adopted emissions estimation techniques for mining is relatively poor, and inaccuracies and inconsistent emission data has been identified in the Australian National Pollutant Inventory emissions estimates;
- there are no emissions data for coal mines in Queensland, the largest source of particulates in Australia today.

To address these deficiencies in the existing commonly used datasets, a research program was designed to address the key question:

What are the appropriate emission rates to define PM_{10} and $PM_{2.5}$ particulate emissions from sources at open cut coal mines in different locations in Australia?

This has been addressed through a research programme that provides the empirical data and analysis for addressing the following specific focus questions:

i. What are the key sources of emissions in Australian open cut coal mines?

ii. What emission rates are currently used to define particulate emissions from mining activities at open cut coal mines, and how appropriate are they?

iii. Are there significant variation in emission rates within mines due to local features of the mine or operations?

iv. What are the appropriate emission rates to adopt for mines in different regions of Australia?

The research has involved collection of source emission samples from a range of activities at Australian open cut coal mines supplemented with environmental sampling and particulate chemical and size characterisation.

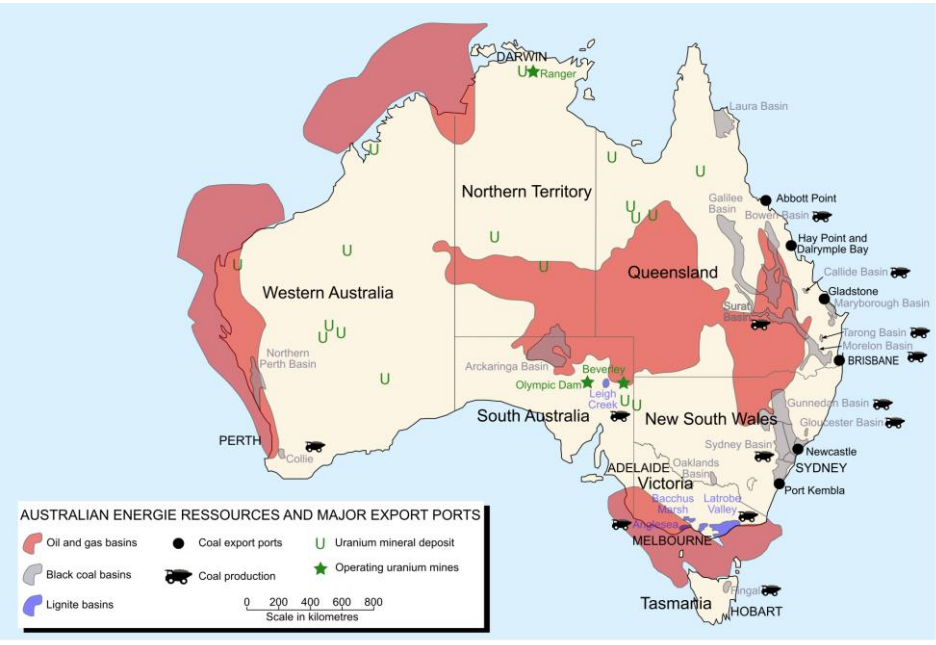
3 RESEARCH METHODOLOGY

3.1 INTRODUCTION

3.1.1 Research Programme

To investigate the research questions, three research projects funded by the Australian Coal Association Research Program have been completed. Firstly, a broad investigation of particulate size distribution and composition was completed to identify key aspects of mining particulates and the relative significance of particulate sources within and external to an open cut coal mine. Secondly, TSP emission rates for haul routes and other open erodible surfaces were measured and quantified. Finally, to address the identified gap in existing knowledge relating to size fractionated emission rates, quantification of $PM_{2.5}$ emission rates for a broad range of open cut coal mining sources was carried out in two representative coal mining areas.

The overall approach and methodology for each of these studies is discussed in the following sections. For each of the studies, fieldwork was completed at one or more operational open cut coal mines in key mining regions in Queensland (QLD) and New South Wales (NSW). The research funding body, ACARP, requested that the mine locations were not specifically identified in publications relating to the research. For this reason, the mines are not identified by name in this thesis. Figure 3-1 identifies the black coal mining regions (New South Wales - Hunter Valley, Central Queensland - Bowen Basin and South-East Queensland - Tarong Basin) in which the sampling was completed. Each mine site where sampling was completed is identified in the research by State (NSW or QLD) and, as more than one mine was sampled for each State, with an individual number to identify each mine.



Source: Australian Government Department of Resources Energy and Tourism, 2008; Historicair, 2019

Figure 3-1 - Key Coal Mining Regions in Australia

3.1.2 Rationale for Selection of Sampling Techniques

The selection of appropriate particulate sampling techniques for each of the research studies is of primary importance. A variety of methodologies are available for sampling of particulate size fractions. Each method exhibits differing accuracies based on the method of sample collection, sizing technique adopted, analysis method, the environment being sampled, the particulate concentrations being measured and the accuracy of instrument calibrations (Mahowald et al, 2014; Winkel et al, 2014).

Whilst the relative accuracy and limitations of particulate sampling techniques is of relevance, overall, the key limitation affecting the selection of the sampling methodologies for the research related to safety restrictions imposed by the mine and the lack of mains power at the sample locations in the operational areas of the mine. As a result, sampling techniques were limited to battery operated methodologies. High volume gravimetric methods and TEOM samplers are approved reference methods for particulate sampling in Australia (QLD DEHP, 1997; NSW EPA, 2006) with a commensurate degree of accuracy, however both require mains power hence were excluded from the study. Therefore, alternative particulate sampling methods were adopted for the research.

The approach adopted for selection of portable sampling methods was to use gravimetric methods where possible, for consistency with the approved reference methods, and to maintain a relative high degree of accuracy. Due to the power supply constraint, this necessitated adoption of low volume gravimetric techniques. For sampling of $PM_{2.5}$, low volume gravimetric techniques are only suitable where extended sample durations (over many days) were possible. Therefore, for the $PM_{2.5}$ emission sampling for Phase 3 of the research, portable, battery operated beta-attenuation monitors were selected for the sampling. Research has demonstrated that under low humidity conditions and in the absence of significant volatile compounds, there is a significant level of comparability between low volume gravimetric methods and beta attenuation sampling (Gebicki et al, 2012; Salminen et al, 2003; Triantafyllou et al 2016). Due to the crustal nature of the majority of particulates and the relatively low humidity sampling environments, these conditions are satisfied for the mine sampling locations. Therefore, adoption of the beta-attenuation sampling method for $PM_{2.5}$ emissions was appropriate.

The specific sampling methods adopted for each phase of the research are discussed in detail in the following sections.

3.2 PHASE 1 – INITIAL INVESTIGATION

3.2.1 Overview

The initial investigation phase involved two distinct study elements. Firstly, short-term real-time sampling was completed downwind of a range of coal mine sources to confirm particle size distributions and to gauge the relative significance of different sources in terms of particulate emissions. Secondly, longer term sampling was completed both within and at the boundary of coal mines to provide data relating to particulate size distributions. A selection of the boundary samples were further analysed by microscopy and energy dispersive x-ray photometry (EDS) to determine particle size and chemical composition.

Table 3-1 presents a summary of the overall sampling programme adopted for this research phase. Particulate monitoring was completed at three Australian open cut coal mines: one mine in Central Queensland (QLD), identified as QLD Mine 1, and two in New South Wales (NSW) identified as NSW Mines 2 and 3. The sampling methods are discussed in the following sections.

3.2.2 Environmental Sampling

A low volume gravimetric technique was adopted to measure the longer-term concentrations beyond the boundary of each of the three mines for two particulate size fractions – PM_{10} and $PM_{2.5}$. The sampling involved use of three Minivol Portable Air Samplers (Airmetrics, USA). The Minivol samplers were fitted with Millipore PTFE filters and size selective inlet impactors for PM_{10} and $PM_{2.5}$. The Minivol samplers are approved by the US EPA for the purposes of 'saturation samplers', where multiple particulate samples are to be collected simultaneously at different sample locations.

The samplers were calibrated using a flow rotameter prior to, and at the end of each sample. A minimum sample duration of 24 hours per filter was adopted; this was extended to 3 days per filter for the majority of samples to improve the sensitivity of the gravimetric analysis. Field blanks were analysed and the results of the gravimetric analysis corrected by the difference in the blank filter mass changes. The blank corrections resulted in adjustments of <5 % of the average mass increase of the sampled filters. Impactor plates were cleaned and greased on a weekly basis in accordance with the manufacturer's recommendations. Filters were weighed pre- and post- sampling in a temperature and humidity-controlled laboratory using a digital balance with a resolution of 1 µg.

3.2.3 Quality Assurance

The accuracy of gravimetric sampling is largely determined by three key factors:

i. efficiency of size selective inlets;

ii. accuracy of air flow calibration.

iii. error attached to the filter weighing;

The size selective inlets utilised in the sampling have been tested and approved by the US EPA to ensure they comply with the required degree of accuracy. Air flow is a key determinant in ensuring the effectiveness of the size selective inlets. Therefore, air flow rates at the start and finish of each sample period were recorded, and calibration of the air flow of the sampling instruments was undertaken during the monitoring programme.

The error attached to filter weighing techniques was minimised by:

- ensuring a suitable mass increase was achieved on the sample filter;
- undertaking all mass analyses in controlled temperature and humidity conditions;
- allowing filters to reach equilibrium for a period of 24 hours in a temperature and humidity-controlled room prior to weighing;
- providing field blanks to allow post analysis quality assurance checks and, where necessary, adjustment of results;
- ensuring the selected filters were not prone to electro-static influences.

All of these factors were taken into consideration in the sampling and analysis process. Overall, the quality assurance of filter sample results revealed a high degree of repeatability when random checks of the original analyses were subsequently undertaken.

Sampling was completed for a minimum sampling period of 7 days for PM_{10} and $PM_{2.5}$ at each sampling location. The overall average concentrations in the PM_{10} and $PM_{2.5}$ size fractions were subsequently determined to allow comparison of the size fractionation of the particulate material. At Mine 1 (QLD) sampling was completed at a range of positions external and within the mine to assess the influence of a range of mining activities on ambient particulate concentrations. For the mines in NSW (Mine 2 and 3), the Minivol samplers were co-located with total suspended particulate (TSP) high volume samplers operated by the mining companies, to allow for comparison of the $PM_{2.5}$ and PM_{10} concentrations with overall TSP concentrations. The high-volume samplers were operated by the mines for 24-hour periods, to provide average 24-hour TSP concentrations. The sample locations were external to the mine, in close proximity to the nearest off-site residential land uses.

3.2.4 Source Sampling

As mining operations tend to vary in terms of activity type and location on a daily basis and, for some activities, on an hour by hour basis, the low volume sampling technique was impractical to use for source emission size fraction characterisation. As an alternative, a real time optical particulate monitor (Model 1.105, GRIMM Aerosol Technik, Ainring, Germany) was used, allowing for sampling in a particle range 0.3-20 µm over periods of 10 – 18 minutes when the activity of interest was occurring. The sampling position was located downwind of the activity at distances of 2 m – 150 m depending on the nature of the activity and safety restrictions. The Grimm sampler was selected as the most suitable optical sampler, as alternate instruments such as the Dustrak are capable of measuring only a single size fraction at any one time. The Grimm sampler provided for measurement of a number of particle size fractions simultaneously hence was more suitable for assessing size fractionation for the emission sources.

3.2.5 Additional Analysis

Particulate size and composition information was obtained by completing additional analysis of a selection of gravimetric filters. Two types of analysis were undertaken on the filters. Firstly, optical analysis was undertaken to assess the type and typical size of particulates present. The optical analysis involved selecting at random an area on each filter, determining the number of particles present in each area and classing the size of each individual particle. The presence of pollen particles on the filters was also determined during the visual analysis.

Subsequent elemental composition analysis of individual particles was undertaken using energy-dispersive x-ray spectrometry (EDS). Again, random areas of each filter were selected for the purposes of the analysis. The EDS analysis was completed by Queensland University of Technology. The elemental composition data was determined for silica, carbon, soil, and ‘other’. The ‘other’ category includes particulates such as salt (NaCl), and compounds containing two or more of the element’s sodium, magnesium, sulphur, chlorine, calcium, iron and zinc. Due to image and analytical resolution limitations, particles with a diameter of less than 1 µm in diameter were not considered in the analysis.

Table 3-1: Summary of Sampling Programme

Mine Details	Sampling Type	Sources of Emissions	Monitoring Position	Sampling Technique	Sampling Duration	Number of Filters
Mine 1 - QLD	Source	Product Stockpile	120 m downwind	Optical – Grimm	18 minutes	n/a
	Source	Coal Loading and Haul	50 m downwind	Optical – Grimm	17 minutes	n/a
	Source	Topsoil Spreading	80 – 150 downwind	Optical – Grimm	12 minutes	n/a
	Source	Dragline on Overburden	50 downwind	Optical – Grimm	15 minutes	n/a
	Source	Pre-strip	60 downwind	Optical – Grimm	14 minutes	n/a
	Source	Coal Preparation Plant	30 m, 60 m and 120 to product stockpile at Coal Preparation Plant	Optical – Grimm	18 minutes at each position	n/a
	Boundary	Mine haul road	50 m	Gravimetric – Minivol	21 days	7 – $PM_{2.5}$
	Boundary	Product stockpile	550 m	Gravimetric – Minivol	20 days	7 – $PM_{2.5}$
	Boundary	Background position representing nearest receptor		Gravimetric – Minivol	20 days	7 – $PM_{2.5}$
	Transect Position 1	Product stockpile	200 m	Gravimetric – Minivol	20 days – $PM_{2.5}$ 22 days – PM_{10}	7 – $PM_{2.5}$ 9 – PM_{10}
	Transect Position 2	Product stockpile and mine processing area	400 m	Gravimetric – Minivol	20 days – $PM_{2.5}$ 22 days – PM_{10}	7 – $PM_{2.5}$ 9 – PM_{10}
	Transect Position 3	Product stockpile and mine processing area	1,100 m	Gravimetric – Minivol	20 days – $PM_{2.5}$ 22 days – PM_{10}	7 – $PM_{2.5}$ 9 – PM_{10}
Mine 2 - NSW	Source	Coal Haul	11 m to edge of road	Optical	68 minutes	n/a
	Source	Coal Haul	2 m to edge of road	Optical	8 minutes	n/a
	Source	Dragline	On board	Optical	26 minutes	n/a

Mine Details	Sampling Type	Sources of Emissions	Monitoring Position	Sampling Technique	Sampling Duration	Number of Filters
	Source	Dumping into Coal Hopper	10 – 15 m downwind	Optical	16 minutes	n/a
	Source	Drill	5 m downwind	Optical	9 minutes	n/a
	Source	Coal & Overburden loading	10 – 30 m downwind	Optical	28 minutes	n/a
	Source	Coal Haul	2 m to edge of road	Optical	22 minutes	n/a
	Source	Spontaneous Combustion	20 m downwind	Optical	7 minutes	n/a
	External to Mine Boundary	Operating mine as a whole	3 km to active working pit	Gravimetric – Minivol	9 days – $PM_{2.5}$ 9 days – PM_{10}	4 – $PM_{2.5}$ 4 – PM_{10}
	External to Mine Boundary	Operating mine as a whole	4 km to active working pit	Gravimetric – Minivol	9 days – $PM_{2.5}$ 9 days – PM_{10}	4 – $PM_{2.5}$ 4 – PM_{10}
Mine 3 – NSW	Source	Coal Loading	50 m	Optical	58 minutes	n/a
	Source	Shovel	On board	Optical	31 minutes	n/a
	Source	Drill	On board	Optical	21 minutes	n/a
	Source	Dozer	50 m	Optical	14 minutes	n/a
	Source	Coal Dumping	2 m	Optical	33 minutes	n/a
	Source	Coal preparation plant	5 m from hopper	Optical	32 minutes	n/a
	Source	Coal Haul	5 m	Optical	7 minutes	n/a
	External to Mine Boundary	Operating mine as a whole	1 km to active working pit	Gravimetric - Minivol and high-volume sampler operated by mine	9 days – $PM_{2.5}$ 9 days – PM_{10}	4 – $PM_{2.5}$ 4 – PM_{10}

3.2.6 Sampling Limitations

Optical particulate counters such as the Grimm rely on light scattering techniques to identify particle number and an estimate of particulate size as a geometric mean. These instruments apply an estimate of particle density to extrapolate from the particle size and number to an estimated concentration of particulates in the air stream. The response of optical instruments such as the Grimm is dependent upon the size, shape and refractive index of the particulates. Therefore, the measured concentrations may be dependent upon the shape and size range of particulates experienced in a given locality, which may affect the accuracy of the particulate measurements (Renard, Thaury et al, 2010). The Grimm sampler allows the use of a pre-weighed filter in the sample airstream discharge to provide for calibration of the results of the optical analysis with the mass concentration determined using a gravimetric technique. Due to problems removing the pre-weighed filter from the filter housing of the Grimm sampler, it was not possible to complete a calibration of the instrument.

Furthermore, the Grimm counter, consistent with similar optical based particulate monitoring units, typically operate at low flow rates that are insufficient to carry particulates >35 µm into the instrument through the sampling inlet. As the TSP size fraction includes particulates >35 µm in diameter, the Grimm sampler may underestimate TSP concentrations and this is a further limitation of the method. The high-volume method samples particulates with a diameter of up to 100 µm, hence the Grimm sampling method could underestimate the mass concentrations.

Investigations of the accuracy of optical particulate samplers such as the Grimm unit is well documented in the literature, the potential for significant measurement errors has been identified (Heim et al, 2008; Rivas et al, 2015) and the potential underestimation compared to reference methods has been documented (Viana et al, 2015). The necessity of calibrating optical instruments through co-location with a gravimetric sampler to reduce bias (which may be associated with resolution of the particulate shape, size, colour and the low flow rate of optical samplers) have been documented in the literature (Yanosky, Williams et al, 2002; Cheng, 2008; Tasic, 2012; Degan 2015).

Due to these limitations, the inability to cross-calibrate the instrument with the gravimetric filter and the varying distances to operations during the sampling, direct comparison of the optical derived source concentration data is not meaningful. Therefore, these monitoring data have been considered in terms of mass percentage in each size fraction only.

A further limitation of the Grimm sampler for this study was the inability of the sampler to determine the $PM_{2.5}$ fraction. As a surrogate, the study adopts the PM_2 fraction as determined by the Grimm instrument.

Geological data representative of the sources of emissions at the mines was not completed at the time of sampling. This would have been of assistance in correlating the results of the study to specific local conditions. In particular, this information would have been of assistance in determining the potential for differing mass densities of the particulates for different sources of emissions. The optical microscopy and EDS analysis provides some information in this regard, as discussed in Section 4.4.

3.3 PHASE 2 – PARTICULATE EMISSION RATES FROM SURFACES

3.3.1 Overview

Determination of particulate emission rates for fugitive surface sources at open cut coal mines is a complex issue. A range of methods have been utilised in previous published studies including:

- downwind isokinetic sampling using high volume samplers (NERDCC, 1988);

- upwind-downwind method using high volume samplers (Axetell & Cowherd, 1981; Frankell, 1993);
- exposure profiling technique involving isokinetic measurements immediately downwind of the source at multiple points in the vertical plane (Frankell, 1993; US EPA, 1998b);
- low volume sampling techniques (Pietersma, Stetler et al, 1996);
- quasi-stack method in which the source is partially or fully enclosed and stack sampling approaches utilised (Frankell, 1993; US EPA, 1998b); and
- wind tunnel testing (Carras, Riley et al, 1999; McKenna Neuman, Boulton et al, 2009; Strong, Leys et al, 2016).

Of the sampling methods available, the approaches that enclose or condition the test environment offer the greatest opportunity for minimising external influences such as meteorology (e.g. the quasi-stack and wind tunnel methodologies). Portable wind tunnels have been in use since the early 1950s where they originally were used for testing of erosion rates for agricultural surfaces (Gillette, 1978; Zingg, 1951). Design considerations and validation have been addressed in numerous published papers (Bocharov, 1984, Carras, Riley et al, 1999, James, Pulgarin et al, 2001, Maurer, Herrmann et al, 2006, Pietersma, Stetler et al, 1996, Raupach & Leys, 1990). For this research, the wind tunnel method was selected as the most appropriate for testing emission rates from open surface sources at coal mines, as the emission source has similar characteristics to agricultural erosion.

3.3.2 Source Characteristics

Particulate emission rates from open area emission sources are primarily governed by wind erosion of the surface with wind speed the parameter of primary importance (Strong, Leys et al, 2016), degree of stability of the surface (e.g., crusting), surface moisture content and surface silt content and silt loading (Sharratt & Vaddella, 2014). Surface moisture content and silt loadings are readily tested by standard laboratory methods. The degree of stability of the surface is a determinant of wind erodibility, and wind erosion of particulates can be tested using a portable wind tunnel of suitable design.

Raupach and Leys (1990) defined the key processes associated with surface wind erosion as follows:

- suspension of particles (particle size of < 20 μm);
- saltation processes (particle size of 20 – 1,000 μm);
- creep (particle size of >1,000 μm).

Of these processes, saltation is of primary significance from a particulate emission generation perspective. Saltation involves the impact of medium sized particles forcibly on the surface and, in turn, causing dislodgment of other particles from the surface. Creep involves the rolling motion of the largest particles across the surface. Therefore, to adequately simulate particle erosion from open surfaces the wind tunnel design must provide a logarithmic mean wind profile that is uniform over the eroding surface. The generation of an equilibrium boundary layer of a depth sufficient to contain the particle processes is suitable for this purpose (Maurer, Herrmann et al, 2006). In addition, saltation processes must be introduced to ensure that this important particle generation process occurs across the test face of the wind tunnel.

3.3.3 Wind Tunnel Design

The wind tunnel design adopted for the study was based on previous designs including ones provided by (Pietersma, Stetler et al, 1996) and (Raupach & Leys, 1990). Figure 3-2 shows a schematic diagram of the adopted wind tunnel design, with specific components of the design highlighted.

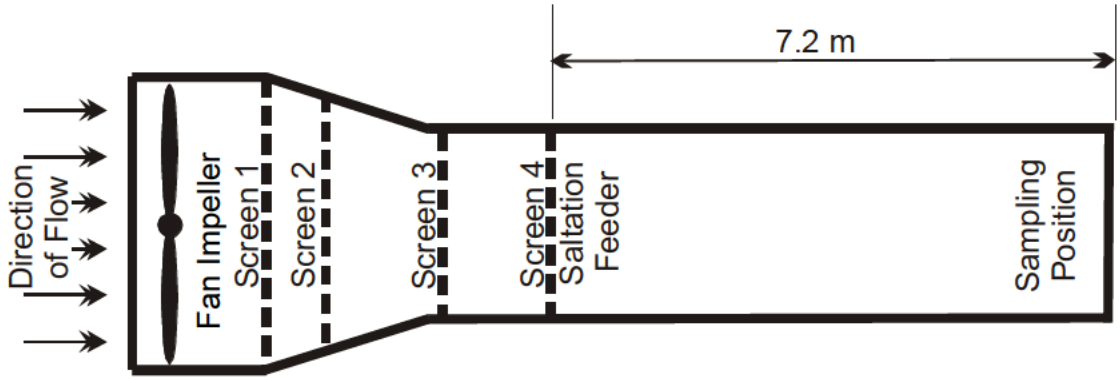


Figure 3-2: Wind Tunnel Schematic Diagram

The key elements of the wind tunnel are:

1. Variable speed axial fan of a size sufficient to generate the appropriate range of wind speeds within the wind tunnel.
2. Flow conditioning devices immediately downwind of the fan to introduce laminar flow (Screens 1 – 3). The screens introduce progressively smaller apertures, to transition the air flow from turbulent to laminar. Screens 1 and 2 comprised plastic mesh, and Screen 3 was constructed from a bed of plastic drinking straws glued together in a tight formation.
3. Flow conditioning device to introduce a logarithmic boundary layer downwind of the laminar flow section, just prior to commencement of the open test section of the wind tunnel (Screen 4). This comprised a series of thin, circular movable bars in a vertical and horizontal formation, to allow for adjustments to be made to the vertical profile for calibration purposes.
4. A 7.2 m working section with the base open to the eroding surface. Tunnel cross-section of 1 m wide, 1.2 m high.
5. Saltation particle feeder capable of introducing particulates at a variable feed rate to the 7.2 m long test section.

The wind tunnel trailer was designed to be capable of being towed by a standard four-wheel drive vehicle to minimise costs associated with transport and operation. To minimise weight, the tunnel was constructed from aluminium. A winch and rotating platform to allow lowering of the tunnel transition section to the test area was provided.

The performance of the wind tunnel was validated at a field test site prior to commencing the mine emission sampling, to confirm that the appropriate aerodynamic features were satisfactorily achieved. Both vertical and horizontal velocity profiles were determined using a TSI Velocicalc Hotwire Anemometer (TSI, USA). Aerodynamic criteria describing the depth and stability of the turbulent

boundary layer were determined for the wind tunnel and compared with empirical equations. The wind tunnel was validated up to a maximum simulated wind speed of 12 m s^{-1} at a height of 1 m. Tunnel validation data confirmed that the velocity profile throughout the sampling zone of the tunnel was a suitable representation of a vertical atmospheric wind speed profile. The fan RPM rates for each wind speed were determined during the validation phase, to allow these wind speeds to be generated for the test phase. Tunnel validation data (refer to Figure 3-3) confirmed that the velocity profile throughout the sampling zone of the tunnel was a suitable representation of a vertical atmospheric profile.

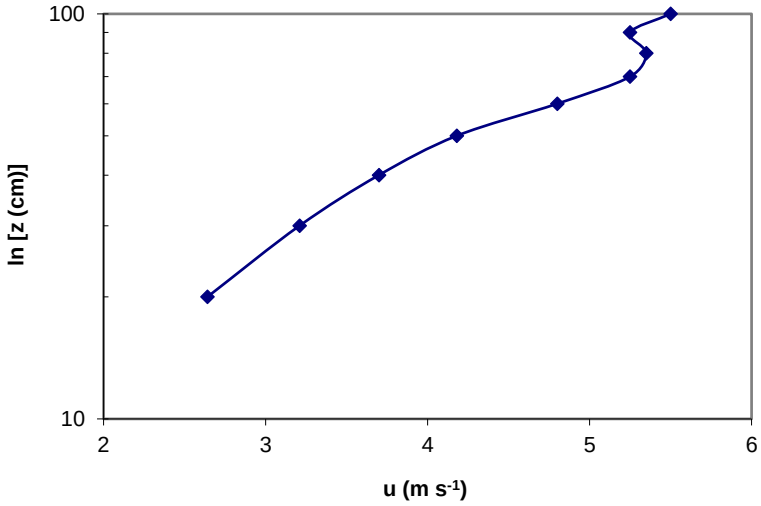


Figure 3-3: Natural Logarithm of Velocity (u) with Tunnel Height

Additional calibration and validation of the wind tunnel aerodynamics and wind field was also undertaken prior to commencement of testing at each mine included in the field trials.

3.3.4 Particulate Sampling Methodology

A wide variety of particulate sampling approaches have been adopted for collection of samples from portable wind tunnels. Examples include:

- modified high volume or dichotomous sampler, either non-isokinetic (US EPA, 1998) or isokinetic (SPCC, 1988);
- passive collection vanes, either quasi-isokinetic or non-isokinetic (Shao, McTainsh et al, 1993)
- real time monitoring using nephelometer or laser based sensors (James, Pulgarin et al, 2001);
- techniques involving vacuum suction of the majority of the emissions into a fabric filter bag (Carras, Riley et al, 1999).

Of these, three approaches were adopted for the research project as follows.

Firstly, to allow for identification of the point at which loose surface material is evacuated from the tunnel test surface and steady-state erosion processes have commenced, a real time screening approach was selected. This involved use of an OSIRIS real time nephelometer (Turnkey Instruments, Northwich, UK).

The second measurement approach was designed to sample PM_{10} particulates during the steady-state erosion phase operations of the wind tunnel. Minivol (Airmetrics, Springfield, OR USA) low volume gravimetric samplers with US EPA approved size separation inlets were utilised for this purpose. The Minivol low volume sampling technique[1] involved operation of the sample inlet and filter housing at a position within the end of the working section of the tunnel. A connection pipe to the pump was utilised to allow the pump unit to be remotely located (thus minimising wind field disturbance). The sample flow rate for the unit, 5 litres min^{-1}, is required to satisfy the design parameters for the PM_{10} size selective inlet. Millipore fluoropore filters were utilised for the Minivol sampling.

The third approach involved isokinetic gravimetric sampling using sample nozzles inserted into the tunnel. Isokinetic sampling involves drawing sample air into the sample apparatus at the same velocity as the sample air stream that is passing the sample point. This approach prevents entrainment of additional particles in the sampled air flow. For example, if the sampling velocity is higher than the gas stream velocity, a greater proportion of fine fraction particulate matter could be drawn into the sample from the surrounding air stream. Conversely, if the sample velocity is lower than the gas stream velocity, momentum effects could draw larger particles into the sample device while finer particles could pass around the sample head. The isokinetic method is not specifically referenced in the existing literature on wind tunnel sampling, although techniques for modified ambient samplers that attempt to sample in an isokinetic manner are referenced (Pietersma, Stetler et al, 1996). The isokinetic approach involved collection of total suspended particulates (TSP) on 47 mm teflon (PTFE) filters. The sample nozzle was inserted through an aperture in the side of the wind tunnel. A 47mm filter housing was utilised in conjunction with an inline gas meter and rotameter to establish isokinetic sample rates. Sample air volumes were typically > 1 m^3 over the duration of each test.

Of the three methods adopted, the isokinetic approach was considered to be the least likely to cause disturbance to wind fields within the tunnel. Entrainment of particles in the sample air stream is also minimised due to the adoption of isokinetic sampling principles.

Sampling was undertaken at two heights in the tunnel over the test period. The results of the two test positions represent an average concentration across the natural wind profile zone of the tunnel. All measurements were made above the saltation zone (0.3 m or above) to ensure that saltation particles were not introduced into the samples. Samples were collected for four different wind speeds.

3.3.5 Introduction of Saltation Processes

Saltation processes (and hence particulate emission rates) vary for different locations as a result of soil type and silt content. Even within a single mine, a range of different silt fractions can be present. To ensure this variability was considered in the testing, specific saltation particle sizes were selected for each individual test site.

Prior to commencing the test programme, soil samples were collected from each test site and analysed for particle size distribution. Size separated river sand was recombined in the same particle size distribution as found at each test site for use as saltation particles. Only particles in the size range 50 μm to 1 mm were included. Particles of this size are too large to become suspended in the atmosphere, hence do not cause interference when measuring erosion rates for suspended particulates.

[1] This is the technique that was used for the environmental sampling described in Section 4.

Saltation particle feed rates were determined on the basis of an initial wind tunnel test run undertaken at each sample location. The wind tunnel was configured and operated for a period of 60 seconds for each of the four wind speeds. During this period the OSIRIS instrument was used to monitor TSP concentrations in the tunnel at a height of 0.15 m.

The data collected was analysed to determine the maximum rate of saltation for each wind speed. This approximation of the amount of saltation particles generated was then replicated for the wind tunnel tests at that site through use of a variable control on the saltation feeder.

Inclusion of saltation processes in the test process allows simulation of a sustained natural wind event (with the exception of gustiness). One potential limitation of this approach is the significantly lengthened sample period required to achieve sufficient mass increase on the test filters to obtain a valid measurement. Previous studies concentrated on short duration test scenarios (without additional saltation particles) generally using similar sampling techniques (Carras, Riley et al, 1999; James, Pulgarin et al, 2001; Pietersma, Stetler et al, 1996; Raupach & Leys, 1990). A commonly reported problem with these studies, however, was that lack of a detectable change in filter weight with usable results being limited to those collected using nephelometer instruments (James, Pulgarin et al, 2001). Significantly extended sampling periods were adopted for this study in an attempt to obtain valid test results for a quantitative gravimetric method rather than relying on real time methods that generally provide only an estimate of actual mass concentrations.

3.3.6 Additional Tests

For haul route test sites, sampling was also completed to assess haul route erosion with and without surface watering. To measure the application rate of water to the haul route surface, a metal tray with a known surface area was located on the test surface where the water cart was to pass. After the water cart passed over the surface, the tray was collected and the retained water, that would normally enter the surface, was measured in a measuring cylinder. Based on this information, the rate of water application per square metre was determined.

3.3.7 Sampling Procedure

Following selection of the testing area the wind tunnel trailer was parked perpendicular to the site. The transition section of the tunnel along with the fan and motor assembly were mounted on a simple swivel and hinge system to allow the test unit to be swung off the bed of the trailer and lowered into position on the sampling site. Small height adjustments to account for variations in ground height are made using a hydraulic jack system mounted on the trailer drawbar. These height adjustments also ensure that a good seal was achieved between the fan/motor section and the transition section.

Following the lowering of the transition section into position the working sections of the tunnel were moved into place. Adjustments to the height of the working sections of the tunnel were made as necessary (to ensure a good seal with the preceding sections). For crusted surfaces the sections were placed carefully to ensure the crusted surface was not broken. Soil from the surrounding area was placed on the neoprene flaps attached to the bottom of the working sections to ensure a seal was maintained with the ground and also to prevent excess dust generation inside the tunnel by any localised cross winds.

The sampling apparatus was assembled inside the tunnel at a position 1 m from its open end. All sampling was completed within 0.5 m of the centre-line of the tunnel to ensure aerodynamic interference by the side walls was minimised and to remain within the validated flow area of the wind tunnel. All sampling lines exit the end of the tunnel at ground level to further reduce any pressure impacts on the air flow at the sampling position.

The internal wind speed within the wind tunnel was set according to the revolutions per minute (RPM) of the fan using a revolution counter placed on the fan shaft. Use of this type of speed monitoring device allows for any slip present in the belts driving the fan to be accounted for. Fan speed was monitored continuously throughout the testing period to ensure that speed variation was minimised. The fan RPM rates for each wind speed were determined during the calibrations of the wind tunnel prior to each test phase.

Each test was run for 90 minutes with the samples taken at two heights: 0.3 m (the top of the saltation range) and 0.75 m (the centre of the remaining vertical testing zone). Throughout testing the fan speed was checked every 15 minutes to ensure that consistency of wind speeds was maintained. At the same interval, checks of sampling flow rates were made to ensure back pressure did not increase significantly due to filter loading.

At the completion of a test run the sampling apparatus was removed from the tunnel and the sampled filters collected for analysis. Prior to stopping the fan, the rotational speed for the next test was set.

3.3.8 Sample Analysis Methodology

As described in Section 3.2.3 for the Phase 1 research, a gravimetric analysis approach was adopted for the filter analysis for this second research component. This involved use of an analytical balance sensitive to 1 μg. The balance was positioned in an air-conditioned laboratory at Griffith University, and within a physical enclosure to buffer temperature and humidity variation. A simple humidity stabilisation mechanism (potassium permanganate crystals) was utilised within the enclosure to assist in humidity control. Continuous temperature and humidity readings were recorded throughout all gravimetric analysis procedures. Using this approach humidity was maintained to 50 % +/- 1% within the enclosure, well within the range of variability allowed by reference methods for gravimetric analysis (Australia, 2006).

3.3.9 Quality Assurance

During the sample phase, the critical component of the sampling related to the air flow rates for the sampling devices. This was important for two reasons. Firstly, accuracy of flow rates is necessary to allow calculation of resultant sample concentrations. Secondly, achieving the correct flow rates is essential in ensuring isokinetic flows were achieved for the TSP samples, and the appropriate flow rate for the cyclone in the case of the PM_{10} sampling. For the isokinetic sampling train, flow rates were measured for each sample using a calibrated dry gas meter. The Minivol samplers were calibrated using a portable rotameter.

For the analytical phase, quality assurance was maintained throughout the gravimetric analysis process in accordance with the requirements of *Australian Standard 3580.9.9 - Methods for sampling and analysis of ambient air - Determination of suspended particulate matter - PM_{10} low volume sampler - Gravimetric method (Australia, 2006)*. This involved maintaining the appropriate laboratory environment conditions throughout the filter conditioning and weighing phases, both prior to and following field sampling.

3.4 PHASE 3 – $PM_{2.5}$ EMISSION RATES

3.4.1 Overview

A range of methods are available for the estimation of emissions of particulates: mass balance calculations; engineering calculations; sampling or direct measurements; emission factors; and alternative (approved) techniques (Environment Australia, 2015; US EPA, 2013a). For this research, a direct measurement approach was adopted. Completing direct sampling at an open cut coal mine is challenging

due to the variability (temporal and spatial) of the fugitive sources that are encountered in open cut coal mines. Further constraints that applied to this investigation were the absence of mains power in the vicinity of the activities to be sampled, a requirement to maintain a safe separation distance from the mine activity, and the changing nature of the mine activities which necessitated relocating the sampling instrumentation on a daily basis, and, on occasion, during the same sampling day.

In light of these constraints, a downwind sampling method was adopted as the most practicable solution. This method is well documented (Axetell & Cowherd, 1981; Bieringer, Young et al, 2017; Frankell, 1993; Jia, Al-Ansari et al, 2013; Sastry, Chandar et al, 2015) and relies on completion of a series of measurements along a downwind transect, with subsequent analysis using Gaussian techniques to calculate the emission rate (NERDCC, 1988; Smith, 1995), based on the concentration and associated dispersion parameters measured at the time.

The methodology adopted for the particulate source emission testing in this study was based on a previous study completed in the Hunter Valley by the former State Pollution Control Commission (SPCC), NSW Coal Association and the National Energy Research, Development and Demonstration Council (NERDDC, Australia) (NERDCC, 1988). The approach adopted for this study was consistent with the upwind – downwind emissions estimation methodologies published by the US EPA (Axetell & Cowherd, 1981), and adopted for development of emission factors for a range of fugitive particulate sources.

The adopted approach is best described as parametric source testing and is consistent with an emission factor rating of A or B based on the US EPA rating scheme (refer to Figure 2-1). To further improve the data quality, analysis has been completed for both background and non-background corrected data points using simple statistical methods. In addition, the following adjustments have been applied to the dataset, using the methodologies drafted by the US EPA (US EPA, 2013b):

- (parametric) adjustment of data points below the method detection limit (BDL); and
- removal of outliers.

3.4.2 Sampling Methods

3.4.2.1 Introduction

The measurement techniques adopted for the sampling component of this research are summarised in Table 3-2, and described in the following sections. The sampling instrumentation was selected on the basis of published standards and guidelines.

Table 3-2: Sampling Techniques

Measurement Parameter	Instrumentation/Technique	Details
$PM_{2.5}$ at 4 positions on a downwind transect from the emission source	Met-One E-BAM, fitted with US EPA approved reference $PM_{2.5}$ cyclone separator	Data recorded for 1-minute averages, validated at the end of each hour to provide hourly data. Data resolution: 1 µg m^{-3} detection threshold: < 6 µg m^{-3}, accuracy: +/- 10 %. (Standards Australia, 2013)
$PM_{2.5}$ background concentrations at an upwind position	TSI DustTrak model 8530	Data resolution: 1 µg m^{-3} Particle size range: 0.1 to 10 µm, fitted with $PM_{2.5}$ impactor for sampling. Flow accuracy: +/- 1 % (Standards Australia, 2015).
Wind speed and direction.(US EPA, 2000)	Davis weather station for initial stage of sampling. Met-One 034B windset coupled with a Campbell Scientific CR800 data logger (US EPA, 2000)	Davis weather station: 1-minute data resolution. Met-One 034B: 1 second data resolution.
Soil moisture and silt content.	Calibrated laboratory oven and soil sieve set (Standards Australia, 2009)	Gravimetric analysis method adopted for determination of soil moisture. Manual sieve method adopted for determining silt fraction.
Flow Calibration	Bios Defender 510 DryCal	Used for flow calibration for the E-BAM instruments.

3.4.2.2 Particulates

3.4.2.2.1 Beta Attenuation Sampler

The extended sampling time issues associated with the gravimetric sampling technique used for particulate sampling in previous studies was overcome by adopting a portable Beta Attenuation (BAM) sampling method. BAM sampling is an approved Federal Reference Method in the United States. The portable BAM method adopted for the study is broadly consistent with the measurement methodology described in Australian Standard method AS 3581.9.11: 2008, Methods for sampling and analysis of ambient air Method 9.12: Determination of suspended particulate matter - $PM_{2.5}$ beta attenuation monitors (Standards Australia, 2013).

Portable E-BAM samplers (Met One Instruments Inc, 2008) were utilised in the study. The E-BAM instrument has a data resolution of 1 µg m^{-3}, a detection threshold of < 6 µg m^{-3} and an overall accuracy of +/- 10 %. This was considered acceptable for the purposes of the study, as these data quality parameters conform with the requirements of AS/NZS 3580.9.12 (Standards Australia, 2013). The optional sharp cut cyclone size selective inlet was utilised for separation of the $PM_{2.5}$ particulates from the sample airstream. This cyclone is the only model approved by the US EPA for regulatory compliance (US EPA, 2016a).

The E-BAM automatically measures and records airborne $PM_{2.5}$ particulate concentrations in near real time using the principle of beta ray attenuation. This method provides a simple determination of concentration in units of milligrams of particulate per cubic meter of air. A small ^{14}C (Carbon 14) element emits a constant source of high-energy electrons known as beta particles. These beta particles are

detected and counted by a sensitive scintillation detector. A vacuum pump pulls a measured amount of dust-laden air through the filter tape, which is positioned between the source and the detector thereby causing an attenuation of the beta particle signal. The degree of attenuation of the beta particle signal is used to determine the mass concentration of particulate matter on the filter tape, and the volumetric concentration of particulate matter in ambient air.

3.4.2.2.2 Mathematical Basis for BAM Sampling

The BAM sampling method relies on high-energy electrons emanating from the radioactive decay of 14C (carbon-14) interacting with nearby matter such that they lose their energy and, in some cases, are absorbed by this matter. These high-energy electrons emitted through radioactive decay are known as beta rays and the process is known as beta-ray attenuation. When matter is placed between the radioactive 14C source and a device designed to detect beta rays, the beta rays are absorbed and/or their energy diminished. This results in a reduction in the number of beta particles detected. The magnitude of the reduction in detected beta particles is a function of the mass of the absorbing matter between the 14C beta source and the detector.

The mathematical basis for the calculation of particulate mass concentration using the BAM technique is described by the manufacturer, Met One (Met One Instruments Inc, 2008), as follows.

The number of beta particles passing through absorbing matter, such as dust deposited on a filter tape, decrease nearly exponentially with the mass through which they much pass. Equation 1 shows this relationship.

$$I = I_0 e^{-\mu x} \quad \text{(Equation 1)}$$

In Equation 1, I is the measured beta ray intensity (counts per unit time), of the attenuated beta ray (dust laden filter tape), I_0 is the measured beta ray intensity of the un-attenuated beta ray (clean filter tape), μ is the absorption cross section of the material absorbing the beta rays ($cm^{2/}g$), and x is the mass density of the absorbing matter (g/cm^2).

Equation 1 very closely resembles the Lambert-Beers Law, which is used in spectrometric analysis. Just as the Lambert-Beers Law is an idealisation of what is actually observed, it is also an idealised simplification of the true processes occurring meant to simplify the corresponding mathematics. However, experimental measurement shows that in properly designed monitors the use of this equation introduces no substantial error.

Equation 1 may be rearranged to solve for x, the mass density of the absorbing matter:

$$-\frac{1}{\mu} \ln\left[\frac{I}{I_0}\right] = \frac{1}{\mu} \ln\left[\frac{I_0}{I}\right] = x \quad \text{(Equation 2)}$$

The absorption cross section is experimentally determined during the calibration process. Once I and I_0 are experimentally measured the predicted mass density, x, can be calculated.

In practice, ambient air is sampled at a constant flow rate (Q) for a specified time Δt. This sampled air is passed through a filter of surface area A. Once x, the mass density of collected particles, has been determined, it is possible to calculate the ambient concentration of particulate matter ($\mu g/m^3$) as follows:

$$c\left(\frac{\mu g}{m^3}\right) = \frac{10^6 \, A(cm^2)}{Q\left(\frac{liter}{min}\right) \Delta t(min) \mu \left(\frac{cm^2}{g}\right)} \quad \text{(Equation 3)}$$

In Equation 3, c is the ambient particulate concentration (μg/m^3), A is the cross-sectional area on the tape over which dust is being deposited (cm^2), Q is the rate at which particulate matter is being collected on the filter tape (liters/minute), and Δt is the sampling time (minutes). Combining these equations yields to the final expression for the ambient particulate concentration in terms of measured quantities. This is shown in Equation 4.

$$c\left(\frac{\mu g}{m^3}\right) = \frac{10^6\, A(cm^2)}{Q\left(\frac{liter}{min}\right)\Delta t(min)\mu\left(\frac{cm^2}{g}\right)} \ln\left(\frac{I_0}{I}\right) \quad \text{(Equation 4)}$$

The absorption cross-section of the beta attenuation monitor, μ, is almost insensitive to the nature of the matter being measured. This makes the instrument very insensitive to the chemical composition of the material being collected.

3.4.2.2.3 Sampler Operation

A full calibration of each of the instruments was completed prior to shipping the samplers to the test sites. At the start of each test, once the E-BAMs were in the transect location, a leak check and flow check was performed. This was repeated upon the completion of each four-hour measurement sequence. A leak check value of 0.6 litres per minute or less and a flow calibration within 0.6 litres per minute of the required flow rate were considered pass values as recommended by Met One. If either of these tests failed at the completion of the measurement, the measurement data for that period were considered to be invalid. The nozzle on each instrument was visually inspected before and after each measurement to ensure there was no particulate residue attached to the nozzle that could affect the test results.

The E-BAM sampling was completed under ambient temperature and humidity. For beta attenuation sampling, it has been demonstrated that there is potential for relative humidity in excess of 60 – 80 % to result in over-estimation of particulate concentrations relative to gravimetric sampling methods, particularly for temperatures below 16°C (Chang, Tsai et al, 2001; Takahashi, Minoura et al, 2008; Triantafyllou, Diapouli et al, 2016). The influence of relative humidity was minimised through use of a heated inlet on the sampling device, in accordance with the manufacturer's recommendations. Temperature data was recorded throughout the sampling to allow adjustment of sampled air volumes to standard conditions.

The E-BAM sampler provides hourly data as well as a near real-time average for a pre-defined period. The hourly concentration measurement is based on two four-minute long beta counts at the beginning and end of each 60-minute measurement cycle. The first measurement establishes a zero measurement and the final reading at the end of the one-hour sampling period gives a span value, with the difference between the two measurements providing the average concentration across the hourly measurement period. The E-BAM samplers were set to move the tape forward every hour, ensuring a new section of filter tape was used for each hour of measurement. Data was obtained at an interval of one minute. This allowed near to real-time data for each measurement sequence.

It is noted that the hourly average measurement is not a statistical average of the individual one-minute measurements. The hourly average is based on the difference between the initial and final beta measurement across the entire one-hour measurement period, hence is a more accurate determination of the average particulate concentrations than a statistical average of the individual real time measurements.

There are also periods where the E-BAM can give negative readings if the actual particulate concentration is very low. This is due to the small amount of leakage which is present in the system and the random noise of the instrument. Therefore, the real-time measurements may not be an accurate representation of the particulate concentration at any given point but may be useful in providing an indication of the short-term upper range of the measurements as well as any trends in the data.

3.4.2.2.4 Background Sampling

Particulate sampling was also completed at an upwind location to allow correction of the emission data for ambient background. TSI Dustraks (real time optical based particulate sampling devices) were used for this purpose. As this sampling method differs significantly for the beta attenuation method, a second optical sampler was co-located with the E-BAM at the first transect position for a number of samples at each mine, to allow a correction coefficient to be determined. The correction coefficient was applied to adjust the optical sampler data to an E-BAM equivalent concentration. To confirm the validity of the method adopted for determining the calibration coefficient, additional testing was completed using collocated optical and E-BAM samplers at the background location and also at transect position 1 for the same test. The data from this additional testing resulted in the same average correction coefficient for the background position and transect 1 position, thus confirming the suitability of the adopted approach for correcting the optical data.

3.4.2.2.5 Instrument Calibration

Prior to commencement of sampling at each location, a series of quality assurance checks were completed to ensure the accuracy of the E-BAM data in accordance with the manufacturer's recommendations. These involved an air leak check, flow rate calibration, particulate concentration zero and span check and an instrument self-check in accordance with the documented manufacturer's procedure. The Dustrak optical particulate samplers were calibrated by an external approved laboratory prior to the sampling, and field checks of zero and flow were completed for each sampling phase to confirm the on-going calibration status of the instruments.

3.4.2.2.6 Wind Speed and Direction

As the method requires that the sampling transect is downwind of the plume emitted from the activity of interest, the wind speed and direction were measured prior to commencement of sampling at each location to allow selection of a downwind sampling transect position. The wind direction was subsequently monitored throughout the sampling to allow adjustment of the transect position where changes in wind direction resulted in the transect being >45° from the transect alignment. A typical wind rose for a transect sampling period is presented in Figure 3-4. The wind speed and direction measurements were completed at the sample position closest to the emission source.

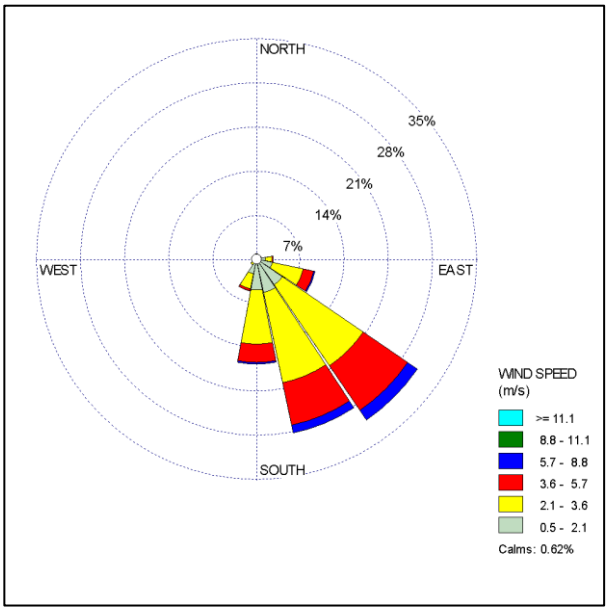


Figure 3-4: Example Windrose from Sampling Transect

3.4.2.3 Plume Height and Width

An estimation of initial plume height and horizontal spread was also completed for each sampling location, based on direct observation of the visible particulate plume and review of video footage of the sampling period. The plume height and width were estimated by the sampling team based on these visual observations. The largest plume heights and widths were measured for the dragline activities, with an average width of 133 m and height of 58 m observed immediately downwind of the source. This is to be expected given the size and height of a dragline.

For point sources, the typical initial plume height was 6 m, and the initial plume width an average of 7.5 m. Narrower widths were typically observed where site features constrained the plume size, for example proximity to earth berms.

For line sources only the plume height is relevant, as the source width is continuous. The plume height was generally defined by the turbulent wake to the rear of the vehicle. As mine haul vehicles were generally a similar size, the plume heights for the various haul activities were very similar (average height of 5 m).

3.4.2.4 Soil Samples

For each test, a sample of the material or surface from which dust was being emitted was collected. This was undertaken during a break in operations or after sampling was completed. Samples were sealed in plastic bags and returned to the laboratory. Each sample was then, allowed to dry in an oven for a period of at least 24 hours and then reweighed. The weight loss was determined as the amount of moisture present in the sample (Standards Australia, 2009). Moisture content of the material was then determined. For the coal samples that were taken as part of the measurement of coal loading and coal dumping activities, the drying period was limited to 6 hours to minimise the risk of combustion given the significant proportion of coal in the samples.

The silt content of the samples was determined from the oven dried material using a dry sieving method. Samples were passed through a nest of sieves ranging from 1,000 µm to 75 µm. The silt content for each sample was determined as the <75 µm soil fraction (Standards Australia, 1993).

3.4.2.5 Ancillary data

Additional data collected during the sampling included plant and equipment types; material throughputs; duration of each activity cycle; number of vehicle movements; and particulate control methods adopted (e.g. haul route watering). These data were required as inputs to the subsequent calculation of emission rates specific to vehicle movement rates or unit of material throughput.

3.4.3 Data Analysis Methodology

3.4.3.1 Introduction

The empirical data obtained during the field trials was used as the basis for determining emission factors for the activities considered. A Gaussian approach was adopted based on the methodology for development of a number of the current emission factors utilised in the Australian National Pollutant Inventory Emission Estimation methods (Environment Australia, 2012b; NERDCC, 1988). Emission factors were determined separately for line and point sources using simplified Gaussian dispersion equations.

3.4.3.2 Calculation of Line Source Emissions

Due to the nature of line sources, the spread of the plume is calculated in the vertical direction only as the whole plume will move across the downwind transect in the horizontal direction. The vertical plume spread parameter incorporates a virtual distance term which accounts for the initial size of plume in the vertical axis. Firstly, the vertical height of the plume was estimated during field observations. This height is divided by 2.15 (Turner, 1970) to provide the σ_{z0} parameter.

The virtual distance (x_0) is then defined as:

$$x_0 = (\sigma_{z0})^{1/b} \quad \text{(Equation 5)}$$

Where,

b = dimensionless empirical parameter; and

σ_{z0} = initial vertical plume spread parameter (m)

The vertical plume spread parameter (σ_z) is a function of the stability class and the downwind distance of the sampler, defined as follows:

$$\sigma_z = a(x + x_0)^b \quad \text{(Equation 6)}$$

where,

a = dimensionless empirical parameter;

b = dimensionless empirical parameter;

x = downwind distance (m); and

x_0 = virtual distance (m) as described above.

The two empirical parameters based on the stability class are presented in Table 3-3.

The emission rate q in $gs^{-1}m^{-1}$ is defined as follows:

$$q = \frac{1}{2}\chi sin(\theta)\left(\sqrt{2\pi}\right)\sigma_z u \quad \text{(Equation 7)}$$

where,

χ = plume centreline concentration at distance x downwind from the source (g/m^3);

θ = angle between wind direction and line source (degrees);

σ_z = vertical plume spread parameter (m); and

u = mean wind speed (m/s).

Table 3-3: Empirical Parameters

Stability Class[a]	Description[a]	a	b
A	Very unstable	0.180	0.945
B	Unstable	0.145	0.932
C	Slightly unstable	0.110	0.915
D	Neutral	0.085	0.870

Source: Zimmerman, J R and Thompson, R S National Environmental Research Centre, Users Guide for HIWAY. A Highway Air Pollution Model, 1975.
[a] Stability class and descriptions as per Pasquill, F, The estimation of the dispersion of windborne material. Meteorological Magazine. 90: 33–49. February 1961.
Note: Night time stability classes E and F are not considered, as all sampling was completed during the daytime.

3.4.3.3 Calculation of Point Source Emissions

As described above, several assumptions can be made about the location of the plume centreline for line sources; these result in the simplification of the line source equation. This is not the case for point sources, as the transect could be located off the plume centreline in both the vertical and horizontal directions due to varying winds and emission heights. Therefore, two adjustment factors for the horizontal and vertical axes are introduced. These factors are functions of both the effective distance between the sampler and the plume centreline, and the plume spread parameter for the relevant axis.

The vertical plume spread parameter (σ_z) is the same as defined for the sources while the horizontal plume spread parameter is defined as

$$\sigma_y = \frac{\sigma_\theta x}{57.3} + \sigma_{yo} \quad \text{(Equation 8)}$$

where,

σ_θ = sigma theta (standard deviation of mean wind direction);

x = source to sampler distance (m); and

σ_{y0} = observed plume width at the source divided by 4.30 m ((Turner, 1970)).

In order to calculate the distance from the plume centreline to the sampler in the horizontal direction (y), the resultant wind direction relative to the samplers is determined and the distance is then calculated by trigonometry. The distance from the plume centreline to the sampler in the vertical direction (z) is defined as the difference between the height of the emission point and the height of the samplers. The reduction factors in the horizontal and vertical direction (R_y and R_z respectively) are defined as:

$$R_y = exp\left(-0.5\frac{y^2}{\sigma_y{}^2}\right) and R_z = exp\left(-0.5\frac{z^2}{\sigma_z{}^2}\right) \text{ (Equation 9)}$$

Finally, the emission rate (Q) for a point source in gs^{-1} is defined as follows:

$$Q = \frac{\chi\pi\sigma_y\sigma_z u}{R_y R_z} \text{ (Equation 10)}$$

The Pasquill-Gifford stability class during the measurement was determined by the Sigma-Theta method described by Slade (Slade, 1968). This involves observing each 20-minute interval of the meteorological data during the measurement period and determining the most extreme wind direction in degrees during each interval.

The Sigma-Theta value is then determined by:

$$\sigma_\theta = \frac{\Delta\theta_{max}}{6} \text{ (Equation 11)}$$

where, $\Delta\theta_{max}$ = the extreme range of wind direction over a 20-minute interval.

The stability class is then determined from the value of σ_θ from Equation 11. The stability class boundary values have been adjusted for a measurement height of 2 m by the following method (US EPA, 2000; Yamartino, 1984):

$$(Z/10)^{(P_\theta)} \text{ (Equation 12)}$$

where, Z = measurement height in metres and P_θ is determined from Table 3-4

Table 3-4: Parameters for Determining Atmospheric Stability Class - Sigma Theta Method

σ_θ	Stability Class	P_θ
$\sigma_\theta > 24.8°$	A	-0.06
$22.3° < \sigma_\theta < 24.8°$	B	-0.15
$16.4° < \sigma_\theta < 22.3°$	C	-0.17
$10.9° < \sigma_\theta < 16.4°$	D	-0.23
$\Sigma_\theta < 10.9°$	E	-0.38

Source: Turner D B, 1970

3.4.3.4 Calculation of Source Specific Emission Rates

The emission rates were determined in terms of g/s for point sources and g $s^{-1}m^{-1}$ for line sources. These emission rates were further analysed to determine emission rates specific to the tested activity, using the

activity rates for each test as recorded at the time of sampling. Emission rates from haul roads were expressed in kilograms per vehicle kilometre travelled (VKT). For point sources, throughput specific emission rates were determined by dividing the emission rate in grams by the unit throughput, such as the number of holes drilled, or the quantity of material loaded or dumped.

3.4.4 Data Quality

Average emissions differ significantly from source to source. The extent of variability that exists, even among similar individual sources, can be large depending on the process, control system, and pollutant. Even when the major process variables are accounted for, the emission factors developed may be the result of averaging source tests that differ by factors of five or more (US EPA, 1995b). This feature is exhibited in the dataset that formed the basis for many of the PM_{10} emission factors adopted in the current National Pollutant Inventory Mining Handbook (Environment Australia, 2012b). Review of the SPCC report (NERDCC, 1988) that provided these data confirms that the typical standard deviation for the dataset was up to 2.9 times the average emission factor. This illustrates the variability that is present amongst emissions of this type for currently adopted emission factors.

The existing NPI emission estimation techniques developed for Australian mines adopt size fractionation percentages derived from studies completed in the Hunter Valley, New South Wales in 1986 (SPCC, 1986). These size fractions were determined from samples collected in operating mines, with the samples subsequently analysed using a Malvern Mastersizer. The Malvern Mastersizer is an optical based method for determining particle size, hence provides data in terms of mean geometric diameter as opposed to aerodynamic diameter. The SPCC study confirms that optical microscope evaluation of a selection of dust samples was completed to confirm that the aspect ratios of the particles remained less than 3:1, indicating that the size fractions provided a reasonable estimate of aerodynamic diameter.

3.4.5 Particulate Bound Moisture

The results of the temperature and humidity sampling completed for each test confirm that the particulate samples were collected at temperatures > 16°C and the maximum average relative humidity for individual sampling periods was 55.1 %. On this basis, the potential for over-estimation of particulate concentrations due to particulate bound water is considered to be negligible (Takahashi, Minoura et al, 2008). Water was applied to the road surface during sampling as a standard operating practice for management of dust, hence the potential for increased particulate bound moisture to be present for these sources, despite the absence of a significant influence from relative humidity, has been considered. Takahashi et al indicate that the differences in particulate concentrations measured by beta-attenuation sampling relative to gravimetric sampling methods is not significant at estimated particle moisture concentrations of 12 % or less. The maximum measured moisture content for all samples collected in this study was 6 %, therefore particulate bound moisture is well within the threshold estimated by Takahashi as resulting in over estimation of particulate concentrations. Therefore, no adjustment of the measured concentrations to account for particle bound moisture was necessary.

3.5 RESEARCH RIGOUR

Practical components of the research have been completed in accordance with published peer reviewed methodologies and standard procedures. A range of quality assurance measures have been adopted. For field work and laboratory analysis these included pre and post calibration of field instrumentation, provision of field and laboratory blanks, temperature and humidity controlled analytical environments and selection of appropriate instrumentation and methodologies.

All data analysis has been subject to a quality assurance review of calculation methodologies and the overall results, to ensure accuracy and validity of the datasets.

The guidance of regulatory agencies such as the US EPA in relation to developing a high-quality emission dataset has been adopted in completing this research. In particular, the need for extended sampling under representative meteorological conditions has been considered in selecting the appropriate time of year for sampling, and suitable sampling time periods.

3.6 Ethical Issues

No animal or human subjects are involved in the research that has been completed, and no ethical issues have been identified as an issue in completing this research.

3.7 Copyright and Prior Publication

The research presented for the purposes of this PhD programme has not previously been published in international peer reviewed academic journals. Research reports relating to the individual research components presented in Chapters 4 to 6 were prepared for ACARP and are available on request.

The copyright for all publications prepared for this PhD programme remains with the Author and, where relevant, the journals that have published these papers. Permission to reproduce these publications has been granted as follows:

- Appendix A: Characterisation of Particulate Emissions from Australian Open Cut Coal Mines: Towards Improved Emissions Estimates. Richardson C, Rutherford S, et al. *Journal of Air and Waste Management Association*, 68 (2018), 6, 598 – 607.
  Permission to include the Author's Original Manuscript in this thesis was granted by Taylor & Francis on 13 March 2019 (reference: P031319-03/UAVM).

- Appendix B: Wind Speed Dependent Particulate Emission Rates for Open Surfaces in Open Cut Black Coal Mines. Richardson C, Rutherford S, et al. *Journal of Environmental Management*, 232 (2019) 537 – 544.
  Elsevier grant the author the right to include the published article in a thesis or dissertation, as detailed on the Elsevier website at https://www.elsevier.com/about/our-business/policies/copyright#Author-rights and as confirmed by the online Copyright Clearance Centre Rightslink Service at https://s100.copyright.com.

- Appendix C: Open Cut Black Coal Mines: Empirical Verification of $PM_{2\,5}$ Emission Estimation Techniques. Richardson C, Rutherford S, et al. *Journal of Atmospheric Research* 216 (2019) 151 - 159.
  Elsevier grant the author the right to include the published article in a thesis or dissertation, as detailed on the Elsevier website at https://www.elsevier.com/about/our-business/policies/copyright#Author-rights and as confirmed by the online Copyright Clearance Centre Rightslink Service at https://s100.copyright.com.

4 KEY FACTORS INFLUENCING PARTICLE CHARACTER IN AUSTRALIAN OPEN CUT BLACK COAL MINES

4.1 INTRODUCTION

Chapter 2 provides a review of the current literature relating to particulate emission estimation for Australian open cut black coal mines. This demonstrates that there is limited empirical data relating to fine particulate emissions, and the current emission estimation techniques are largely based on TSP emission research completed in coal mines in the United States and Australian over 25 years ago.

As described in Section 0, the initial phase of research for this PhD programme involved a broad investigation of particulate size distributions and compositional analysis. Sampling was completed at 1 mine in Queensland (QLD mine 1) and two mines in New South Wales (NSW mines 1 and 2).

This investigation involved two distinct study elements. Firstly, short-term real-time sampling was completed downwind of a range of coal mine sources to confirm particle size distributions and to gauge the relative significance of different sources in terms of particulate emissions. This sampling involved use of real time light-scattering particulate analysers. Secondly, longer term sampling was completed both within and at the boundary of coal mines to provide data relating to particulate size distributions.

The results of this research are described in the following sections.

4.2 SOURCE SAMPLING RESULTS

The results of the particulate source sampling at each of the three mines are presented in Table 4-1 to Table 4-3.

The size fractions (Table 4-1 to Table 4-3*)* also show significant variability for the different mine activities, with a maximum PM_2 proportion of 11 % and 61 % for PM_{10} recorded for the coal preparation plant at NSW Mine 2. The highest PM_{10} proportions were measured downwind of coal preparation plant (66 %) and drilling (63 %). This demonstrates that adoption of an average particulate size fractionation for all mining activities could significantly under estimate PM_{10} emissions from specific activities, as some operations generate higher than average PM_{10} fractions.

Table 4-1: Source Emission Data – Mine 1 (QLD)

			Average Concentration (μg m^{-3})			Particle Size Distribution (%)		
Ref. No	**Sample Period (minutes)**	**Particulate Source and Approximate Distance**	**TSP**	**PM_{10}**	**PM_2[1]**	**TSP**	**PM_{10}**	**PM_2[1]**
1	18	Product Stockpile (120 m)	53	34	2	100	64	4
2	18	Coal Loading and Haul (50 m)	2,614	1,451	43	100	56	2
3	12	Topsoil Spreading (80 – 150 m)	1,176	585	7	100	50	< 1
4	15	Dragline on Overburden (50 m)	6,015	170	3	100	3	< 1
5	14	Pre-strip (60 m)	5,970	1,681	12	100	28	< 1
6	18	Coal Preparation Plant (Product Stockpile at 120 m)	70	46	3	100	66	4
7	8	Coal Preparation Plant (Product Stockpile at 30 m)	256	94	4	100	37	2
8	18	Coal Preparation Plant (Product Stockpile at 60 m)	124	71	4	100	57	3

[1] The monitoring instrument measured $PM_{2\,0}$ rather than $PM_{2\,5}$.

Table 4-2: Source Emission Data – Mine 1 (NSW)

			Average Concentration (µg m^{-3})			Particle Size Distribution (%)		
Ref. No.	**Sample Period (Minutes)**	**Particulate Source & Approximate Distance**	**TSP**	**PM_{10}**	**PM_2[1]**	**TSP**	**PM_{10}**	**PM_2[1]**
1	68	Coal Haul (11 m road edge)	728	204	12	100	28	2
2	8	Coal Haul (2 m road edge)	3,849	1,282	71	100	3	2
3	26	Dragline (on board)	3,084	707	69	100	23	2
4	16	Dumping into Coal Hopper (10 –15 m)	2,254	1,002	78	100	45	4
5	9	Drill (5 m)	52,952	33,445	2,334	100	63	4
6	28	Coal loading (10 m) Overburden loading (30 m)	4,108	1,671	103	100	41	3
7	22	Coal Haul (2 m road edge)	4,934	1,916	117	100	39	2
8	7	Spontaneous Combustion (20 m)	1,200	391	26	100	33	2

[1] The monitoring instrument measured $PM_{2\,0}$ rather than $PM_{2\,5}$.

Table 4-3: Source Emission Data – Mine 2 (NSW)

			Average Concentration (μg m^{-3})			**Particle Size Distribution (%)**		
Ref. No.	**Sample Period (Minutes)**	Particulate Source & Approximate Distance	TSP	PM_{10}	PM_2[1]	TSP	PM_{10}	PM_2[1]
1	58	Coal Loading (50 m)	3,098	1,444	83	100	47	3
2	31	Shovel (on board)	1,509	457	38	100	30	3
3	21	Drill (on board)	32,177	9,291	535	100	29	2
4	14	Dozer (50 m)	1,583	448	31	100	28	
5	33	Coal Dumping (2 m)	2,501	924	92	100	37	
6	32	Coal preparation plant (5 m from hopper)	649	394	70	100	61	11
7	7	Coal Haul (5 m)	1,200	391	26	100	33	2

[1] The monitoring instrument measured $PM_{2\,0}$ rather than $PM_{2\,5}$.

Average particle size fractions for each mine are presented in Table 4-4 indicating that the average particle size fractions for the two NSW coal mines are almost identical. The average particle size fraction for the QLD mine indicates a higher proportion of PM_{10} material overall (42 %), however the PM_2 size fraction of 2 % is similar to that observed for the NSW mines. This suggests that regional geological conditions or other local features may be influencing the particulate size fraction for these mining activities. The overall average PM_{10} to TSP mass fraction for all samples of 40 % is consistent with a mass fraction of 36 % PM_{10} to TSP as reported for mines in Columbia (Huertas et al, 2012).

Table 4-4: Average Particle Size Distribution – Source Sampling

Data Source	**Particle Size Distribution %**		
	TSP	**PM_{10}/TSP**	**PM_2/TSP**
Mine 1 (QLD)	100	45	2
Mine 3 (NSW)	100	38	2
Mine 2 (NSW)	100	38	3
Average	**100**	**40**	**3**

4.3 ENVIRONMENTAL SAMPLING RESULTS

Table 4-5 presents the results of the sampling completed external to the boundary of the two mines in NSW. The estimated separation distance to the nearest active pit at each mine is also identified. These results demonstrate that a markedly different particle size fractionation occurs external to the mine boundary relative to the near source sampling data (Table 4-4). For the sampling completed external to the mine, TSP is comprised of a higher proportion of PM_{10} at an average of 55 % compared to an average of 40 % observed for the near source data. Of greater significance is the proportion of $PM_{2\,5}$, at 25 % of TSP, compared to an average of 2 % for the near source PM_2 measurements. It is likely that, because finer particulates are transported over long distances from the source, the relative proportions of fine dust tend to increase with distance from a specific source(Cattle, Hemi et al, 2012). This demonstrates that mine particulates pose a greater risk in a regional setting that is suggested by the size fractionation of the source emissions.

Table 4-5: Average Particle Size Distribution External to Mine Boundary

Data Source	**Particle Size Distribution %**		
	TSP	**PM_{10}/TSP**	**$PM_{2.5}$/TSP**
Mine 3 (1 km to active pit)	100	66	21
Mine 2 (3 km to active pit)	100	50	24
Mine 2 (4 km to active pit)	100	48	28
Average	**100**	**55**	**24**

Table 4-6 presents a comparison of the proportion of PM_{10}/TSP for different activities at each of the mines, and also presents the size fractions recommended by the US EPA and for the Australian NPI emissions equations. This comparison demonstrates that there is reasonable consistency between the NPI size fractions and the measurement data for the various mine activities considered in the sa mpling. US EPA size fraction factors are available for only two of the sources considered – draglines and bulldozers. In both cases, the US EPA fractions for PM_{10} are more than double the fractions determined by the monitoring at the three mines, and also more than double the PM_{10} fractions indicated by the NPI emission factors.

Emission estimation factors for $PM_{2.5}$ are not provided in the 2012 NPI Mining handbook, as reporting of fugitive $PM_{2.5}$ emissions is not currently required. Therefore, Table 4-7 presents a comparison of the measured $PM_{2.5}$/TSP proportions with the $PM_{2.5}$ scaling factors defined by the US EPA for specific mining activities. This comparison indicates that the measured dataset provides a similar proportion of $PM_{2.5}$ to the US EPA scaling factors.

Table 4-6: Size Fraction (PM_{10}/TSP) Comparison

Activity	NPI[1]	US EPA[1]	QLD	NSW 1	NSW 2
Draglines (on overburden)	43	75	3	23	-
Excavators/Shovels/Front-end loaders (on overburden)	47	-	28	41	30
Excavators/Shovels/Front-end loaders (on coal)	48	-	56	41	47
Bulldozers on coal	32	75	-	-	28
Trucks (dumping coal)	42	-	-	45	37
Drilling	52	-	-	63	29

[1] Recommended size fraction published in NPI (Environment Australia, 2012b)and US EPA (US EPA, 1998a).

Table 4-7: Size Fraction Comparison ($PM_{2.5}$/TSP)

Activity	US EPA	QLD	NSW 1	NSW 2
Draglines (on overburden)	1.7	<1	2	-
Bulldozers on coal	2.2	-	-	2
Bulldozer on material other than coal	1.05	-	-	2
Graders	0.31	<1	-	-

Source: US EPA: United States Environmental Protection Agency (1995) AP42 Chapter 11, Section 09, Western Surface Coal Mining

4.4 COMPOSITIONAL ANALYSIS RESULTS

Results of the optical electron microscope and energy-dispersive x-ray spectrometry analysis are presented in Table 4-8 (particle size) and Table 4-9 (particle composition).

Table 4-8 demonstrates that particles with a size range in excess of the relevant fraction ($PM_{2.5}$ or PM_{10}) were observed on many of the filters. This feature is related to the efficiency of size selective particulate sampling inlets. Generally, size selective inlets are 50 % efficient in the size range of interest. Hence, for a $PM_{2.5}$ sample it is to be expected that a significant proportion of particulates exceed the cut-off point of 2.5 μm.

An example of the distribution of particulates on the glass fibre filters when viewed through the electron microscope is presented in Figure 4-1 for a $PM_{2.5}$ sample.

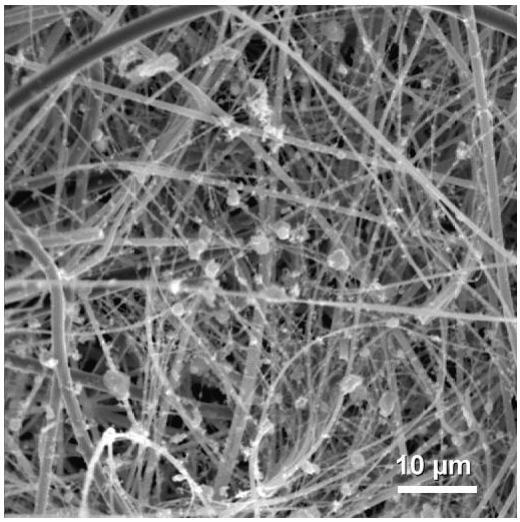


Figure 4-1: Electron Microscopy Analysis of Sample DH-26, $PM_{2.5}$

Table 4-8: Electron Microscopy Analysis of Particle Sizes

Reference Number/ Size Fraction Sampled	Monitoring Positions	All Particles (excluding Pollen)		Carbonaceous Particles	Pollen / Spores
		Size Range (number analysed)	Mean	Range (number analysed)	Number Identified (size range)
S1-01: $PM_{2.5}$	QLD Mine 1 – due west of active pit, 50 m to haul road	1 – 9 μm (17)	2.82 μm	8 - 9 μm (2)	4 (5 – 8 μm)
S2-05: $PM_{2.5}$	QLD Mine 1 – 550 m downwind of processing plant	1.2 – 11 μm (52)	2.93 μm	1 – 11 μm (16)	None found
S3-05: $PM_{2.5}$	QLD Mine 1 – background position	1 – 11 μm (59)	3.50 μm	2 – 11 μm (6)	4 (3 – 9μm)
S3 – 12: $PM_{2.5}$	QLD Mine 1 – 1 km downwind of processing plant	1.3 – 10 μm (41)	2.86 μm	1.7 – 10 μm (9)	2 (6 – 8 μm)
S3 – 19: PM_{10}	QLD Mine 1 – 1 km downwind of processing plant	1.2 – 15 μm (70)	3.68 μm	4 – 13 μm (4)	9 (4 – 14 μm)
H2 – 2: $PM_{2.5}$	NSW Mine 1 – approximately 4 km from active pit	1.0 – 10 μm (19)	2.74 μm	2 – 10 μm (5)	None found
DH – 25: PM_{10}	NSW Mine 2– Approximately 1 km to active pit	1.5 – 100[1] μm (75)	7.02 μm	4 – 100 μm (10)	9 (6 – 12 μm)
DH – 26: $PM_{2.5}$	NSW Mine 2 – Approximately 1 km to active pit	1 – 18 μm (39)	3.09 μm	8 - 13 μm (2)	None found

[1] The largest mineral particle was ~20 μm. The maximum size of 100 μm relates to a carbon particle that may have resulted from contamination of the filter during sample preparation.

Table 4-9: Electron Microscopy Analysis of Predominant Elements Present

Reference Number/ Size Fraction Sampled	Monitoring Positions	Number of particles (excl. pollen)	Percentage (%) of particles in each category				
			Clay or soil	Iron	Silica	Other salt or mineral[1]	Carbon
S1-01: $PM_{2.5}$	QLD Mine 1 – due west of active pit, 50 m to haul road	17	77	0	12	0	12
S2-05: $PM_{2.5}$	QLD Mine 1 – 550 m downwind of processing plant	52	54	0	12	4	31
S3-05: $PM_{2.5}$	QLD Mine 1 – background position	59	63	3	19	5	10
S3–12: $PM_{2.5}$	QLD Mine 1 – 1 km downwind of processing plant	41	51	2	15	10	22
S3–19: PM_{10}	QLD Mine 1 – 1 km downwind of processing plant	70	67	1	9	17	6
H2–2: $PM_{2.5}$	NSW Mine 1 – approximately 4 km from active pit	19	42	5	16	11	26
DH–25: PM_{10}	NSW Mine 2 – approximately 1 km to active pit	75	71	4	11	1	13
DH–26: $PM_{2.5}$	NSW Mine 2 – approximately 1 km to active pit	39	72	3	8	13	5

[1] This includes particles such as salt (NaCl), and compounds (salts), possibly of agricultural origin, which contained two or more of the elements Na, Mg, S, Cl, Ca, Fe and Zn.

Based on the results presented in Table 4-9, in terms of the source of particulates, on average 75 % of the measured particulate loading is from a soil/ clay source or comprised silica. These particulates could result from overburden sources, rehabilitation activities at the mine or erosion of open surfaces. Given the location of the samplers external to the mine, on farmland, there may also be a contribution from agricultural operations and erosion of farmland areas external to the mine. There is no clear difference between the particulate composition of the PM_{10} size fraction and the $PM_{2\,5}$. This is not unexpected, given that the majority of the particulates in these size ranges are likely to be sourced from local geological material – whether from mining activities or other local sources such as agricultural activities. As noted by Huertas, Huertas et al (2012b) due to the similarity in particulate emissions from mines and regional soils, it is not possible to determine the origin of particulate matter and hence the contribution of the mine to regional particulate concentrations. The relatively homogenous nature of the particulates, with 75 % overburden or soil related, indicates that the potential for significant differences in mass density is limited to <25 % of the sampled material.

An assessment of the likely coal content of the samples can be made from the percentage presence of carbon in the particulates. Although a range of other minerals, including silica, may be present in coal, these minerals may also be present in the local geologically sourced material. Hence, the percentage carbon in the results provides the most appropriate indicator of the likely significance of coal in the particulate samples. As would be expected, the highest percentage coal composition of 31 % (sample S2-05) was collected 550 m downwind of the processing plant at Mine 1. The average carbon composition for all samples was 16 %, thus indicating that coal comprises a relatively small proportion of the overall particulate composition. It is noted that the carbon fraction will contain material from a range of sources, including vegetation and insects, hence the percentage carbon represents the maximum possible proportion of coal in the sample.

Few pollen spores were identified, even for the background monitoring site (S3-05). This partly reflects the generally coarser size of pollen spores, but also confirms the greater significance of geological sources of material in the measured particulate emissions even at background locations.

4.5 DISCUSSION

The results of the sampling confirm that particulate size fractions vary for different mine activities and, to a lesser extent, may also differ between different geographical areas. Of greater significance, comparison of the measured particle size fractions confirms that the PM_{10} size fraction scaling factors documented by the US EPA in the AP 42 emissions estimation manual for surface coal mine are significantly different, and if applied in an Australian mining context would over-estimate PM_{10} emissions. This highlights the need for region specific emission inventories to be developed and is consistent with the observations of the US EPA regarding the potential for over estimation of PM_{10} emissions where the AP 42 factors are adopted (US EPA, 1998a). On this basis, it can be concluded that a degree of caution must be adopted when applying particulate emissions estimate methods derived on the basis of a specific region, to another region or overseas location.

Comparison of the measured PM_{10} emission fractions with the emissions estimations presented in the Australian NPI handbook for mining shows reasonable consistency for the majority of sources of PM_{10} emissions. This indicates that the currently adopted PM_{10} emissions estimates in Australia are more appropriate than application of the factors adopted in the United States. Some variability remains, and for a number of mining activities high quality emission factors are not available due to the absence of local fugitive emission sampling data. Overall, it is considered appropriate to develop improved PM_{10} emission factors for those open cut coal mining activities where high-quality data based on emissions

sampling completed in Australian mines is not currently available. This is consistent with the conclusions of a study published by the Western Australian Department of Environment in 2005 (Sinclair Knight Mertz, 2005).

For $PM_{2.5}$, the measurement data is consistent with the scaling factors currently recommended by the US EPA. However, recommended $PM_{2.5}$ emission estimation methods for fugitive releases from Australian coal mines are not currently available as there is no requirement to collate and publish these emissions. Given the variability identified between the US EPA AP 42 emissions data for PM_{10} when compared to Australian emissions, it is considered appropriate to develop Australian emissions estimation methods for $PM_{2.5}$ from open cut coal mining activities. Furthermore, the measurement dataset demonstrates that the relative significance of $PM_{2.5}$ related mine emissions as a proportion of the overall particulate matter increases with distance from the source. When this is considered in the context with the greater health risk associated with the $PM_{2.5}$ size fraction relative to TSP and PM_{10}, it is clear that development of accurate emission estimation techniques for fugitive $PM_{2.5}$ releases from coal mines is a significant gap in our current knowledge.

In terms of the most significant source of particulate emissions from open cut coal mines, the compositional analysis confirms that local geological sources dominate. This is consistent with the observations of Mudd (2007 and 2009) with respect to the increasing proportion of over burden per unit of extracted coal, and the findings of Kaufman et al (2002) and Deshmukh et al (2012).

4.6 PUBLISHED PAPER 1

Appendix A presents a published paper relating to this research investigation of the characteristics of fine particles released from Australian Open Cut Coal Mines. The paper was co-authored with my joint principal supervisors Professor Igor Agranovski and Dr Shannon Rutherford. The bibliographic details for the paper are as follows:

> Charaterisation of Particulate Emissions from Australian Open Cut Coal Mines: Towards Improved Emissions Estimates (2018). Richardson C, Rutherford S, et al. *Journal of Air and Waste Management Association,*Vol 68, No 6, 598 - 607.

My contribution to the paper included completion of all of the fieldwork components, gravimetric sample analysis, data analysis, drafting the manuscript, preparation of all figures and tables in preparation for publication.

5 REGIONAL VARIABILITY OF PARTICLE EMISSIONS

5.1 INTRODUCTION

Chapter 4 provides an initial analysis of the characteristics of fine particles from open cut coal mines in Australia. The research demonstrates that the particulate size fraction varies for different mining activities, and that the region in which the mine is located may also influence the characteristics of the particulates emitted to the atmosphere. This indicates that development of region-specific emission estimation methods is necessary to allow accurate prediction of particulate emissions to inform regulatory decisions and for use in modelling predictions.

On the basis of this preliminary investigation, it was concluded that further research was needed to develop accurate emission factors for Australian open cut coal mines, to provide for more accurate quantification and prediction of the impacts of mining related to particulates, which are a key input to environmental impact assessments and regulatory policy decisions.

The research presented in this Chapter relates to determination of surface erosion related emission rates for different regions in Australia. The sampling methodology adopted for this research is described in detail in Chapter 3. The sampling locations and results are described in the following sections.

5.2 SAMPLING PROGRAMME

5.2.1 Sampling Sites

Three mines suitable for representing a range of soil types (which in turn affects silt content) and differing moisture and climatic conditions were selected for inclusion in the sampling programme.

To achieve this objective sampling was completed at the same Queensland mine (QLD Mine 1 – located in Central QLD) as for the initial research project (Chapter 4), a second mine in Queensland (QLD Mine 2 – located in South-Eastern QLD) and NSW Mine 1 (as per Chapter 4).

At each mine, individual test locations were selected where available to represent the following:

- heavy vehicle haul road;
- light vehicle haul road;
- dragline walk road;
- tailings dam/spoil piles; and
- coal and/or rejects stockpiles.

Due to accessibility difficulties or other sampling constraints such as weather conditions, it was not possible to complete testing of each of the above types of test location for each of three mines.

All test locations were sampled for dust emission potential at a range of wind speeds, with moisture content and silt fraction also tested for each sample location. Additional tests were undertaken for some sites to determine the effectiveness of control measures such as watering and natural processes such as surface crusting. Haul roads were a particular focus for the tests of watering regimes as watering is a commonly adopted dust suppression mechanism in Australian coal mines.

The portable wind tunnel (refer to Section 3.3.3) was designed to provide easy setup` on-site while maintaining the existing surface structure of the soil. Test sites were chosen carefully to ensure the following conditions were met:

i. The test site was representative of typical conditions at the mine.

ii. The test site was suitable for the tunnel to be operated (sufficiently level).

iii. External interference to sampling was minimised.

The first two conditions were met through discussion with mine staff and a preliminary site inspection. For condition 3, it was necessary to identify sites that were not heavily used while still providing a representative test site. This was necessary for two reasons. Firstly, this approach reduced the risk of test results being biased due to dust generated by vehicles passing the sampling position or nearby activities and therefore contaminating the intake air to the fan. Secondly, completing the testing away from an actively worked area of the mine was necessary from a safety perspective. Setting up a 15 m to 20 m wind tunnel on any actively worked area of the mine or a vehicle road (especially a heavy vehicle route) provides a significant hazard for mobile plant and vehicles.

5.2.2 Influence of Background Concentrations

To minimise the influence of background concentrations, test locations were selected that were not influenced significantly by localised sources of particulate emissions in the mine and measurements were only completed under light conditions. Limitation of impacts from other potential dust sources on-site was further addressed by the positioning of the tunnel at the site. To minimise the risk of dust generated by passing vehicles causing bias in the results, the wind tunnel was set up such that the clean air intake was upwind of the location of any dust sources (passing traffic etc.) whenever possible. This is consistent with the US EPA (US EPA, 1988a) and Australian (Environment Australia, 2012b; NERDCC, 1988) research and methodologies. On this basis, the currently adopted open area emission rates, derived from wind tunnel testing, are not corrected for background concentrations.

5.2.3 Sampling Constraints

As noted previously, there were certain limitations on accessibility and availability for some of the test sites at the three mines. Other factors also affected the field trials including the occurrence of unexpected rainfall and high ambient wind speeds during some stages of the research programme.

Prior to commencing the fieldwork at each test site, the absence of significant rainfall during the previous 14 days was confirmed, and the medium range weather forecast checked to confirm whether rain or high winds were expected. Rainfall is the only meteorological condition that renders the wind tunnel method problematic, as rainfall dampens the test surfaces and reduces emission rates. To provide comparable data, sampling was completed when natural moisture did not provide a significant reduction in particulate emissions from the test surfaces. Wind speeds do not directly affect the measurements within the wind tunnel, as the sampling area is contained. However, as the tunnel working sections are constructed of light weight aluminium, there is a risk of the units blowing over in high winds hence the inability to sample under such conditions.

During the field work phase some of the testing had to be abandoned due to the occurrence of rain or high winds. Due to time and budgetary constraints, in some instances it was not possible to return to the site to repeat or recommence specific tests.

One further factor that affected the first phase of testing, that had not been anticipated, was the sucking of insects into the wind tunnel. During the field trials at QLD mine 1 there were large numbers of Cicadas present. The air intake of the wind tunnel drew in the insects, and there were risks that the flow conditioning elements of the tunnel could become blocked. As a result, an insect mesh had to be placed over the wind tunnel air intake. This risk had not been noted by previous researchers and may be peculiar to test conditions in Australia. The rotational speed was adjusted to account for the friction introduced by the insect mesh prior to sampling.

5.2.4 Test Runs

A summary of the final particulate sampling runs that were completed at each mine using the portable wind tunnel is presented in Table 5-1.

Table 5-1: Summary of Sampling at Each Mine Site

Test Run:	QLD Mine 1	QLD Mine 2	NSW Mine 1
Haul Road	✓	✓	✓
Dragline Walk Road	✓	-	-
Spoil Pile	✓	✓	✓
Rejects	✓	✓	-
Tailings	-	✓	-
Coal Stockpile	-	-	✓

For each of the test sites, a range of surface wind speeds were tested as follows:

- 5 m s^{-1};
- 6.7 m s^{-1};
- 8.9 m s^{-1}; and
- 10.8 m s^{-1}.

The wind speeds were selected to represent the range of wind speeds occurring under normal atmospheric conditions in Australia. The lower wind speed of 5 m s^{-1} was adopted as this is the cut-off point used in the NPI handbook for the threshold velocity. At lower wind speeds the NPI handbook assumes that wind erosion related emissions are negligible.

It should be noted that the selected wind speeds are measured 1 m from the surface being tested. The standard reference measurement position for meteorological stations is a height of 10 m, i.e., close to or at gradient wind conditions[2]. Thus, average reported wind speeds for meteorological stations will

[2] Gradient wind is the wind velocity that is not subject to retardation influences by roughness elements at the ground surface.

normally be significantly higher than for measurements made near to the ground surface where retardation effects occur as a result of surface obstructions.

5.3 RESULTS

5.3.1 Surface Erosion Emission Rates

Table 5-2 presents the emission rates for the tested wind speeds for each of the surface types without surface watering. These data represent continuous particulate emission rates for wind speeds in excess of 5 m s^{-1} 100 % of the time, and where rainfall has not occurred in the preceding 14 days. The highest measured average emission rate of 7.11 kg ha^{-1} hr^{-1} was for the dragline walk road at QLD Mine 2. For this source, the emission rate at 10.8 m s^{-1} for QLD Mine 1 is more than 4 times the rate at 5 m s^{-1}. The highest emission rate at a specific wind speed is for crusted spoil at 10.8 m s^{-1}. At this wind speed the emission rate of 12.14 kg ha^{-1} hr^{-1} is 8 times the emission rate at 5 m s^{-1}.

Table 5-2: TSP Emission Rates (Without Surface Watering)

Mine	Source	Emission Rate (kg ha^{-1} hr^{-1})				Average
		5 m s^{-1}	6.7 m s^{-1}	8.9 m s^{-1}	10.8 m s^{-1}	
QLD mine 1	Haul Road	2.78	5.51	8.15	12.00	7.11
QLD mine 1	Haul Road (after truck passed)	2.39	1.46	4.78	4.27	3.23
QLD mine 2	Haul Road	1.97	-*	3.28	3.93	3.06
NSW mine 1	Haul Road	-*	3.31	8.43	8.97	6.91
QLD mine 2	Dragline Walk Road	5.41	6.91	-*	9.24	7.19
QLD mine 2	Spoil (Uncrusted)	2.90	6.78	6.46	8.69	6.21
NSW mine 1	Spoil (Uncrusted)	1.57	3.81	3.84	6.07	3.82
QLD mine 1	Spoil (Crusted)	1.48	1.77	5.11	12.14	5.12
QLD mine 1	Spoil (Uncrusted)	6.06	-*	5.57	-*	5.82
QLD mine 2	Rejects	3.57	1.38	2.09	6.34	3.34
QLD mine 1	Rejects	4.33	3.40	4.07	7.07	4.72
QLD mine 1	Tailings	-*	-*	-*	0.02	0.02
QLD mine 2	Dragline stockpile	-*	-*	-*	6.09	6.09
	Average	**3.25**	**3.81**	**5.18**	**7.07**	**4.83**

* Valid sample data is not available.

The lowest emission rate at a specific wind speed is also for crusted spoil, with an emission rate of 1.48 kg ha^{-1} hr^{-1}. This indicates that at the lower wind speeds surface crusting is reducing the emissions to below typical surface emission rates, and at the higher wind speeds the surface crust could be compromised, thus allowing significant emissions to occur. The emission rates at the intermediate wind speeds are consistent with this and indicate that the surface crusting is beneficial at a wind speed of

6.7 m s^{-1}. Surface crusting no longer provides a benefit at 8.9 m s^{-1} as emission rates at this wind speed were similar to emission rates for other surfaces at this wind speed.

The results of the surface emission rate testing with surface watering controls are presented in Table 5-3. The emission rates at each wind speed, and the overall average, with surface watering are lower than the emission rates without watering as presented in Table 5-2.

Table 5-3: TSP Emission Rates with Surface Watering

Mine	Source	Emission Rate (kg ha^{-1} hr^{-1})				Average
		5 m s^{-1}	6.7 m s^{-1}	8.9 m s^{-1}	10.8 m s^{-1}	
QLD Mine 1	Haul Road (after truck passed)	2.27	2.15	2.61	4.13	2.79
NSW Mine 1	Haul Road	4.6	2.79	6.00	5.75	4.79
NSW Mine 1	Stockpile (washed coal)[a]	1.96	1.55	3.74	5.38	3.16
QLD Mine 2	Haul Road	1.72	0.67	2.46	2.84	1.92
	Average	**2.64**	**1.79**	**4.36**	**5.52**	**3.6**

[a] Coal was wet due to washing, not watering

Summaries of the average emission rates by region and by activity are presented in Table 5-4 and Table 5-5. The overall average emission rates by region are similar and range from 4.7 kg ha^{-1} hr^{-1} to 4.82 kg ha^{-1} hr^{-1}. There is greater variability in the emission rates for the different activities. The lowest emission rates were measured for the tailings dam (0.02 kg ha^{-1} hr^{-1}), and this is due to the surface being in a moist state. The highest activity emission rate is for the dragline stockpile, at 6.09 kg ha^{-1} hr^{-1}.

Table 5-4: Average TSP Emission Rates for Each Mine (Without Surface Watering)

Mine	Emission Rate (kg ha^{-1} hr^{-1})				Average
	5 m s^{-1}	6.7 m s^{-1}	8.9 m s^{-1}	10.8 m s^{-1}	
QLD Mine 1	3.41	3.04	5.54	7.1	4.77
NSW Mine 1	1.57	3.56	6.14	7.52	4.70
QLD Mine 2	3.46	5.02	3.94	6.86	4.82
Average:	**3.25**	**3.81**	**5.18**	**7.07**	**4.83**

Table 5-5: TSP Emission Rates by Activity (With Surface Watering)

Source (No. of Samples)	Emission Rate (kg ha^{-1} hr^{-1})				Average
	5 m s^{-1}	6.7 m s^{-1}	8.9 m s^{-1}	10.8 m s^{-1}	
Haul Road (4)	2.38	3.43	6.16	7.29	4.81
Spoil (4)	3.00	4.12	5.25	8.97	5.33
Rejects Stockpile (2)	3.95	2.39	3.08	6.71	4.03
Dragline Walk Road (1)	5.41	6.91	-*	9.24	5.39
Tailings (1)	-*	-*	-*	0.02	0.02
Dragline Stockpile (1)	-*	-*	-*	6.09	6.09

5.3.2 Wind Speed Dependent TSP emission rates

The particulate emission dataset has been further analysed to determine equations for predicting emission rates at specific wind speeds. An overall emission equation has been developed for the dataset, as well as region specific emission equations for the three mining regions included in the monitoring programme. Activity specific emission rates have also been calculated for haul routes. Insufficient sample points are available for other activities to allow calculation of statistically robust emission equations.

Graphs presenting the wind speed dependent particulate emission rate equations are presented in Figure 5-2 to Figure 5-5. Linear equations are presented on the graphs for wind speeds in the range 5 ms^{-1} to 12 m s^{-1}. The lower bound wind speed of 5 m s^{-1} represents the minimum wind speed for wind erosion of particulates from an open surface (US EPA, 1995a). 12 m s^{-1} represents the upper validated range of wind speeds for the wind tunnel. The linear equations presented on the graphs allow calculation of emission rates for intermediate wind speeds (x = wind speed, y = emission rate). This allows development of region-specific emission rates based on local wind speed frequency data.

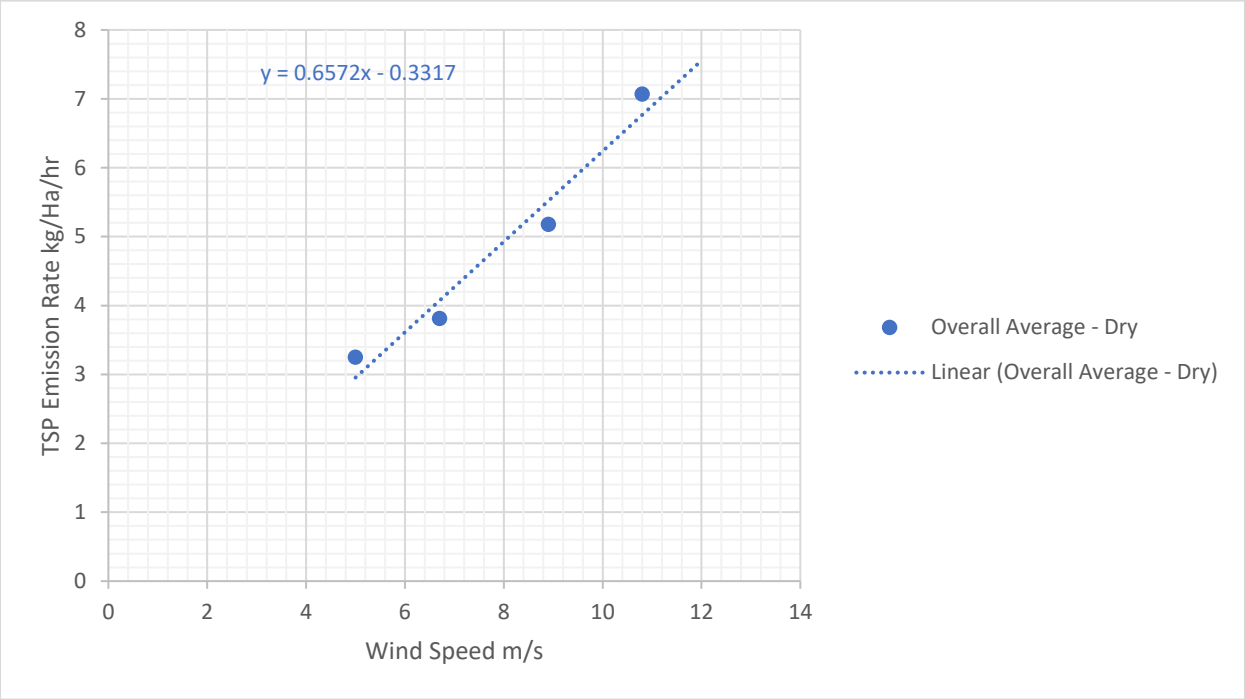


Figure 5-1: Overall Average TSP Emission Rates – Without Surface Watering

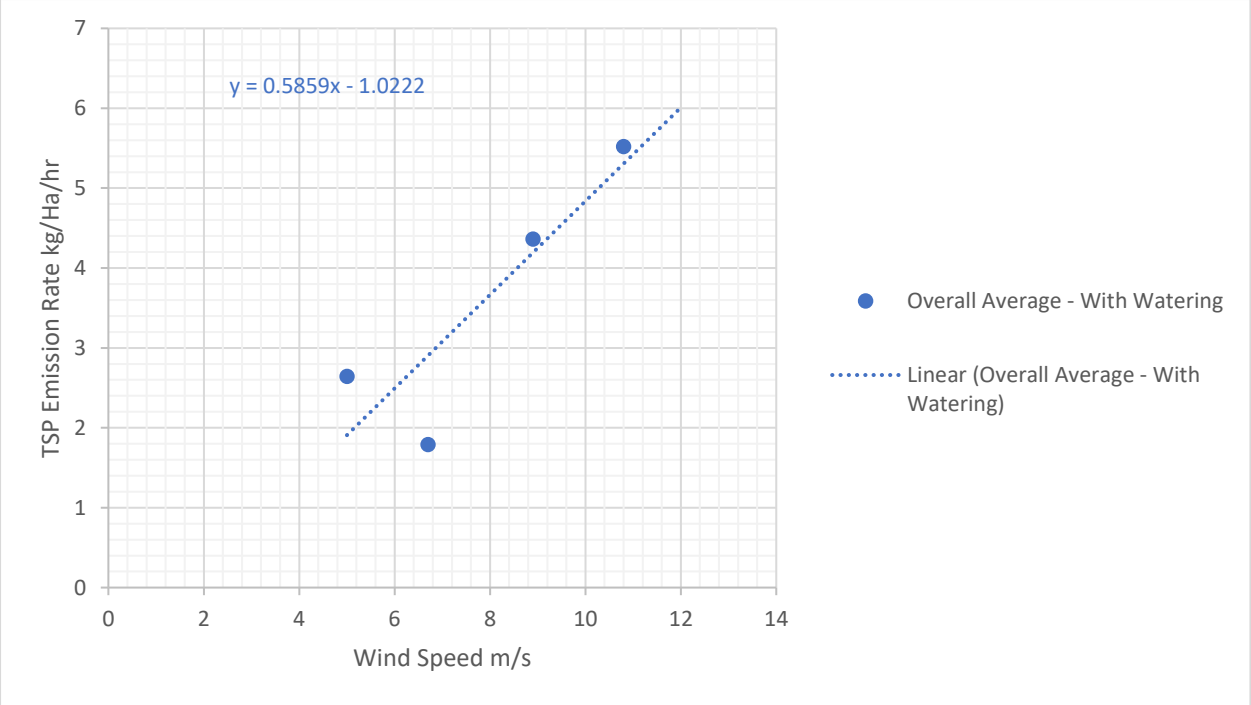


Figure 5-2: Overall Average TSP Emission Rates – With Surface Watering

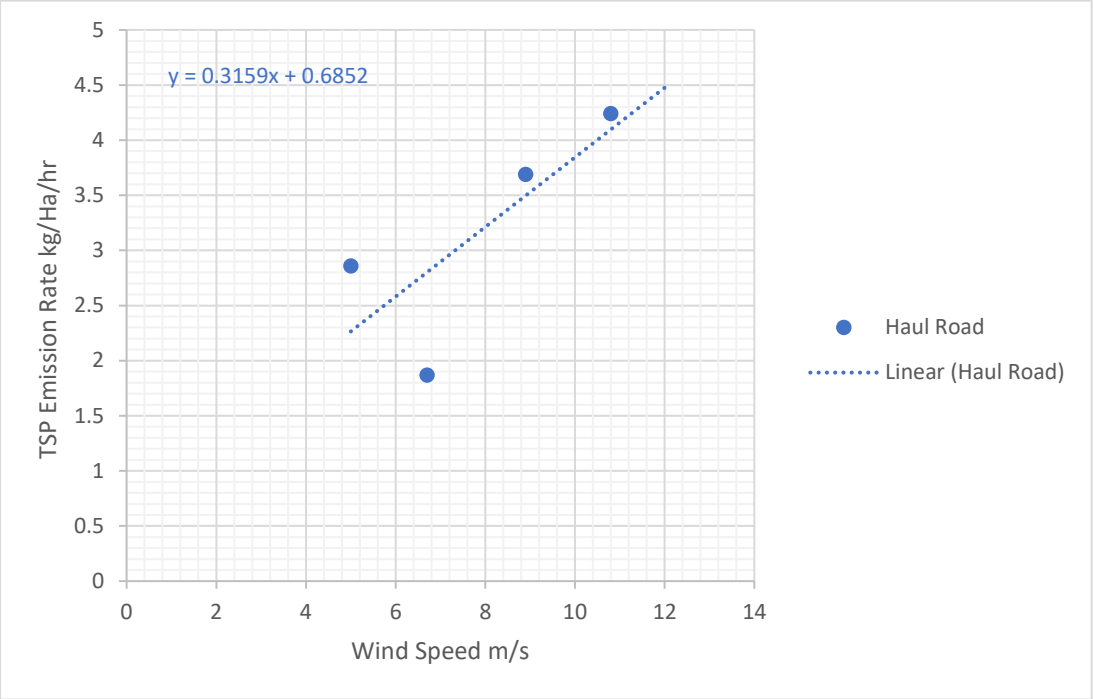


Figure 5-3: Average Haul Route TSP Emission Rates – With Surface Watering

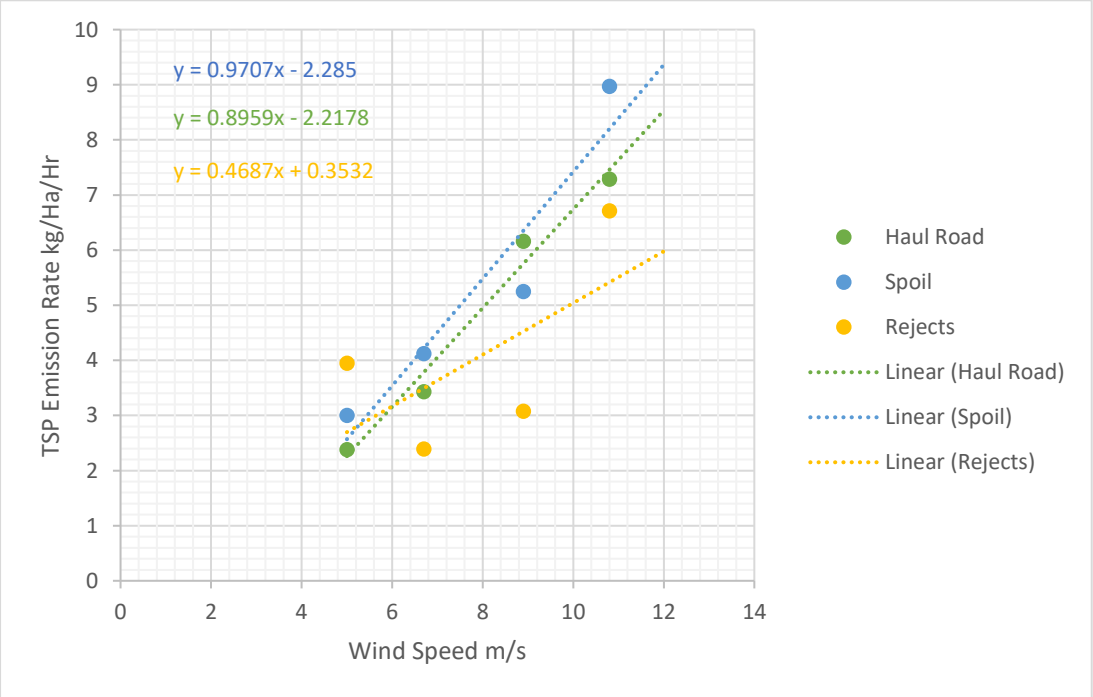


Figure 5-4: Average TSP Emission Rates by Activity – Without Surface Watering

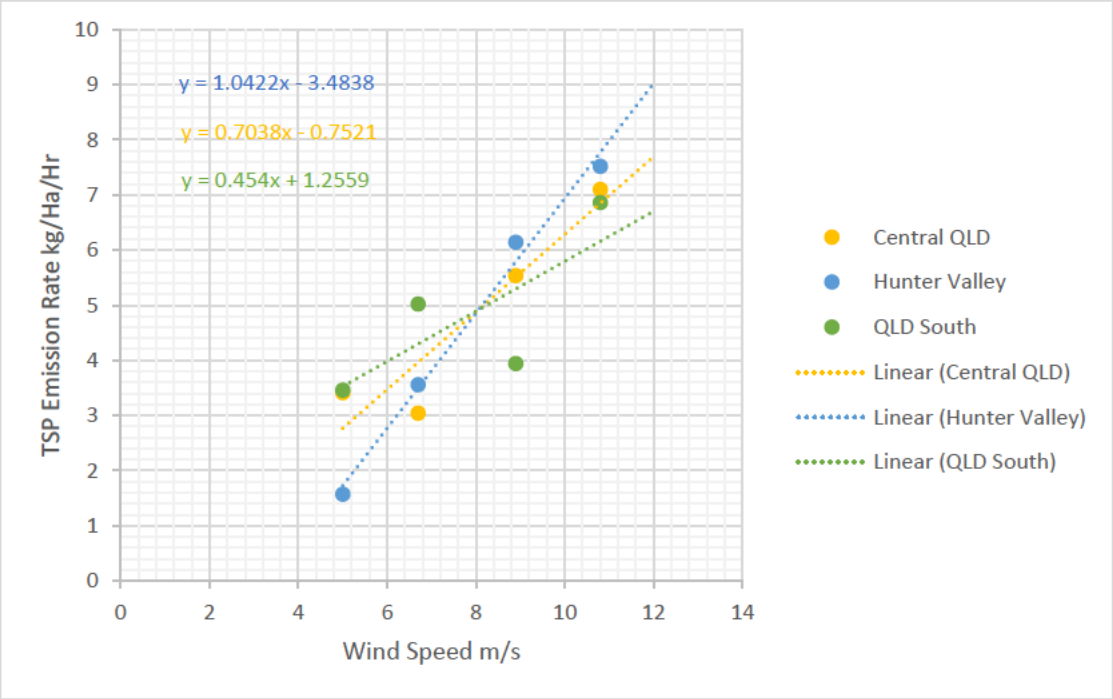


Figure 5-5: Average TSP Emission Rates by Region – Without Surface Watering

5.4 DISCUSSION

5.4.1 Overview

Overall, the measured emission rates demonstrate that particulate emission rates increase as the wind speed increases (Table 5-2 and Table 5-3, Figure 5-1 to Figure 5-5). This trend is less apparent for the watered surfaces (Table 5-3), indicating that the erosion threshold wind velocity increases where the surface moisture content is higher. This is consistent with the action of the cohesive forces of the adsorbed water film surrounding the soil particles reducing the potential for wind erosion and entrainment of the particulate matter (Wiggs, Baird et al, 2004). Comparison of the emission rates without surface watering (Table 5-2) and with surface watering (Table 5-3) demonstrates that surface watering resulted in an overall average control efficiency of 25 %.

The average TSP emission rates without watering (Table 5-4) are consistent for the three mine regions, with similar TSP emission rates determined as an average across all wind speeds tested (a range of 4.70 – 4.82 kg ha hr^{-1}). It is noted that the research presented in Chapter 4 indicates a higher proportion of PM_{10} emissions on average for all mining activities in QLD relative to NSW as a percentage of TSP. This is not exhibited for the TSP emission data for open surfaces.

Review of the emission rates by mine region at specific wind speeds confirms that there was limited variability between the TSP emission rates at 5 m s^{-1} and 6.7 m s^{-1} with average emission rates of 3.25 and 3.81 kg ha hr^{-1} respectively. At 8.9 m s^{-1} there is a 36 % increase in average particulate emission rates relative to emissions at 6.7 m s^{-1}, and an 86 % increase to 7.07 kg ha hr^{-1} at 10.8 m s^{-1}, relative to the emission rate at 6.7 m s^{-1}. These results indicate that at wind speeds at or below 6.7 m s^{-1} the action of the wind and saltation particulate erosion processes on the mine surface is insufficient to result in

significant surface erosion. At wind speeds of 8.7 m s^{-1} and above there is a significant increase in emissions, indicating that the threshold friction speed has been reached at which both particle suspension and saltation processes occur (Gillies & Lancaster, 2013). These threshold wind speeds are slightly lower than reported for wind erosion of open surfaces in the Athabasca oil Sands Region in Alberta, Canada, where the lowest threshold wind speed for saltation processes was determined as 9.44 m/s, and more typically occurred at 10.3 m s^{-1} (Wang, Chow et al, 2015).

As noted in Section 5.3.1, the sampling results for the crusted and uncrusted soil indicate that at the lower wind speeds surface crusting is reducing the emissions to below typical surface emission rates, however at the higher wind speeds the surface crust appears to be compromised, as a significant increase in emissions occurs. The emission rates indicate that the surface crusting is beneficial at a wind speed of 6.7 m s^{-1}, and no longer provides a benefit at 8.9 m s^{-1} as emission rates at this wind speed were similar to emission rates for other surfaces at this wind speed.

5.4.2 Evaluation of Measured Emission Rates

The existing published emission rates (Environment Australia, 2012a; US EPA, 1998a) define an average 'default' emission rate of 0.4 kg ha^{-1} hr^{-1} for TSP from open erodible surfaces at black coal mines. The measured data represent emission rates at specific wind speeds, during dry conditions. Therefore, based on historic wind speed and rainfall data for the test regions, site specific average wind erosion emission rates for the tested surfaces were determined. Table 5-6 presents a comparison of the measured emission rate (dry, > 5 m/s wind speed) with average emission factors adjusted for annual rainfall and wind speed profiles, and the US EPA and Australian NPI default surface erosion emission rate.

Table 5-6: Measured Emission Rates Compared to Weather Corrected and Default Emission Rates

Location	**TSP (kg ha^{-1} hr^{-1})**			
	Average Emission Rate (Wind > 5 m/s, no rain)	**Average Emission Rate Adjusted for Wind Speed and Rainfall**	**Default Emission Rate NPI[1]**	**Default Emission Rate AP42[2]**
QLD Mine 1	4.8	0.07	0.4	0.1
NSW Mine 2	4.7	0.35	0.4	0.1
QLD Mine 2	4.8	0.42	0.4	0.1
Average	**4.8**	**0.28**	**0.4**	**0.1**

[1] Default emission rates for TSP from open erodible surfaces (Environment Australia, 2012a),

[2] Default emission rates for TSP from open erodible (US EPA, 1998a)

The results presented in Table 5-6 show that the measured, weather corrected average emission rates for QLD mine 2 and NSW mine 1 are within 15 % of the NPI default emission rate. However, the measured emission rate for QLD mine 1 is substantially lower than the default 0.4 kg ha^{-1} hr^{-1} emission rate used for estimating emissions from Australian coal mines and is similar to the default emission rate recommended in the US EPA AP42 (US EPA, 1998a). For QLD mine 1 (located in Central QLD) there is a low occurrence of wind speeds in excess of 5 m s^{-1} and a higher annual rainfall than the other test sites, based on observations obtained from the nearest Bureau of Meteorology meteorological monitoring station to the mine (Bureau of Meteorology, 2018). This results in lower calculated

emission rates (Table 5-6) due to the local meteorological conditions. This comparison demonstrates the importance of local wind speed and rainfall to overall annual emission rates from surface erosion. The availability of emission rates at specific wind speeds, as provided in this study, allow for calculation of region-specific surface erosion emission rates and adjustment for the local wind speed conditions. This has important implications for dispersion modelling studies, as adoption of the default emission rates could significantly under- or over-estimate actual emissions for a specific locality such as Central QLD.

The measured emission rates without surface watering are equivalent to the worst-case continuous emission rate under dry, windy conditions, thus representing the upper bound of emission rates likely to occur in the short term. The unadjusted emission rates represent worst case, short term dust erosion events, hence are suitable for use in predictions of short-term dust nuisance or short-term acute health impacts in the vicinity of open cut coal mines.

Whilst there is extensive research relating to particulate concentrations in the vicinity of coal mines (Aneja, Isherwood et al, 2012; Gautam, Prasad et al, 2016; Ghose, 2007a; Huertas, Huertas et al, 2014; Pokorná, Hovorka et al, 2016), there is limited empirical data in the published literature relating to surface emission rates from coal mining. As noted previously, the US EPA (US EPA, 1998a) and NERDCC (NERDCC, 1988) studies that form the basis of the emission estimation equations developed for open cut mines in the US and Australia are dated. A more recent investigation by Chakraborty et al (Chakraborty, Ahmad et al, 2002) presents measured emission rates from surface erosion in exposed open cut pits at eight coal mines in India with an average emission rate of 0.5 kg ha^{-1} hr^{-1}. This emission rate is similar to the default emission rate of 0.4 kg ha^{-1} hr^{-1} adopted in the Australian NPI methodology. Comparison with the empirical data measured in this study comparison demonstrates that the average emission rates for Australian mines, when adjusted for local weather conditions, are significantly lower than the average reported for Indian mines.

5.5 CONCLUSIONS

Using a wind tunnel approach to sampling, this research has identified wind speed specific surface emission rates for a range of open area sources at coal mines in Australia. The empirical data demonstrates that at wind speeds of 5 m s^{-1} – 6.7 m s^{-1} there is limited variation in particulate emission rates. At higher wind speeds, emission rates increase significantly, and at a wind speed of 10.8 m s^{-1} particulate emission rates are 86 % higher than at a wind speed of 6.7 m s^{-1}. This demonstrates the significance of higher wind gusts in generating particulate emissions, and the necessity of considering local meteorological data when developing particulate emission inventories. The data also indicates that surface crusting is beneficial in reducing surface emissions up to a wind speed of 8.9 m/s.

The empirical data developed in this study has increased our knowledge of surface particulate emissions in coal mines. Firstly, wind speed specific emission rates have been determined. This allows for calculation of region-specific emission rates based on local meteorological conditions. Secondly, emission rates have been determined for a range of open erodible surfaces in open cut black coal mines. Currently, a single generic default emission rate is adopted to represent these sources, hence the research provides for more detailed analysis and calculation of site-specific particulate emission rates.

5.6 PUBLISHED PAPER 2

Appendix B presents a published paper relating to this research investigation of particulate emissions from surface erosion in Australian open cut coal mines. The paper was co-authored with my joint

principal supervisors Professor Igor Agranovski and Dr Shannon Rutherford. The bibliographic details for the paper are as follows:

> Wind Speed Dependent Particulate Emission Rates for Open Surfaces in Open Cut Black Coal Mines, Richardson C, Rutherford S, et al (2019). *Journal of Environmental Management*, 232 (2019) 537 - 544.

My contribution to the paper included development of the research methodology, supervising the construction of the sampling wind tunnel, supervising the fieldwork, completing the gravimetric sample analysis, data analysis, drafting the manuscript, preparation of all figures and tables in preparation for publication.

6 FINE PARTICULATE EMISSION RATES FOR AUSTRALIAN OPEN CUT BLACK COAL MINES

6.1 INTRODUCTION

The previous chapter considers the variability of particulate emission rates from open surfaces in coal mines, and the significance of wind speed and regional differences on these emission rates. The next phase of research for this PhD programme focused specifically on a current gap in our knowledge – the lack of empirical data relating to $PM_{2.5}$ emissions from open cut coal mines. This research investigated resulted in the development of activity specific emission rates for $PM_{2.5}$ for Australian open cut coal mines.

The sampling methodologies are described in Chapter 3. The following sections identify the sample locations and results of the investigation.

6.2 SAMPLING PROGRAMME

6.2.1 Sampling Locations

Research has shown that mining particulate emission rates may vary on a regional basis or even within mines where significant geological differences occur (Ghose, 2007a; Huertas, Huertas, Izquierdo et al, 2012) . The sampling programme was completed in two Australian mining regions. Sampling was completed in Central Queensland at the same mine as the previous studies (QLD Mine 1) and New South Wales Mine 1. Additional testing was completed at two additional mines in the NSW Hunter Valley, denoted as NSW Mines 3 and 4. The three NSW mines were all located in close proximity to each other in the Hunter Valley, hence are reported together in this research as regional parameters such as soil type and climate were very similar between the sites. Each mine operated a number of pits and working faces, hence allowing for sampling of the same activity at a number of different locations within the mine subject to local conditions and work schedules.

Table 6-1 summarises the mining activities and number of test sites per activity for the sampling programme. The number of hours that a full transect of samples were completed (i.e. 4 positions x 1-hour sample per position), and the total number of hours of sampling for each individual transect position is also identified.

When identifying the suitability of specific sampling locations for the study, the following parameters were considered:

- absence of significant rainfall in the previous month. Sampling was only completed where materials or surfaces were in a dry condition, except where watering was a standard control technique (e.g. for haul routes);

- light to moderate winds. Sampling could not be completed during zero or very light wind conditions, as there was no downstream plume of emissions. Sampling was not completed under average wind speeds of > 5 m/s which would result in rapid dispersion of emissions leading to difficulties in accurately characterising the width and height of the downwind plume. Typical average wind speeds for the measurements were in the range 2 – 3 m/s, with less than 1 % of 1-minute weather observations exceeding 8 m/s.

- logistical and safety issues. The availability of target activities (such as dragline operation) during the proposed sampling period was a specific issue, and close liaison with the mine operator was necessary to ensure fieldwork coincided with target activities.

- The initial sampling position on the transect was typically located 20 – 60 m from the source of emissions. For some operations, such as the dragline, larger separations were necessary for safety reasons and the initial transect Position 1 was located 100 m or more from the source. The intermediate and final positions on the transect were largely defined by localised constraints such as proximity to earth berms and pit edges. The average distance for Position 4 on the transect was 103 m from the initial (Position 1) sampling location, with the intermediate positions equally spaced between these points.

Figure 6-1 to Figure 6-3 present examples of the sampling instrumentation in position for three different mining activities.

Table 6-1: Summary of Test Locations

Mine Activity	**Source Type**	**QLD mine 1**			**NSW mines**		
		Sites Tested	1-hour Data Points[a,b]	Transect Hrs	Sites Tested	1-hour Data Points[a,b]	Transect Hrs
Coal Haul Road	Line	2	25	7	3	30	3
Overburden Haul Road	Line	2	19	4	4	36	6
Drilling	Point	3	41	6	3	40	6
Dragline	Point	3	45	7	1	21	5
Overburden Loading	Point	2	24	5	8	86	18
Overburden Dumping	Point	2	25	4	2	12	2
Coal Loading	Point	3	54	9	2	26	2
Coal Dumping	Point	2	24	5	1	18	3.5

[a] This is the total number of one-hour measurement data points that were considered in determining the emission factors for each activity.
[b] The total number of hours is less than 4 per transect in some instances. This is due to instruments failing post sampling quality assurance checks or due to site constraints limiting the number of samplers that could be positioned along the transect to less than 4 positions.

Figure 6-1: Dragline Sampling Transect (Air Noise Environment Pty Ltd, 2015a)

Figure 6-2: Overburden Loading Sampling Transect (Air Noise Environment Pty Ltd, 2015b)

Figure 6-3: Drilling Sampling Transect (Air Noise Environment Pty Ltd, 2015c)

6.3 RESULTS

The emission factors determined from the sampling data for each activity are presented for the NSW mines and QLD Mine 1 (Central QLD) in Table 6-2. The calculated standard deviation is presented for each emission factor. These results have been corrected for measured background concentrations, hence represent activity specific emission rates.

Table 6-2: $PM_{2.5}$ Emission Rates (Background Corrected)

Mine Activity	Overall Average		QLD Mine 1		NSW Mines		1-Hour Data Points	Units
	Emission Rate	Standard Deviation	Emission Rate	Standard Deviation	Emission Rate	Standard Deviation		
Coal Haul Road	0.0803	0.1260	0.0321	0.0283	0.1912	0.1869	55	kg/VKT
Overburden Haul Road	0.0357	0.0371	0.0199	0.0220	0.0470	0.0417	55	kg/VKT
Drilling	0.4727	1.0747	0.0801	0.1265	0.8383	1.4025	81	kg/hole
Dragline	0.0135	0.0237	0.0149	0.0295	0.0113	0.0069	66	kg/BCM[a]
Overburden Loading	0.0012	0.0015	0.0004	0.0004	0.0014	0.0016	110	kg/t
Overburden Dumping	0.0005	0.0007	0.0004	0.0007	0.0007	0.0007	37	kg/t
Coal Loading	0.0039	0.0087	0.0046	0.0093	0.0003	0.0003	80	kg/t
Coal Dumping	0.0046	0.0064	0.0083	0.0070	0.0004	0.0006	42	kg/t

[a] Bank Cubic Metre – one metre cubed corrected for the density of the material

The standard deviations for the dataset are relatively high at 1.0 to 2.3 times the overall average sample result. This indicates a significant degree of variability associated with the results. The US EPA has noted that standard deviations of up to 5 are typical for measured emission rates (US EPA, 1995b). Hence, this is a recognised feature of empirical emission sampling for fugitive sources. To improve data quality, in accordance with the draft US EPA emission data calculation methodology (US EPA, 2013), the raw data has been further analysed to remove samples below the method detection threshold (MDL) and to remove data outliers. The resultant emission rates are presented in Table 6-3. The standard deviations of the average data set are improved, as a result of this data processing, now ranging from 0.7 – 1.7 times the calculated emission rate. This is below the factor of 2.9 exhibited by the data used for the current NPI emission rates (NERDCC, 1988). The background and MDL corrected emission rates are considered in detail in Section 6.4.

Table 6-3: $PM_{2.5}$ Emission Rates (Background and MDL Corrected)

Mine Activity	Overall Average		QLD Mine 1		NSW Mines		1-Hour Data Points	Units
	Emission Rate	Standard Deviation	Emission Rate	Standard Deviation	Emission Rate	Standard Deviation		
Coal Haul Road	0.0715	0.1215	0.0272	0.0286	0.1912	0.1868	40	kg/VKT
Overburden Haul Road	0.0336	0.0275	0.0224	0.0221	0.0411	0.0286	40	kg/VKT
Drilling	0.1691	0.2885	0.0437	0.0414	0.3000	0.3704	47	kg/hole
Dragline	0.0074	0.0073	0.0048	0.0065	0.0113	0.0069	49	kg/BCM[a]
Overburden Loading	0.0009	0.0008	0.0004	0.0002	0.0011	0.0008	93	kg/t
Overburden Dumping	0.0003	0.0005	0.0003	0.0004	0.0005	0.0006	24	kg/t
Coal Loading	0.0006	0.0004	0.0007	0.0003	0.0002	0.0002	46	kg/t
Coal Dumping	0.0046	0.0064	0.0083	0.0070	0.0004	0.0006	35	kg/t

[a] Bank Cubic Metre – one metre cubed corrected for the density of the material

6.4 DISCUSSION

6.4.1 Spatial Variability

Table 6-4 presents a comparison of the emission rates for the QLD Mine 1 and the New South Wales Mines. The comparison demonstrates that the emission rates for QLD and NSW vary by more than a factor of 2 except for overburden haul and overburden dumping. Overall, the NSW emission rates are higher, except in the case of coal loading and dumping where they are lower than the average QLD emission rates. Even in the case of coal loading and dumping, the emission rate variation is well within the range of up to two orders of magnitude observed in the US EPA empirical particulate emissions analysis study for sample datasets with 10 or more data points (Axetell & Cowherd, 1981).

The variability between the two Australian coal mining regions is consistent with sampling studies completed overseas (Chakraborty, Ahmad et al, 2002; Huertas, Camacho et al, 2012) that identify that mining activity emission rates can vary on a regional basis. Chakraborty et al (2002) present particulate emission data for 10 mining regions of India, and these data indicate an average variability by a factor of up to 2.2 between the regions in India for the emission sources considered for the QLD and NSW mines.

Table 6-4: Comparison of Regional Emission Rates - Australia

Mine Activity	QLD	NSW	Ratio – QLD:NSW	Units	QLD – Transect Hrs	NSW – Transect Hrs
Coal Haul Road	0.0272	0.1912	0.14	kg/VKT	7	3
Overburden Haul Road	0.0224	0.0411	0.55	kg/VKT	4	6
Drilling	0.0437	0.3000	0.15	kg/VKT	6	6
Dragline	0.0048	0.0113	0.42	kg/hole	7	5
Overburden Loading	0.0004	0.0011	0.36	kg/BCMa	5	18
Overburden Dumping	0.0003	0.0005	0.60	kg/t	4	2
Coal Loading	0.0007	0.0002	3.50	kg/t	9	2
Coal Dumping	0.0067	0.0003	22.33	kg/t	5	3.5

[a] Bank Cubic Metre – one metre cubed corrected for the density of the material. [b] Number of hours that measurements were completed using a transect of 4 instruments downwind for that mine activity.

To determine whether region specific features were the cause of these differences, Table 6-5 presents a comparison of silt and moisture content data for the mining activities tested. Review of the % silt content data indicates that the data for QLD is typically at the lower range of the measured silt contents for the majority of parameters. This indicates potential for particulate emission rates to be lower than the NSW mines. Moisture content are also higher for QLD for the coal haul roads, overburden haul roads and overburden dumping, however the moisture contents for the dragline and coal dumping are lower than those measured in NSW.

These features may be the cause of the higher calculated emission rates for coal haul, overburden haul and overburden loading for the NSW data. In the case of drilling, the silt fraction is significantly higher than for both regions, with a range of 4.05 – 8.86 %. The higher silt content is associated with the nature of the drilling process, which is likely to result in the breakdown of the natural geological material that comes in to contact with the drill bit to the finest particle sizes possible. This increases the potential for emissions of particulate matter from drilling, hence drilling sources are an important component of a mine particulate emission inventory despite the relatively localised nature of emissions from this activity.

Table 6-5: Regional Silt and Moisture Content

Mine Activity	Silt Content %		Moisture Content %	
	QLD	NSW	QLD	NSW
Coal Haul Road	0.87 (1)	0.08 – 3.22 (2)	4.75 (1)	1.24 – 2.18 (2)
Overburden Haul Road	0.9 (1)	3.2 (1)	4.8 (1)	2.2 (1)
Drilling	6.90 (1)	4.05 – 8.86 (2)	3.44 (1)	2.90 – 6.00 (2)
Dragline	0.82 (1)	1.18 (1)	2.45 (1)	3.19 (1)
Overburden Loading	0.59 (1)	0.13 - 3.92 (4)	5.05 (1)	0.83 – 2.62 (4)
Overburden Dumping	2.93 (1)	N/A	5.01 (1)	N/A
Coal Loading	0.05 – 0.47 (3)	N/A	1.32 – 1.58 % (3)	N/A
Coal Dumping	0.03 (1)	0.04 (1)	1.31 (1)	2.51 (1)

Note: values in brackets are number of measurements for that parameter

6.4.2 Comparison with Existing Emission Rates

Directly comparable empirically derived emission datasets for $PM_{2.5}$ are not currently available. While a range of empirical datasets are available for Total Suspended Particulates and PM_{10} (Chakraborty, Ahmad et al, 2002; Chaulya, 2006; Lal & Tripathy, 2012), calculation of $PM_{2.5}$ emission rates relies primarily on application of $PM_{2.5}$ fractionation estimates from particulate size analysis data to empirically derived total suspended particulate emission rates. Therefore, to determine whether the measured $PM_{2.5}$ emission rates differ significantly from those calculated using the currently adopted emission estimation methods, a comparison is made to emission rates estimated using the currently adopted US EPA AP42 (US EPA, 1998a; US EPA, 1998b) and Australian National Pollutant inventory (Environment Australia, 2012a) methods in Table 6-6 and Table 6-7.

The calculated emission rates have accounted for the particulate control techniques applied during the time of the sampling, to provide a direct comparison. Average silt and moisture content values for the relevant activity as measured in the research study are adopted for the purposes of the calculations, thus the emission estimates account for key local variables.

Table 6-6: Comparison with US EPA AP42 Emission Estimations

Mine Activity	US EPA Method $PM_{2.5}$	Average Study $PM_{2.5}$	Difference	Ratio	Units
Coal Haul Road[a]	0.0664	0.0715	0.0051	1.077	kg/VKT
Overburden Haul Road[a]	0.0935	0.0336	-0.0599	0.359	kg/VKT
Drilling[b]	0.0531	0.1691	0.1160	3.185	kg/hole
Dragline[a]	0.0009	0.0074	0.0065	8.421	kg/BCM
Overburden Loading[a]	0.0010	0.0009	-0.0001	0.943	kg/t
Overburden Dumping[a]	0.0003	0.0003	0.0000	0.943	kg/t
Coal Loading[b]	0.0188	0.0006	-0.0182	0.032	kg/t
Coal Dumping[a]	0.0002	0.0041	0.0039	19.340	kg/t

[a] US EPA $PM_{2.5}$ fraction scaling factor applied to calculated TSP.

[b] Australian SPCC 1986 $PM_{2.5}$ fraction scaling factor applied to calculated TSP

$PM_{2.5}$ fractionation factors are not published in the current Australian NPI Mining manual for fugitive emission sources at open cut coal mines. Therefore, an estimated $PM_{2.5}$ particle size fraction has been applied for the Australian NPI method based on research completed in the Hunter Valley (State Pollution Control Commission NSW, 1986) in calculating the emission rates presented in Table 6-7.

The comparison of emission rates in Table 6-7 confirms that the emissions estimated using the Australian NPI calculation methods are within a factor of 2 of the average measured emission rates for coal haul roads, overburden haul roads, draglines, overburden loading and overburden dumping identified in this study. The US EPA emissions estimates are also similar for coal haul roads, overburden loading and dumping, and greater than a factor of 2 for the remaining activities.

The activities with the most significant differences for both calculation methodologies are coal dumping, drilling and coal loading and for the US EPA emission factors only, draglines and overburden haul. In the case of coal dumping, the average measured dataset is skewed by high emission rates for the Queensland mine; the data for the New South Wales mines (0.0003 kg/t) is equivalent to the average of the US EPA and Australian calculated emission rates (0.0002 kg/t and 0.0004 kg/t respectively). The coal dumping operations sampled in Queensland included stockpile (dozer) and nearby work area maintenance (grader) activities, and this is reflected in the higher emission rates. Whilst stockpile and work area maintenance activities are a regular feature of the coal stockpile activities, the NSW emission rates represent more typical emission rates for coal dumping only.

Table 6-7: Comparison with Australian NPI Emission Estimations

Mine Activity	Australian Method $PM_{2.5}$[a]	Average Study $PM_{2.5}$	Difference	Ratio	Units
Coal Haul Road	0.0511	0.0715	0.0204	1.398	kg/VKT
Overburden Haul Road	0.0303	0.0336	0.0033	1.107	kg/VKT
Drilling	0.0531	0.1691	0.1160	3.185	kg/hole
Dragline	0.0042	0.0074	0.0032	1.762	kg/BCM
Overburden Loading	0.0013	0.0009	-0.0004	0.720	kg/t
Overburden Dumping	0.0005	0.0003	-0.0002	0.625	kg/t
Coal Loading	0.0015	0.0006	-0.0009	0.414	kg/t
Coal Dumping	0.0004	0.0041	0.0037	10.250	kg/t

[a] NPI Default TSP emission rate multiplied by Australian SPCC 1986 0 µm – 2.5 µm size fraction.

Similarly, the drilling emission rates measured for the Queensland mine (0.0437 kg/hole) are within 18 % of the calculated emission rates using the US EPA and Australian NPI methods (0.0531 kg/hole). This indicates the higher emission rate determined for the NSW mines may relate to specific local conditions. The measured coal loading emission rate is significantly lower than the estimated emission rate, indicating that the current emission estimation methods may over-estimate $PM_{2.5}$ from this activity.

For draglines, the US EPA emission rate is more than 8 times lower than the measured emission rate. This indicates that the US EPA AP42 emission estimation method for draglines may significantly underestimate emissions when applied in other regions of the world and, possibly, in US coal mines.

Finally, in the case of overburden haul, the US EPA emission rate (0.0935 kg/vkt) is more than 3 times higher than the measured emission rate (0.0336 kg/vkt) and the emission rate calculated using the Australian NPI calculation method (0.0303 kg/vkt). This indicates a potential for the US EPA calculated emission rate to significantly overestimate $PM_{2.5}$ emissions from over burden haul.

6.5 CONCLUSIONS

This research has determined empirically derived emission rates for $PM_{2.5}$ for a range of open cut mining activities. The results of the study confirm that there may be significant variability in emission rates for different mines, and differences in silt fractions and material moisture contents may be the cause of this variability for a number of the empirically derived emission rates.

Average coal haul route emission rates for $PM_{2.5}$ as derived in this study are consistent with the currently adopted site specific calculation methodologies except where control in the form of watering is considered in the calculation. Where the calculation method is applied, the control efficiency of haul route watering may be significantly over estimated. The Australian NPI default emission estimation

method for overburden haul is also consistent with the empirically derived emission rates and the US EPA calculation method results in over estimation by a factor of three. As with coal haul, where over burden haul emission rates are calculated with watering controls, the resultant emission rate is a significant under estimate of emissions in practice.

Based on the empirical data, the existing US EPA $PM_{2.5}$ emissions estimation techniques for draglines underestimates Australian emissions significantly. This indicates the need to adopt region specific emission rates for dragline emissions.

For coal dumping, significant variability was observed, and this was related to differences in ancillary activities occurring at the time of sampling.

In the case of drilling, the average measured emission rates were higher than the calculated emissions, however the dataset was skewed by the significantly different emission rates observed between the mine test sites.

For coal loading, measured $PM_{2.5}$ emission rates are significantly lower than the calculated emissions, and the current calculation methods may over estimate emissions from this source.

The emissions dataset presented in this paper provides $PM_{2.5}$ emission rates for open cut coal mining activities, based on empirical data. This represents an advance over current emissions estimation techniques which rely on application of $PM_{2.5}$ correction factors to TSP or PM_{10} emission rates to allow estimation of $PM_{2.5}$ emissions.

The study has also highlighted the significant variability associated with empirical measurements of fugitive particulate emissions. This is of particular relevance when risk assessments of existing or proposed future mines are being completed. This feature of the emission data may introduce a greater degree of uncertainty to predictive modelling, and this should be considered in any decision-making process that relies on these data.

6.6 PUBLISHED PAPER 3

The research methodology and results are presented in the paper presented in Appendix C, that was co-authored with my joint principal supervisors Professor Igor Agranovski and Dr Shannon Rutherford. The bibliographic details for the paper are as follows:

> Open Cut Black Coal Mines: Empirical Verification of $PM_{2.5}$ Emission Estimation Techniques. Richardson C, Rutherford S, et al (2019). *Journal of Atmospheric Research*, 216, 151 - 159.

My contribution to the paper included development of research methodology and sampling/analysis methods, participation in fieldwork, QA and data analysis, drafting the manuscript, preparation of all figures and tables in preparation for publication.

7 HAUL ROUTE PARTICULATE EMISSION CONTROL

7.1 INTRODUCTION

In addition to addressing the focus questions outlined in Chapter 2, the body of research produced for this thesis has provided empirical data relating to haul route emission controls. These findings are important, as haul routes are one of the primary sources of emissions at open cut mines. The research findings relating to haul route watering and silt contents for Australian mines are presented in the following sections.

7.2 HAUL ROUTE WATERING AS AN EMISSION CONTROL TECHNIQUE

Haul routes in mines are usually formed by grading open ground. Because the haul route surface is not sealed and, given the quantity and size of vehicles using the haul routes in open cut coal mines, the haul route surface is a significant source of emissions. Watering of haul routes is the primary control method adopted in open cut coal mines for reducing emissions. The control efficiency of haul route watering has been investigated by the US EPA, and the resultant control efficiencies have been adopted in the emission estimation methods adopted in the Australian National Pollutant Inventory Emission Estimation Manual for Mining (Environment Australia, 2012b).

The NPI Emissions Estimation handbook for Mining specifies control efficiencies for haul route watering as follows:

- Level 1 watering: 2 litres m^{-2} hr^{-1} (50 % control efficiency); and
- Level 2 watering: >2 litres m^{-2} hr^{-1} (75 % control efficiency).

The US EPA (US EPA, 2006) has also published a control efficiency estimation graph for haul route watering, and this is reproduced in Figure 7-1. The moisture ratio (M) is defined by the US EPA as the surface moisture content of the watered road divided by the surface moisture content prior to watering.

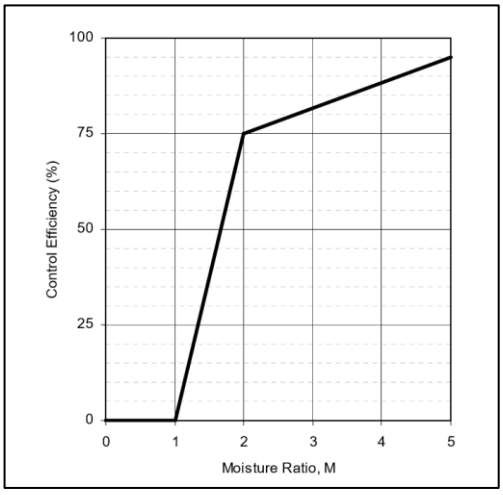


Figure 7-1: Unpaved Road Watering Control Efficiencies (US EPA, 1998b)

7.3 TSP CONTROL RATES BASED ON THE EMPIRICAL MEASUREMENTS

Based on the measured emission rates described in Chapter 5, watering control efficiencies for TSP surface particulate emissions are presented in Table 7-1.

Table 7-1: TSP Emission Rates With and Without Watering

Haul Road Source	kg ha^{-1} hr^{-1} No Watering	% Moisture Prior to Watering	kg ha^{-1} hr^{-1} With Watering	% Moisture After Watering	Watering Rate Litres m^{-2}	% Control Efficiency	Moisture Ratio[a]
QLD Mine 2	3.06	2.0	1.92	9.3	Not Available	37.3	4.7
QLD Mine 1	3.23	Not Available	2.79	8.1	0.85	13.6	-
NSW Mine 1	6.91	1.6	4.79	5.7	0.3	30.7	3.6
Average:	4.4	1.8	3.2	7.7	0.44	27.2	4.2

[a] Moisture ratio is defined as the moisture % after watering divided by the % moisture prior to watering.

Surface watering was completed as a single pass for NSW Mine 1 and QLD Mine 2; for QLD Mine 1 the watercart passed over the surface twice prior to the controlled emission rate test being completed. The double water cart pass for QLD Mine 1 was the standard watering regime adopted at that operation. For the single water cart pass for NSW Mine 1 the application rate was 0.3 litres m^{-2}, and for the double pass at QLD Mine 1 the water application rate was 0.9 litres m^{-2}. If only one of these watering events occurred per hour, the water application rate would be lower than a Level 1 watering regime, as defined in the Australian emission estimation manual for mining (Environment Australia, 2012a), of 2 litres m^{-2} $hour^{-1}$ for a control efficiency of 50 %. To achieve the Level 1 watering rate would require approximately 4 single water cart passes per hour, or 2 double water cart passes per hour.

The research presented in Chapter 6 recorded a range of operational parameters during the testing, including the frequency of water cart use for haul roads. These data are summarised in Table 7-2. The average number of water cart passes as recorded for 13 haul route emission tests was 3 per hour. The most frequently adopted water application regime was two water cart passes per hour, and the mean is skewed by a small number of higher application rates. Both the mean and the mode application rates are lower than required to achieve the application rate of 2 litres m^{-2} $hour^{-1}$ adopted in the NPI emission estimation handbook for achieving a 50 % control efficiency. The average control efficiency is also lower, at 27.2 %, and this is consistent with the water application rate being adopted in the Australian mines.

Table 7-2: Frequency of Haul Route Watering

Mine Location	Haul Type	Single Watering Passes per Hour
QLD Mine 1	Coal Haul (#1)	2
	Overburden Haul (#1)	3
	Overburden Haul (#2)	2
	Overburden Haul (#3)	2
	Overburden Haul (#4)	2
	Coal Haul (#2)	2
NSW Mines	Overburden Haul (#1)	5
	Overburden Haul (#2)	2
	Overburden Haul (#3)	2
	Coal Haul (#1)	4
	Overburden Haul (#4)	3
	Coal Haul (#2)	4
	Coal Haul (#3)	6
	Mode:	2
	Mean:	3

The significance of the water application rate was also tested by reference to specific empirical tests completed to assess the percentage reduction in moisture content over a one-hour period.

Table 7-3 presents the results of two tests completed to assess the reduction in percentage surface moisture over time after watering. This demonstrates that, over a one-hour period, the percentage moisture in the haul route surface decreases by more than 50 %.

Table 7-3: Percentage Moisture Reduction After Watering (Single Water Application)

Haul Road Source	% Moisture Prior to Watering	% Moisture After Watering	% Moisture 1 Hour After Watering
Dragline walk road	1.7 %	12.3 %	6.5 %
Haul road (high ambient temperature)	1.2 %	6.5 %	1.8 %
Average:	**1.45 %**	**9.45 %**	**4.15 %**

A further point of interest with respect to the measured percentage moisture rates in haul route surfaces relates to the pre-watering levels. The US EPA and NPI emission estimation manuals identify typical default moisture contents for different mine activities and surfaces. For haul routes, a default moisture

content of 2 % is identified. The empirical data compiled for this study identifies that, prior to watering, the percentage moisture content is lower for haul routes in Australian open cut coal mines at an average of 1.5 %.

7.4 TSP CONTROL RATES COMPARED TO US EPA CONTROL EFFICIENCIES

The data presented in Table 7-3 indicates an average moisture ratio percentage moisture after watering divided by percentage moisture prior to watering) of 4.2. Based on the US EPA emission factors for unpaved haul roads, as shown in Table 7-4, this would indicate an average watering related particulate control efficiency of >87.5 % (US EPA, 1998b). The empirical data for Australian conditions indicates that lower control efficiencies are achieved in practice than would be expected based on the US EPA AP42 emission control estimation methods for unpaved haul roads.

This indicates that the US EPA AP42 watering control efficiency empirical calculation method is not suitable for application to Australian conditions and should not be used for the calculation of control efficiencies for surface watering. On this basis, application of the moisture ratio as a means for estimating control efficiency could result in over-estimation of the benefit of surface watering as a particulate emission control mechanism, for Australian conditions.

Table 7-4: Moisture Ratio and % Control Efficiency

Haul Road Source	% Control Efficiency	Moisture Ratio[a]	% Control Based on Moisture Ratio	Difference
QLD South	37.3	4.7	> 95 %	57.7 %
Hunter Valley	30.7	3.6	> 80 %	49.3 %
Average:	27.2	4.2	87.5 %	60.3 %

[a] Moisture ratio is defined as the moisture % after watering divided by the % moisture prior to watering

7.5 $PM_{2.5}$ EMISSION HAUL ROUTE WATERING SENSITIVITY ANALYSIS

In terms of the significance of the differences between measured and estimated $PM_{2.5}$ emission rates, haul road emissions are generally the most significant source of mass emissions of particulates in open cut coal mines at >70 % of total particulate emissions (US EPA, 1988b). The dominance of mine haul roads as the primary source of particulate releases is also confirmed by emissions estimates for a range of mining proposals in Australia, as presented in the Environmental Impact Statements for these projects - for example the Foxleigh Plains (Katestone Environmental, 2012) and Bylong Coal (Pacific Environment Ltd, 2015) air quality assessments. Given the significance of haul activities as a source of particulate emissions, further analysis has been completed based on the research presented in Chapter 6 to assess the significance of selection of different variables and assumptions in the calculation methods. Table 7-5 presents a comparison of the measured emission rates for coal and overburden haul routes, with the default emission rates and calculated emission rates as defined in the Australian NPI Emission handbook for Mining (Environment Australia, 2012a).

The comparison presented in Table 7-5 demonstrates that, for coal haul routes, using the Australian NPI method, the calculated emission rate (accounting for surface moisture) is 29 % lower than the measured emission rate determined from the empirical data. Similarly, using the NPI default emission rate with 75 % watering control efficiency, results in a predicted emission rate that is 11 % lower than the

measured emission rate. Whilst these emission rates would result in a degree of underestimation, given the variability associated with fugitive dust emissions estimates, this represents a reasonable correspondence between the measured and calculated datasets. Where the calculated emission rate is also adjusted for emission control in the form of watering, the resultant emission rate is 82 % lower. This approach would result in significant underestimation of potential emissions, hence would not be suitable for use in emissions estimations and modelling studies.

For overburden haul routes, as with the coal haul routes, there is closest correspondence between calculated (with moisture control) and measured emission rates. The default emission rate both with and without controls over estimates emissions, and the calculated emission rate with an adjustment for 75 % control significantly under-estimates emissions by a ratio of >4.

Review of the watering rates occurring when the coal and overburden haul route emission testing was completed confirms that, on average, a water cart traversed the test surface and applied water 2.7 times per hour tested. Therefore, it is concluded that water application at this rate is not sufficient to achieve the 75 % reduction in $PM_{2.5}$ emissions from the haul route emission source assumed in the NPI emission estimation calculation method.

The analysis presented in Table 7-5 confirms that for coal haul and overburden haul routes, application of calculated emission rates that account for measured surface moisture contents results in emissions estimates that are closest to the measured emission rates. It is also concluded that, where an emission calculation accounts for material moisture content, then application of additional corrections to take account of watering for particulate control may result in significant underestimates of $PM_{2.5}$ emissions in practice. This has significant implications for dispersion modelling of proposed mines, as assumptions relating to a 75 % reduction in haul route emissions where watering is adopted as a control technique is likely to over-estimate $PM_{2.5}$ reductions achievable in practice based on current haul route watering practices.

Table 7-5: Analysis of Moisture Control v/s Emission Rate Assumptions

Mine Activity	NPI Emission Estimate $PM_{2.5}$[a]	Average Study - Measured $PM_{2.5}$[b]	Difference (measured minus estimate)	Ratio (measured to estimate)	Units
Coal Haul Road:					
Calculated (accounting for surface moisture)	0.0511	0.0715	0.0204	1.39	kg/VKT
Calculated, 75 % control	0.0128	0.0715	0.0587	5.59	kg/VKT
Default (uncontrolled)	0.2538	0.0715	-0.1823	0.28	kg/VKT
Default, 75 % control	0.0635	0.0715	0.0080	1.13	kg/VKT
Overburden Haul Road:					
Calculated (accounting for surface moisture)	0.0303	0.0336	0.0033	1.11	kg/VKT
Calculated, 75 % control	0.0076	0.0336	0.0260	4.43	kg/VKT
Default (uncontrolled)	0.2538	0.0336	-0.2202	0.13	kg/VKT
Default, 75 % control	0.0635	0.0336	-0.0299	0.53	kg/VKT

[a] NPI Default TSP emission rate multiplied by NERDDC 0 µm – 2.5 µm size fraction.[b] With haul route watering controls at an average rate of 2.7 water cart passes per hour occurring during the measurements.

7.6 CONCLUSIONS

The investigation of particulate control efficiencies from haul route surface watering indicates that lower control efficiencies may be achieved in practice compared to the estimation method based on moisture ratios developed by the US EPA. On this basis, the empirical method developed by the US EPA is not appropriate for calculation of surface watering control efficiencies for Australian conditions. Similarly, the current NPI default control efficiency for haul route watering overestimates the actual control efficiency based on current watering regimes in Australian coal mines.

Overall, the necessity of completing surface watering at a rate of four times per hour has been demonstrated as necessary to achieve emission reductions of 50 % or more under Australian conditions.

8 CONCLUSIONS AND KEY FINDINGS

8.1 CHARACTERISTICS OF COAL MINE PARTICULATE EMISSIONS

The empirical data from the various research elements of this PhD programme demonstrate that the particle size fraction varies for different mining activities. Where mechanical abrasion processes are significant, as in the case of drilling, the particle size is skewed toward the finer fraction. The analysis of particulate composition presented in Chapter 4 demonstrates that the particle emissions are predominantly sourced from naturally occurring geological material. Coal comprises less than 13% of the overall emissions. This is significant, as clearly the region in which the mine is located is the key influence on the characteristics of the particulates emitted to the atmosphere. Thus, it is concluded that the type or quality of coal being mined is of much lesser importance in determining the characteristics of the emitted particles than the local geological characteristics and soil types. Furthermore, the mine activity has the greatest influence on the particulate emission characteristics.

This is also significant in the context of the response of Australian communities living in close proximity to coal mines. Typically, particulate emissions from coal mines in Australia are perceived by local communities to be dominated by coal particulate emissions and, therefore, to pose a greater health risk than non-coal related emissions from farming and other rural activities. These issues were raised in submissions to a parliamentary enquiry relating to air quality impacts on health (Commonwealth of Australia 2013).

Key Finding

Regional soil and geology are a significant determinant of particulate characteristics.

The boundary and offsite sampling described in Chapter 4 demonstrates that the relative proportion of fine particulates in the sample increased with distance from the source. The coarse fraction was a more significant proportion of total suspended particulates close to the source of emissions. This highlights the potential for the fine fraction of emissions from a mine to have a more significant influence external to the mine and in the local region, than the coarser fractions which are likely to deposit in close proximity to the source of emissions. When this is considered in the context of the greater health risk associated with the $PM_{2.5}$ size fraction relative to TSP and PM_{10}, it is clear that development of accurate emission estimation techniques for fugitive $PM_{2.5}$ releases from coal mines is fundamental to the accurate prediction of potential mine related health impacts in local communities.

Key Finding

At the boundary of a mine the percentage of fine particulates is close to 50 % lower than in the typical urban environment, the fine particulates have potential to be transported over long distances, hence the mine can remain a significant source of fine particulates in the region.

Comparison of the size fractionation exhibited by the sampling datasets presented in Chapter 4 to that adopted in current Australian emissions estimation methods, confirms that the two are largely similar. However, the sampling dataset exhibits significant differences when compared to the size fractionation presented in the United States Environmental Protection Agency (US EPA) methodology. The USEPA methodologies are commonly adopted where local or regional specific emission rates have not been developed. Hence the research data collated through this PhD programme has demonstrated that development of region-specific emission estimation techniques for PM_{10} and $PM_{2.5}$ from open cut coal mines is necessary to allow accurate prediction of particulate emissions to inform regulatory decisions and for use in modelling predictions.

Key Finding

The size fractionation of particulate emissions from different regions can vary significantly, hence development of regional emission datasets is necessary to improve accuracy of modelled concentrations

8.2 Key Influences on Particulate Emission Rates

The empirically data compiled through the practical aspects of this research has identified that meteorological conditions are a key influence on particulate emission rates. The results presented in Chapter 5 show that the calculated emission rates for Central QLD are significantly lower than for the Hunter Valley and South-East Queensland due to the local meteorological conditions. For Central QLD, there is a very low occurrence of wind speeds above the threshold of 5 m s^{-1} – the threshold wind speed for surface erosion - and higher annual rainfall than the other test sites. This comparison demonstrates that local wind speed and rainfall are a key determinant of overall annual emission rates from surface erosion.

Key Finding

Adjusting for local meteorological conditions significantly affects mining particulate emissions estimates.

8.3 Appropriate Particle emission rates for Australian Coal Mines

Chapters 5 and 6 present empirically determined emission rates for TSP from open erodible surfaces and $PM_{2.5}$ emission rates for a range of activities in Australian open cut coal mines. A summary of the

Key Finding

Emission estimation equations have been determined for calculation TSP emission rates for wind speeds in the range 5 m s^{-1} to 12 m s^{-1}. This allows region specific emission rates to be developed based on local meteorological conditions.

emission rates determined for TSP from open surfaces (Chapter 5) and for $PM_{2.5}$ (Chapter 6) for different activities and three regions of Australia are reproduced below in Table 8-1 to Table 8-3.

Table 8-1: TSP Emission Rates by Activity (With Haul Route Watering Controls)

	5 m s^{-1}	**6.7 m s^{-1}**	**8.9 m s^{-1}**	**10.8 m s^{-1}**	
Haul Road	1.49	2.34	4.56	5.25	3.57
Spoil	1.55	3.66	2.72	6.27	3.33
Rejects Stockpile	2.77	1.72	2.06	3.98	2.90
Dragline Walk Road	3.19	3.79	-	7.02	4.67
Tailings	-	-	-	0.01	n/a
Dragline Stockpile	-	-	3.43	-	n/a

Table 8-2: TSP Emission Rates by Region (Without Haul Route Watering)

Mine	**Emission Rate (kg ha^{-1} hr^{-1}) for Specified Wind Speed**				**Average Emission Rate (kg ha^{-1} hr^{-1})**
	5 m s^{-1}	**6.7 m s^{-1}**	**8.9 m s^{-1}**	**10.8 m s^{-1}**	
Central QLD	2.15	2.18	3.42	5.38	2.98
Hunter Valley	0.41	2.17	4.28	4.94	3.57
QLD South	2.05	4.50	2.76	4.29	3.28
Average:	1.54	2.15	3.49	4.87	3.28

Table 8-3: $PM_{2.5}$ Emission Rates by Region (With Haul Route Watering)

Mine Activity	**QLD Central**	**NSW Hunter Valley**	**Overall Average**	**Units**
Coal Haul Road	0.0272	0.1912	0.0715	kg/VKT
Overburden Haul Road	0.0224	0.0411	0.0336	kg/VKT
Drilling	0.0437	0.3000	0.1691	kg/VKT
Dragline	0.0048	0.0113	0.0074	kg/hole
Overburden Loading	0.0004	0.0011	0.0009	kg/BCM[a]
Overburden Dumping	0.0003	0.0005	0.0003	kg/t
Coal Loading	0.0007	0.0002	0.0006	kg/t
Coal Dumping	0.0067	0.0003	0.0041	kg/t

[a] *Bank Cubic Metre – one metre cubed corrected for the density of the material.*

The empirically determined emission rates represent a significant expansion of our current knowledge relating to particulate emissions for different regions of Australia and for different mine activities. Furthermore, the emission rates determined in this study demonstrate that regional differences between emission rates may be significant for specific activities.

Key Finding

New region and activity specific TSP and $PM_{2.5}$ emission rates have been determined for Australian open cut coal mines.

Ancillary data collected in conjunction with the research presented in this thesis have been used to validate existing control efficiency assumptions as adopted in the NPI emission estimation handbooks. This has determined that typical haul route watering rates are too low to achieve the minimum control efficiency that is currently adopted for watered haul routes. Furthermore, the research has demonstrated that the empirical formulae developed by the US EPA for calculation of watering control efficiencies at varying % surface moisture is not suitable for adoption for Australian mining environments.

Key Finding

The US EPA moisture ratio approach for estimating haul route watering control efficiency is not representative of Australian conditions.

Key Finding

Typical haul route watering rates at Australian mines achieve a control efficiency of 27 %. The water application rate is well below the rate defined in the NPI for achieving 50 % particulate emission control.

The emission rates at the intermediate wind speeds are consistent with this and indicate that the surface crusting is beneficial at a wind speed of 6.7 m s^{-1}. Surface crusting no longer provides a benefit at 8.9 m s^{-1} as emission rates at this wind speed were similar to emission rates for other surfaces at this wind speed.

Key Finding

Surface crusting reduces particulate emissions from surfaces at wind speeds up to 6.7 m s^{-1}.

8.4 RESEARCH CONTRIBUTION TO KNOWLEDGE

The body of research presented in this thesis has provided new knowledge in relation to a range of aspects of fine particulate emissions from Australian coal mines. The new knowledge can be summarised as follows:

- Regional soil and geology are a significant determinant of particulate characteristics
- At the boundary of a mine the % of fine particulates is close to 50 % lower than in the typical urban environment. However, the fine particulates have potential to be transported over long distances, hence the mine can remain a significant source of fine particulates in the region.
- The size fractionation of particulate emissions from different regions can vary significantly, hence development of regional emission datasets is necessary.
- Adjusting for local meteorological conditions significantly affects mining particulate emissions estimates.
- New region and activity specific TSP and $PM_{2.5}$ emission rates have been determined for Australian open cut coal mines
- Emission estimation equations have been determined for calculating TSP emission rates for wind speeds in the range 5 m s^{-1} to 12 m s^{-1}. This allows region specific emission rates to be developed based on local meteorological conditions.
- Typical haul route watering rates at Australian mines achieve a control efficiency of 27 %. The water application rate is well below the rate defined in emission inventories for achieving 50 % particulate emission control.
- The US EPA moisture ratio approach for estimating haul route watering control efficiency is not representative of Australian conditions.
- Surface crusting reduces particulate emissions from surfaces at wind speeds of up to 6.7 m s^{-1}.

8.5 OVERALL CONCLUSIONS

The research presented in this thesis makes an original and extensive contribution to the current research relating to particulate emissions from open cut coal mines.

Overall, the research expands on our current understanding of particulate sources in open cut coal mines with respect to size fractions, chemical and physical composition, and emission rates for a range of mining activities. Key parameters that influence particulate emission potential, including silt content, moisture content and the effects of surface crusting have also been determined. The accuracy and applicability of the currently adopted haul route watering control efficiency calculation has been explored and found to significantly over estimate actual control efficiencies in practice for Australian coal mines.

A key contribution of the research relates to improving our current understanding of particulate emissions from erodible surfaces. This has involved the development of new empirical emission rate equations for TSP at a range of wind speeds. These data will allow regional meteorological conditions to be considered in developing site specific particulate emission estimates in the future, thus providing a potentially significant improvement in the representativeness of future emissions estimates.

Finally, the research has also developed new emission rates for $PM_{2.5}$ from a range of open cut coal mine sources, based on empirical data that was hitherto unavailable for this size fraction.

8.6 LIMITATIONS AND RECOMMENDATIONS FOR FURTHER RESEARCH

One of the limitations of the study was that the sampling programme discussed in Chapter 5 was not able to determine PM_{10} emission rates with a suitable degree of accuracy, hence these data have not been published. Future research could address this gap in our knowledge. Further specific limitations that affected the amount of data collated during each monitoring programme related to access and operational limitations. Despite the field work elements of the research being completed over many months in total, the resultant datasets represent only a small fraction of the time spent on site. While the data obtained is considered representative, additional data points may improve the statistical significance of the datasets and reduce the standard deviation identified for the calculated emission rates.

Overall, the research programme has highlighted the significant variability associated with empirical measurements of fugitive particulate emissions. This is of particular relevance when risk assessments of existing or proposed future mines are being completed. This feature of the emission data may introduce a greater degree of uncertainty to predictive modelling, and this should be considered in any decision-making processes that rely on these data. This is of particular relevant to environmental regulators when policy documents and emission inventory methodologies are being developed or updated.

In terms of recommendations for additional future research, a specific aspect of the research presented in this thesis where additional data or field research could broaden the findings of the study is sampling of geological conditions and emissions rates for key international mining regions. This would broaden the research in terms of the link between natural soil conditions and emission characteristics. Finally, given the importance of watering as a control mechanism in open cut coal mines, expanding the limited data presented in this thesis to represent a broader range of climatic conditions, seasons and different geological areas would further expand knowledge of the effectiveness of specific haul route watering regimes. This additional research, if completed both in Australia and overseas, would further improve the suitability of emission factors and hence provide for more accurate emissions estimation for the key mining regions of the world.

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Appendix A: Paper 1 – Characterisation of Particulate Emissions from Open Cut Coal Mines: Towards Improved Emissions Estimations

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CHARACTERISATION OF PARTICULATE EMISSIONS FROM AUSTRALIAN OPEN CUT COAL MINES: TOWARDS IMPROVED EMISSION ESTIMATES

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Abstract

Given the significance of mining as a source of particulates, accurate characterisation of emissions is important for the development of appropriate emissions estimation techniques for use in modelling predictions and to inform regulatory decisions. The currently available emission estimation methods for Australian open cut coal mines relate primarily to total suspended particulates and PM_{10}, and limited data is available relating to the $PM_{2.5}$ size fraction. To provide an initial analysis of the appropriateness of the currently available emission estimation techniques, this paper presents results of sampling completed at three open cut coal mines in Australia. The monitoring data demonstrates that the particulate size distribution varies for different mining activities, and that the region in which the mine is located influences the characteristics of the particulates emitted to the atmosphere. The proportion of fine particulates in the sample increased with distance from the source, with the coarse fraction being a more significant proportion of total suspended particulates close to the source of emissions. In terms of particulate composition, the results demonstrate that the particulate emissions are predominantly sourced from naturally occurring geological material, and coal comprises less than 13% of the overall emissions. The size fractionation exhibited by the sampling datasets is similar to that adopted in current Australian emissions estimation methods, but differs from the size fractionation presented in the United States Environmental Protection Agency methodology. This indicates that development of region specific emission estimation methods is necessary if accurate emissions estimations are to be completed. Development of region specific emission estimation techniques for PM_{10} and $PM_{2.5}$ from open cut coal mines is necessary to allow accurate prediction of particulate emissions to inform regulatory decisions and for use in modelling predictions.

Introduction

Airborne particulate matter arises from a range of sources, natural and anthropogenic. The relationship between particulates and potential health impacts is well documented (Laden et al, 2000) and most health related criteria and goals are currently based on exposure to a specific mass concentration of particulates for given size fractions. Whilst particulate size and mass concentration are key metrics, the chemical composition of particulates is also an important characteristic in determining potential community health impacts.

Pollution emissions inventories are important tools in guiding regulatory policies and determining the most effective strategies for managing emissions that may result in health impacts (Huertas, Huertas et al, 2012, Weng et al., 2012). Emissions inventories are developed using emission estimation tools, such as monitoring and calculation techniques. The accuracy and suitability of these estimation techniques is fundamental to the development of accurate emissions inventories for use in air quality decision making frameworks that are informed by emission inventories.

In Australia, open cut coal mines are currently the most significant national source of particulate emissions. The National Pollutant Inventory (NPI) for Australia (Environment Australia, 2012 & 2015) identifies that approximately 25 % of emissions of particulates from industrial sources arise from coal mining, as illustrated in Figure 1 (Environment Australia, 2017). The significance of mining as a source of particulates in Australia is unsurprising, given that mineral exports generated 32 % of total export income for Australia in 2014 (DFAT, 2017). The need for source specific data to assist in the overall management of these sources of emissions, has been identified (Weng et al, 2012).

The particulate emission techniques adopted in Australia are based on data from the USA from the 1980's, and data from Australian sampling of specific mine sources completed in the 1990's. Source specific $PM_{2\,5}$ sampling data is very limited for open cut coal mines, hence this size fraction is estimated on the basis of assumed particulate size distributions for different activities. It has been demonstrated that application of the USA emissions estimation methods to other regions may result in significant over

estimates of emissions, and the lack of source specific emissions data for $PM_{2.5}$ is a significant gap in the current understanding of particulate emissions from open cut coal mines.

Particulate emission estimation techniques rely on an accurate understanding of particulate characteristics that are relevant to the nature of emissions arising from mining activities. This paper provides a preliminary analysis of some of the key characteristics associated with open cut coal mining based on sampling data for three Australian open cut coal mines. These data are considered in the context of the particulate characteristics adopted in the current mining particulate emission techniques, to provide an indication of the suitability of the currently adopted methodologies, and to identify gaps in our knowledge and understanding of this issue.

Estimation of Particulate Emissions from Open Cut Coal Mines

Preparation of emissions inventories relies on a range of methodologies for estimation of emissions from specific sources. These methods include mass balance calculations, engineering calculations, sampling or direct measurements, and application of emission factors. Of these, application of emission factors is generally adopted where insufficient data are available to allow the completion of mass balance or engineering calculations, or where direct sampling methods for a specific site are not practicable.

Due to the cost associated with direct sampling for the development of site specific emission factors, emissions estimation is the tool most frequently adopted where emissions estimation of mining particulates is required. The United States Environment Protection Agency (US EPA) publishes a series of emissions factors in the AP42 document for a range of pollutant emissions from various sources, including mining operations (US EPA 1995, updated 1998). The US EPA presents emission estimation equations for TSP (< 30 μm) and PM_{15} based on research completed in the late 1970s and 1980s in AP42 Chapter 11.9. To allow estimation of the PM_{10} and $PM_{2.5}$ emissions from these equations, scaling factors are provided. The US EPA identifies a moderate to low quality rating (C or D) for the PM_{10} and $PM_{2.5}$ mining emission estimates completed using these factors, and reduces these quality ratings further

where site specific data relating to factors such as moisture and silt content are not available. Overall, the US EPA note that there is a tendency for over prediction of particulate matter impacts for PM_{10} when the AP42 emissions estimation methods are adopted. In the absence of alternative methods, the US EPA note that the emission factors may be used for non-regulatory applications with caution and in the context of the likely limitations of the resultant estimates. The resource and technical challenges that are faced when trying to improve the quality of particulate emissions from fugitive dust sources in open cut coal mines are recognised by the US EPA in the fifth edition of AP42 (US EPA, 1995).

The US EPA AP42 mining emission factors are commonly referenced both in the US and elsewhere, despite the identified limitations. A number of studies have investigated the applicability of the US EPA emission factors to regions outside the United States. Adoption of the US EPA emission rates in India has been identified as inappropriate, due to the differences in mining site practices, geological and climatic conditions (Chaulya et al 2002). Even where the emissions estimation methods are applied in mines in the country for which they were developed, the uncertainties can be large. Application of the AP42 equations to mines in the US led to underestimation by a factor of up to 13, through to overestimation by a factor of 1.5 for specific activities (Huertas, Huertas and Diaz, 2012). In 2012 Huertas and colleagues further identified that studies currently related to PM_{10} emissions for open cut coal mining are not available, and that given that PM_{10} is more harmful to health than TSP there is a need for standardized PM_{10} emissions estimation methodologies (Huertas et al, 2012).

In an Australian context, the currently adopted emission estimation methods for TSP and PM_{10} for open cut mining are based on both the US EPA AP42 emission equations, and monitoring studies completed in Australian in 1983 and 1988. The quality rating applied to the emission factors presented in the NPI Manual for Mining (Environment Australia, 2012) differ for each mine source. Where local data is available to verify the emissions equations, a higher quality rating has been applied. No emission rates or scaling factors for $PM_{2\,5}$ are provided in the Australian NPI handbook for mining and the scaling factors provided in the US EPA AP42 are commonly adopted.

Materials and Methods

Three sampling techniques were employed to undertake particulate monitoring at three Australian open cut coal mines: one mine in Queensland (QLD) and two in New South Wales (NSW). Firstly, a low volume gravimetric technique was adopted to measure the longer term concentrations beyond the boundary of the two mines in NSW for two particulate size fractions – PM_{10} and $PM_{2.5}$. The sampling involved use of the Minivol Portable Air Sampler (Airmetrics, Springfield, OR USA). The Minivol samplers were fitted with Millipore PTFE filters and size selective inlet impactors for PM_{10} and $PM_{2.5}$. The samplers were calibrated using a flow rotameter prior to and at the end of each sample. Field blanks were analysed to confirm the accuracy of the gravimetric analysis and impactor plates were cleaned and greased on a weekly basis in accordance with the manufacturers recommendations. Filters were weighed pre- and post- sampling in a temperature and humidity controlled laboratory using a digital balance with a resolution of 1 µg. A minimum sample duration of 24 hours was adopted, and was extended to 3 days for the majority of samples, to improve the sensitivity of the gravimetric analysis.

For the majority of the test locations, sampling was completed for a period of 7 days or more for PM_{10} and $PM_{2.5}$. The overall average concentrations in each size fraction were subsequently determined to allow comparison of the size fractionation of the particulate material. For the mines in NSW, the Minivol samplers were co-located with total suspended particulate (TSP) high volume samplers operated by the mining companies, to allow for comparison of the particulate size fractionation data with overall TSP concentrations.

As mining operations tend to vary in terms of activity type and location on a daily basis and, for some activities, on an hour by hour basis, the low volume sampling technique was impractical to use for source emission characterisation. As an alternative, a real time optical particulate monitor (Model 1.105, GRIMM Aerosol Technik, Ainring, Germany) was used allowing for sampling in a particle range 0.3-20µm over short time periods when the activity of interest was occurring.

The third method of characterization involved microscopy analysis. A selection of the PM_{10} and $PM_{2.5}$ filter samples collected with the low volume gravimetric sampling method external to the mine boundaries were analysed. Elemental composition analysis of individual particulates was undertaken using energy-dispersive x-ray spectrometry (EDS). Due to image and analytical resolution limitations, particulates with a diameter of less than 1 μm in diameter were not considered in the analysis. The elemental composition data was determined for silica, carbon, soil, and 'other'. The 'other' category includes particulates such as salt (NaCl), and compounds containing two or more of the elements sodium, magnesium, sulphur, chlorine, calcium, iron and zinc.

Sampling Limitations

Optical particulate counters such as the Grimm rely on light scattering techniques to identify particle number and an estimate of particulate size. These instruments apply an estimate of particle density to extrapolate from the particle size and number to an estimated concentration of particulates in the air stream. The response of optical instruments such as the Grimm is dependent upon the size distribution, shape and refractive index of the particulates. Therefore, the measured concentrations may be dependent upon the shape and size range of particulates experienced in a given locality, which may affect the accuracy of the particulate measurements (Renard, Thaury et al, 2010). Furthermore, the Grimm counter – in common with similar optical based particulate monitoring units - typically operate at low flow rates that are insufficient to carry particulates >35 μm into the instrument through the sampling inlet. As the TSP size fraction includes particulates >35 μm in diameter, the Grimm sampler may underestimate TSP concentrations and this is a further limitation of the method.

Due to these limitations and the varying distances to operations during the sampling, direct comparison of the optical derived source concentration data is not meaningful. Therefore, these monitoring data have been considered in terms of mass percentage in each size fraction only.

Results

The results of the particulate sampling and the size distributions for the near source sampling at each of the three mines are presented in Figures 2 to 4. The Grimm optical sampler particulate concentration data is the basis for determining the particulate size distribution. The average particle size distributions for each mine are presented in Table 5 indicating that the average particle size distributions for two NSW coal mines are almost identical. The average particle size distribution for the QLD mine indicates a higher proportion of PM_{10} material overall, however the $PM_{2.5}$ size fraction is similar to that observed for the NSW mines. This suggests that regional geological conditions or other local features may be influencing the particulate size distribution for these mining activities.

Figure 6 presents the results of the sampling completed external to the boundary of the two mines in NSW. The estimated separation distance to the nearest active pit at each mine is also identified. These results demonstrate a significantly different particle size distribution occurs external to the mine boundary relative to the near source sampling data (Figure 5). For the sampling completed external to the mine, TSP is comprised of a higher proportion of PM_{10} at an average of 55 % compared to an average of 40 % observed for the near source data. Of greater significance is the proportion of $PM_{2.5}$, at 25 % of TSP, compared to an average of 2 % for the near source PM_2 measurements. It is likely that, because finer particulates are transported over long distances from the source, the relative proportions of fine dust tend to increase with distance from a specific source (Cattle, Karl et al, 2012).

The size fractions also show some variability for the different mine activities, with a maximum $PM_{2.5}$ proportion of 11 % recorded for the coal preparation plant at Mine 3 and a maximum PM_{10} proportion of 61 %. The highest PM_{10} proportions were measured downwind of coal preparation plant (Mines 1 and 3) and drilling (Mine 2) and range from 61 – 66 %. This demonstrates that adoption of an average particulate size distribution for all mining activities could significantly under estimate PM_{10} emissions from specific activities, as some operations generate higher than average PM_{10} fractions.

Table 1 presents a comparison of the proportion of PM_{10}/TSP for different activities at each of the mines, and also presents the size fractions recommended by the US EPA and for the Australian NPI emissions equations. This comparison demonstrates that there is reasonable consistency between the NPI size fractions and the measurement data for the various mine activities considered in the sampling. US EPA size fraction factors are available for only two of the sources considered. In both cases, were the US EPA factors adopted, PM_{10} emissions would be estimated at more than double the emission rate than is demonstrated by the monitoring data and indicated by the NPI emission factors.

Emission estimation factors for $PM_{2.5}$ are not provided in the 2012 NPI Mining handbook, as reporting of fugitive $PM_{2.5}$ emissions is not currently required. Therefore, Table 2 presents a comparison of the measured $PM_{2.5}$/TSP proportions with the $PM_{2.5}$ scaling factors defined by the US EPA for specific mining activities. This comparison indicates that the measured dataset provides a similar proportion of $PM_{2.5}$ as defined in the US EPA scaling factors.

The results of the EDS compositional analysis are presented in Table 3 for a selection of the low volume filters. In terms of the source of particulates, these results indicate that on average 62 % of the measured particulate loading is from a soil or clay source. These particulates could result from overburden sources, rehabilitation activities at the mine or erosion of open surfaces. Given the location of the samplers external to the mine, on farmland, there may also be a contribution from agricultural operations and erosion of farmland areas external to the mine. There is no clear difference between the particulate composition of the PM_{10} size fraction and the $PM_{2.5}$. This is not unexpected, given that the majority of the particulates in these size ranges are likely to be sourced from local geological material.

An assessment of the likely coal content of the samples can be made from the percentage presence of carbon in the particulates. Although a range of other minerals, including silica, may be present in coal, these minerals may also be present in the local geologically sourced material. Hence, percentage carbon provides the most appropriate indicator of the likely significance of coal in the particulate samples. The highest percentage coal composition of 31 % was collected 550 m downwind of the coal

processing plant at Mine 1, QLD. The average carbon composition for all samples was 16 %, thus indicating that coal comprises a relatively small proportion of the overall particulate composition. It is noted that the carbon fraction will contain material from a range of sources, including vegetation and insects, hence the % carbon represents the maximum expected proportion of coal in the sample.

Discussion

The results of the sampling confirm that particulate size distributions vary for different mine activities and, to a lesser extent, may also differ between different geographical areas. Of greater significance, comparison of the measured particle size distributions confirms that the size fraction scaling factors documented by the US EPA in the AP 42 emissions estimation manual for surface coal mine are significantly different, and if applied in an Australian mining context would over-estimate PM_{10} emissions. This highlights the need for region specific emission inventories to be developed and is consistent with the observations of the US EPA regarding the potential for over estimation of PM_{10} emissions where the AP 42 factors are adopted (US EPA, 1995). On this basis, it can be concluded that a degree of caution must be adopted when applying particulate emissions estimate methods derived on the basis of a specific region, to another region or overseas location.

Comparison of the measured PM_{10} emission fractions with the emissions estimations presented in the Australian NPI handbook for mining shows reasonable consistency for PM_{10} emissions. This indicates that the currently adopted PM_{10} emissions estimates in Australia are more appropriate than application of the factors adopted in the United States. Some variability remains, and for a number of mining activities high quality emission factors are not available due to the absence of local fugitive emission sampling data. Overall, it is considered appropriate to develop improved PM_{10} emission factors for those open cut coal mining activities where high quality data based on emissions sampling completed in Australian mines is not currently available. This is consistent with the conclusions of a study published by the Western Australian Department of Environment in 2005 (WA DEP, 2005).

For $PM_{2.5}$, the measurement data is consistent with the scaling factors currently recommended by the US EPA. However, recommended $PM_{2.5}$ emission estimation methods for fugitive releases from Australian coal mines are not currently available as there is no requirement to collate and publish these emissions. Given the variability identified between the US EPA AP 42 emissions data for PM_{10} when compared to Australian emissions, it is considered appropriate to develop Australian emissions estimation methods for $PM_{2.5}$ from open cut coal mining activities. Furthermore, the measurement dataset demonstrates that the relative significance of $PM_{2.5}$ related mine emissions as a proportion of the overall particulate matter increases with distance from the source. When this is considered in the context with the greater health risk associated with the $PM_{2.5}$ size fraction relative to TSP and PM_{10}, it is clear that development of accurate emission estimation techniques for fugitive $PM_{2.5}$ releases from coal mines is a significant gap in our current knowledge.

In terms of the most significant source of particulate emissions from open cut coal mines, the compositional analysis confirms that local geological sources dominate. This is consistent with the observations of Mudd (2007 and 2009) with respect to the increasing proportion of over burden per unit of extracted coal, and the findings of Kaufman et al (2002) and Deshmukh et al (2012).

Conclusions

Findings from the preliminary investigations presented in this paper demonstrate that the widely adopted US EPA emission estimation equations for PM_{10} may result in a significant overestimate of actual emissions for a range of open cut mine related activities. The variable data quality for the available PM_{10} emission estimation techniques has also been highlighted, and it is concluded that it is appropriate to develop improved PM_{10} emission factors for open cut coal mining in Australia where high quality data is not currently available. In relation to $PM_{2.5}$ emissions from open cut coal mines, the absence of Australian emission estimation techniques is a significant current gap in our knowledge relating to particulate emissions from open cut coal mining.

On the basis of this preliminary investigation, further research is needed to develop accurate emission factors for Australian open cut coal mines, to provide for more accurate quantification and prediction of the impacts of mining related PM_{10} and $PM_{2.5}$, which are a key input to environmental impact assessments and regulatory policy decisions.

Acknowledgements

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Tables

Table 1. Comparison of PM_{10}/TSP Particulate Size Fractions – US EPA AP42, NPI and Measured

Activity	NPI	US EPA	QLD	NSW 1	NSW 2
Draglines (on overburden)	43	75	3	23	-
Excavators/Shovels/Front-end loaders (on overburden)	47	-	28	41	30
Excavators/Shovels/Front-end loaders (on coal)	48	-	56	41	47
Bulldozers on coal	32	75	-	-	28
Trucks (dumping coal)	42	-	-	45	37
Drilling	52	-	-	63	29

Table 2. Comparison of $PM_{2\,5}$/TSP Particulate Size Fractions – US EPA AP42 and Measured

Activity	US EPA	QLD	NSW 1	NSW 2
Draglines (on overburden)	1.7	<1	2	-
Bulldozers on coal	2.2	-	-	2
Bulldozer on material other than coal	1.05	-	-	2
Graders	0.31	<1	-	-

Table 3. Chemical Composition of Particulates

Size Fraction	Monitoring Positions	Percentage (%) of particulates in each category				
		Clay or soil	Iron	Silica	Other salt/ mineral	Carbon
$PM_{2.5}$	Mine 1 – due west of active pit, 50 m to haul road	77	0	12	0	12
$PM_{2.5}$	Mine 1 – 550 m downwind of processing plant	54	0	12	4	31
$PM_{2.5}$	Mine 1 – background position	63	3	19	5	10
$PM_{2.5}$	Mine 1 – 1 km downwind of processing plant	51	2	15	10	22
PM_{10}	Mine 1 – 1 km downwind of processing plant	67	1	9	17	6
$PM_{2.5}$	Mine 2 – approximately 4 km from active pit	42	5	16	11	26
PM_{10}	Mine 3 – approximately 1 km to active pit	71	4	11	1	13
$PM_{2.5}$	Mine 3 – approximately 1 km to active pit	72	3	8	13	5
	Average:	**62.1**	**2.3**	**12.8**	**7.6**	**15.6**

Figures

Figure 1. Estimated Australian PM_{10} and $PM_{2.5}$ Emissions by Industry Type as Percentage of Total Emissions in 2014/2015

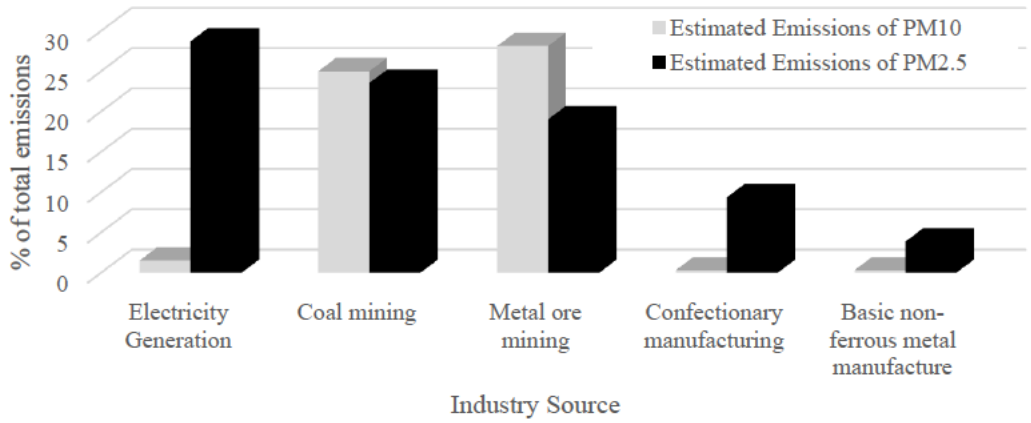


Figure 2. Particle Size Distribution - Mine 1, QLD (Source Sampling)

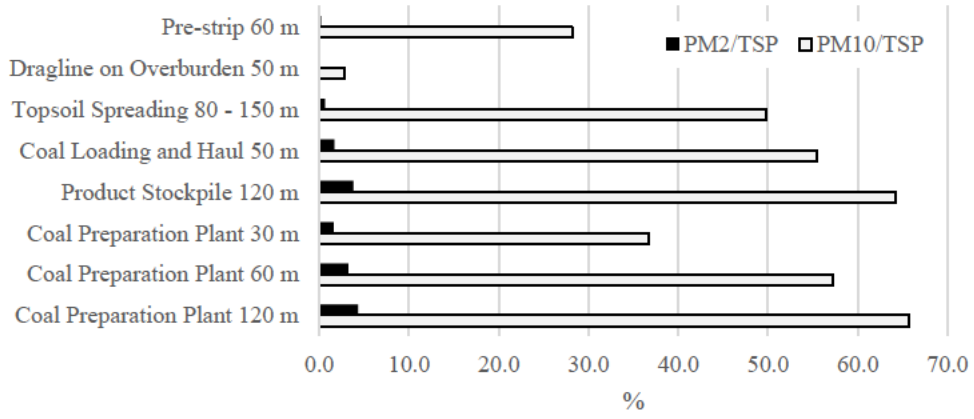


Figure 3. Particle Size Distribution - Mine 2, NSW (Source Sampling)

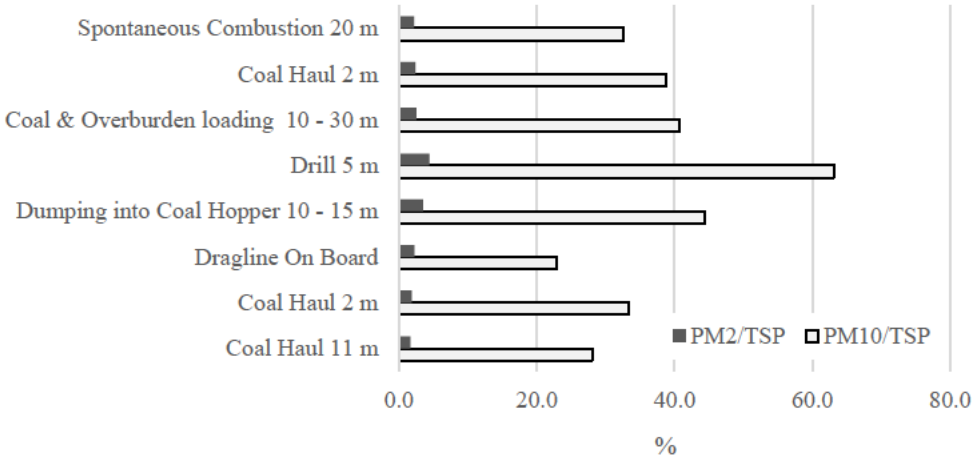


Figure 4. Particle Size Distribution - Mine 3, NSW (Source Sampling)

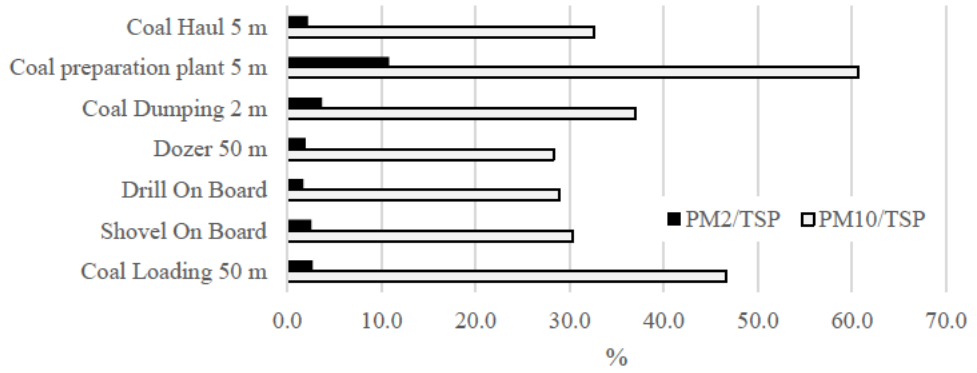


Figure 5. PM_{10} and $PM_{2\,5}$ (Source Sampling)

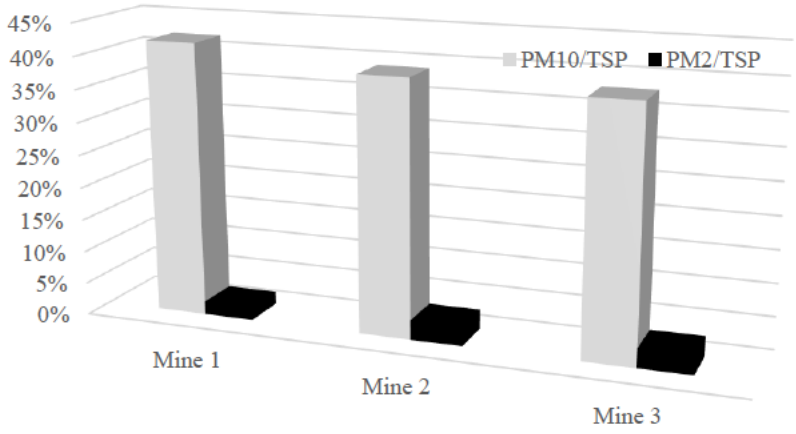
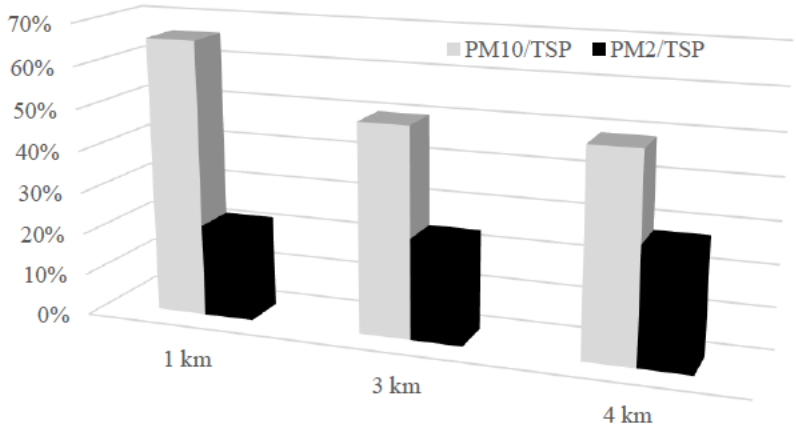


Figure 6. PM_{10} and $PM_{2\,5}$ sampling at distances from active pit



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Appendix B: Paper 2 – Particulate Emission Rates for Open Surfaces in Australian Open Cut Black Coal Mines

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Research article

Particulate emission rates for open surfaces in Australian open cut black coal mines

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ABSTRACT

Accurate estimation of particulate emissions is fundamental to assessing the air quality risks posed by existing and proposed open cut black coal mines. The currently available emission estimation techniques are based on a limited range of empirical studies, and the need for additional research and development of activity and region specific particulate emission estimation methods has been recognised. This paper presents the results of empirical testing of particulate emission rates for open area surfaces in open cut black coal mines in three Australian regions. The emission rates are provided for specific wind speeds, thus allowing adjustment of emission rates for prevailing meteorology in these regions. The influence of surface watering as a control technique is also considered. Particulate emission rates are presented for a range of coal mining sources, and for specific wind speeds. Comparison of the emission rates with the existing published research confirms the emission rates are consistent with current approaches. This research significantly expands our current understanding by presenting emission rates for a range of open area sources, for specific wind speeds and surface watering controls. This more detailed emission estimation dataset provides for adjustments to default particulate emission rates to allow site specific data to be integrated into emissions estimation. This will result in more accurate emissions estimates for existing and future projects and reduce the potential for significant over- or under-estimation of particulate emissions from surface sources in open cut coal mines.

1. Introduction

Pollution emissions inventories are important tools in guiding regulatory policies and determining the most effective strategies for managing emissions across industry sectors. Having established the actual or predicted impact on the atmospheric environment of a specific source or project, a mitigation strategy can be developed (Ghose, 2007a). The availability of representative particulate emissions data and inventories is fundamental to accurate environmental risk assessment and provides the foundation for analysis of the environmental fate of these releases.

Having identified the appropriate emissions data, atmospheric dispersion modelling is the approach commonly adopted to allow prediction of expected concentrations of particulates external to an industrial source, and the overall environmental risk (Ghose, 2007b; Huertas et al., 2012, 2014). Several researchers have commented on the significance of emissions data inputs to atmospheric dispersion modelling as a key source of error or modelling uncertainty as follows. In an analysis of the performance of the atmospheric dispersion models AERMOD and CALPUFF for an open pit quarry situation, Tartakovsky and colleagues identified that due to severe uncertainties in the model inputs, it is necessary to complete extensive model iterations when assessing model performance (Tartakovsky et al., 2013). Further, Holnicki and Nahorski (2015) identified that when basing policy decisions on atmospheric dispersion modelling predictions, it is critical that the uncertainty associated with the methodology is clearly understood. In particular, air pollution emission data from industry, traffic and municipal sources were identified by Holnicki as a primary cause of modelling uncertainty in urban areas, with a potential for negative environmental and health consequences.

Specific issues associated with the imprecision of emissions inventories used for atmospheric dispersion modelling have also been identified (Holnicki and Nahorski, 2015) and, with specific reference to the mining industry, Chakraborty et al. (2002) demonstrated that the average accuracy between measured and calculated emission rates varies significantly ranging from 77% to 80%. Overall, there is a paucity of emissions concentration data for accurate source characterisation to allow modelling of downwind hazards, as identified by

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Abbreviations

AP42	Compilation of Air Pollutant Emission Factors (US EPA)
NPI	National Pollutant Inventory (Australia)
NSW	New South Wales (Australia)
PM_{10}	Particulate matter (PM) with an aerodynamic diameter of 10 micrometres or less
PTFE	Polytetrafluoroethylene
QLD	Queensland, Australia
RPM	Revolutions per minute
TSP	Total Suspended Particulate
US EPA	United States Environmental Protection Agency

Bieringer et al. (2017).

Given the significance of mining as a source of particulate emissions (Chaulya, 2004; Commonwealth of Australia, 2013; Gautam et al., 2016a,b; Huertas et al., 2012; Sastry et al., 2015), the availability of high quality, well documented particulate emission estimation methods that are applicable to key mining regions is a fundamental input to the environmental decision making framework. In particular, the need to adopt activity specific particulate emission techniques to ensure the appropriateness of emissions estimates adopted in national pollutant inventories has been highlighted (Weng et al., 2012).

The published literature has also identified the need for appropriate testing procedures to determine the properties of mineral dust from mining operations (Noble et al., 2017). Patra et al. (2016) reviewed available data relating to the emissions and human health impact of particulate matter from surface mining operations. The need for further research to both determine emission rates of particulate matter gener ated due to mining activities, and to characterize the physical and chemical properties of particulate matter to allow for more compre hensive assessment of potential impacts in the surrounding atmosphere and on the health of mine workers in the mines, was identified (Patra et al., 2016). Ghose (2007a) identified that, in light of the detrimental impacts of coal mining emissions and the move to open cut mines, re search is required to determine the appropriate emission rates to con sider in planning studies for new mines.

In an Australian context, it has been recognised that activity specific particulate emission rates should be adopted to ensure the appro priateness of emissions estimates incorporated into National Pollutant Inventories such as the Australian NPI (Weng et al., 2012). Weng et al. also identified limitations in ability to interpret Australian NPI data due to the lack of clarity in the underlying causes of the total mass emis sions.

To further address this identified deficit in the availability of emissions data, the research programme described in this paper in volved empirical testing of a range of open, erodible surfaces at open cut black coal mines in three regions of Australia: one of the biggest coal mining producers in the world (Gupta, 2014). The resultant dataset provides new wind speed specific emission rates for a range of coal mining activities, and considers the importance of regional variability, moisture content in determining emission rates.

2. Empirical test methods

2.1. Design of sampling apparatus

Determination of particulate emission rates for fugitive sources at open cut coal mines is a complex issue. A range of methods have been utilised in previous published studies including:

- downwind isokinetic sampling using high volume samplers (NERDCC, 1988);
- upwind downwind method using high volume samplers (Axetell and Cowherd, 1981; Frankell, 1993; US EPA, 1998b);
- exposure profiling technique involving isokinetic measurements immediately downwind of the source at multiple points in the ver tical plane (Frankell, 1993; US EPA, 1998b);
- low volume sampling techniques (Pietersma et al., 1996);
- quasi stack method in which the source is partially or fully enclosed and stack sampling approaches utilised (Frankell, 1993; US EPA, 1998b); and
- wind tunnel testing (Carras et al., 1999; Frankell, 1993; James et al., 2001; McKenna Neuman et al., 2009; Raupach and Leys, 1990; Strong et al., 2016).

Of the sampling methods available, the approaches that enclose or condition the test environment offer the greatest opportunity for minimising external influences such as meteorology (eg. the quasi stack and wind tunnel methodologies). Portable wind tunnels have been in use since the early 1950s where they originally were used for testing of erosion rates for agricultural surfaces (Gillette, 1978; Zingg, 1951). Design considerations and validation have been addressed in numerous published papers (Bocharov, 1984; Carras et al., 1999; James et al., 2001; Maurer et al., 2006; Pietersma et al., 1996; Raupach and Leys, 1990). For this research, the wind tunnel method was selected as the most appropriate for testing emission rates from open surface sources at coal mines, as the emission source has similar characteristics to agri cultural erosion.

2.2. Source characteristics

Particulate emission rates from open area emission sources are primarily governed by wind erosion of the surface with wind speed the parameter of primary importance (Strong et al., 2016), degree of sta bility of the surface (eg, crusting), surface moisture content and surface silt content and silt loading (Sharratt and Vadella, 2014). Surface moisture content and silt loadings are readily tested by standard la boratory methods. The degree of stability of the surface is a determi nant of wind erodibility, and wind erosion of particulates can be tested using a portable wind tunnel of suitable design.

Raupach and Leys (1990) defined the key processes associated with surface wind erosion as follows:

- suspension of particles (particle size of $< 20\,\mu m$);
- saltation processes (particle size of 20 1000 μm);
- creep (particle size of $> 1000\,\mu m$).

Of these processes, saltation is of primary significance from a par ticulate emission generation perspective. Saltation involves the impact of medium sized particles forcibly on the surface and, in turn, causing dislodgment of other particles from the surface. Creep involves the rolling motion of the largest particles across the surface. Therefore, to adequately simulate particle erosion from open surfaces the wind tunnel design must provide a logarithmic mean wind profile that is uniform over the eroding surface. The generation of an equilibrium boundary layer of a depth sufficient to contain the particle processes is suitable for this purpose (Maurer et al., 2006). In addition, saltation processes must be introduced to ensure that this important particle generation process occurs across the test face of the wind tunnel.

2.3. Wind tunnel design

The wind tunnel design adopted for the study was based on previous designs including ones provided by (Pietersma et al., 1996; Raupach and Leys, 1990). Fig. 1 shows a schematic diagram of the adopted wind tunnel design, with specific components of the design highlighted. The

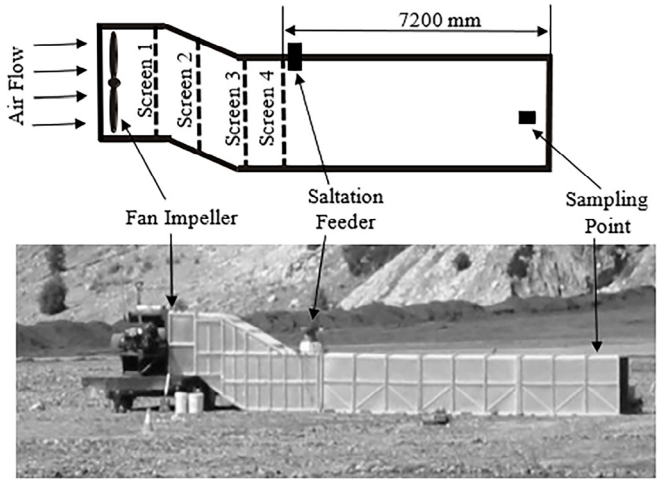

Fig. 1. *Wind Tunnel Schematic Design.*

key elements of the wind tunnel are:

1. Variable speed axial fan of a size sufficient to generate the appro priate range of wind speeds within the wind tunnel.
2. Flow conditioning devices immediately downwind of the fan to in troduce laminar flow (Screens 1 3). The screens introduce pro gressively smaller apertures, to transition the air flow from turbu lent to laminar. Screens 1 and 2 comprised plastic mesh, and Screen 3 was constructed from a bed of plastic drinking straws glued to gether in a tight formation.
3. Flow conditioning device to introduce a logarithmic boundary layer downwind of the laminar flow section, just prior to commencement of the open test section of the wind tunnel (Screen 4). This com prised a series of thin, circular movable bars in a vertical and hor izontal formation, to allow for adjustments to be made to the ver tical profile for calibration purposes.
4. A 7.2 m working section with the base open to the eroding surface. Tunnel cross section of 1 m wide, 1.2 m high.
5. Saltation particle feeder capable of introducing particulates at a variable feed rate to the 7.2 m long test section.

The wind tunnel trailer was designed to be capable of being towed by a standard four wheel drive vehicle to minimise costs associated with transport and operation. To minimise weight, the tunnel was constructed from aluminium and a winch and rotating platform to allow lowering of the tunnel transition section to the test area was provided.

2.4. Design validation

The performance of the wind tunnel was validated at a field test site prior to commencing the mine emission sampling, to confirm that the appropriate aerodynamic features were satisfactorily achieved. Both vertical and horizontal velocity profiles were determined using a TSI Velocicalc Hotwire Anemometer (TSI, St. Paul, MN USA). Aerodynamic criteria describing the depth and stability of the turbulent boundary layer were determined for the wind tunnel and compared with em pirical equations. The wind tunnel was validated up to a maximum simulated wind speed of 12 m s^{-1} at a height of 1 m. Tunnel validation data confirmed that the velocity profile throughout the sampling zone of the tunnel was a suitable representation of a vertical atmospheric wind speed profile. The fan RPM rates for each wind speed were de termined during the validation phase, to allow these wind speeds to be generated for the test phase.

2.5. Test methods

Three sampling approaches were selected for the test programme. Firstly, to allow for identification of the point at which loose surface material is evacuated from the tunnel test surface and steady state erosion processes have commenced, a real time screening approach was selected. This involved use of an OSIRIS real time nephelometer (Turnkey Instruments, Northwich, UK). The second measurement ap proach was designed to sample PM_{10} particulates during the steady state erosion phase operations of the wind tunnel. Minivol™ (Airmetrics, Springfield, OR USA) low volume gravimetric samplers with US EPA approved size separation inlets were utilised for this purpose. The third approach involved isokinetic gravimetric sampling using sample nozzles inserted into the tunnel test (Pietersma et al., 1996). Isokinetic sampling involves drawing sample air into the sample apparatus at the same velocity as the sample air stream that is passing the sample point. This approach prevents entrainment of additional particles in the sampled air flow.

For the isokinetic and Minivol™ methods, sampling was undertaken at two heights in the tunnel over the test period. The results of the two test positions represent an average concentration across the natural wind profile zone of the tunnel. All measurements were made above the saltation zone (0.3 m or above) to ensure that saltation particles were not introduced into the samples.

The isokinetic approach involved collection of total suspended particulates (TSP) on 47 mm teflon (PTFE) filters. The sample nozzle was inserted through an aperture in the side of the wind tunnel. Samples were collected for four different wind speeds. A 47 mm filter housing was utilised in conjunction with an inline gas meter and ro tameter to establish isokinetic sample rates. Sample air volumes were typically > 1 m^3 over the duration of each test.

The Minivol™ low volume sampling technique involved operation of the sample inlet and filter housing at a position within the end of the working section of the tunnel. A connection pipe to the pump was utilised to allow the pump unit to be remotely located (thus minimising wind field disturbance). The sample flow rate for the unit, 5 li tres min^{-1}, is required to satisfy the design parameters for the PM_{10} size selective inlet. Millipore fluoropore filters were utilised for the Minivol™ sampling.

2.6. Study site selection

Sample collection was completed at three open cut coal mines in Australia. Mines were selected in three different regions to represent differing climatic and geological conditions. The mines were in Central Queensland, South East Queensland and the Hunter Valley, New South Wales. At each mine, individual test locations were selected to re present the typical range of fugitive emission sources in an Australian open cut coal mine. They included a heavy vehicle haul road, a light vehicle haul road, a dragline walkroad, tailings dam/spoil piles, and coal and/or rejects stockpiles.

Within each mine, test sites were selected on the basis of providing a representative range of conditions at the mine, a suitably sized level area for installation of the wind tunnel, and a safe working location for the test personnel. Prior to commencing the fieldwork at each test site, the absence of significant rainfall during the previous 14 days was confirmed. This approach was adopted to provide for the sampling programme to determine emission rates for dry surface conditions. Testing was also carried out to assess the effect of watering as a control mechanism for the haul roads.

2.7. Sampling protocol

For each of the test sites, a range of surface wind speeds (5, 6.7, 8.9 and 10.8 m s^{-1}) were tested. They were selected to represent the wind speeds that occur under normal atmospheric conditions in Australia.

The lowest wind speed of 5 m s^{-1} was adopted as this is representative of the 5.4 m s^{-1} threshold velocity adopted in the Australian emission estimation manual for open stockpile erosion (Environment Australia, 2012). At lower wind speeds, the emission equation assumes that the wind erosion related particulate emissions are negligible.

At each location, sampling was undertaken for a 90 minute period for each wind speed. On completion of each test, the sampling appa ratus was removed from the tunnel and the sampled filters collected for analysis. Samples were collected for TSP and PM_{10}.

To minimise the influence of background concentrations, test loca tions were selected that were not influenced significantly by localised sources of particulate emissions in the mine and measurements were only completed under light conditions. This is consistent with the US EPA (US EPA, 1988) and Australian (Commonwealth of Australia, 2012; NERDCC, 1988) research and methodologies, and was also ne cessitated by the light weight nature of the wind tunnel sections, which were found to blow over under moderate wind speeds. On this basis, the currently adopted open area emission rates, derived from wind tunnel testing, are not corrected for background concentrations.

2.8. Sample analysis

A gravimetric analysis approach was adopted for the filter analysis. This involved use of an analytical balance sensitive to 1 μg. The balance was positioned in an air conditioned room, and within a physical en closure to buffer temperature and humidity variation. A simple hu midity stabilisation mechanism (potassium permanganate crystals) was utilised within the enclosure to assist in humidity control. Continuous temperature and humidity readings were recorded throughout all gravimetric analysis procedures. Using this approach humidity was maintained to 50% ± 1% within the enclosure, well within the range of variability allowed by reference methods for gravimetric analysis (Australian Standards, 2006).

For the haul route test sites, sampling was also completed to assess haul route erosion with and without surface watering. To measure the application rate of water to the haul route surface, a metal tray with a known surface area was located on the test surface where the water cart was to pass. After the water cart passed over the surface, the tray was collected and the retained water, that would normally enter the surface, was measured in a measuring cylinder. Based on this information, the rate of water application per square metre was determined.

2.9. Quality assurance and quality control

During the sample phase, the critical component of the sampling related to the air flow rates for the sampling devices. This was important for two reasons. Firstly, accuracy of flow rates is necessary to allow calculation of resultant sample concentrations. Secondly, achieving the correct flow rates is essential in ensuring isokinetic flows were achieved for the TSP samples, and the appropriate flow rate for the cyclone in the case of the PM_{10} sampling. For the isokinetic sam pling train, flow rates were measured for each sample using a calibrated dry gas meter. The MiniVol samplers were calibrated using a portable rotameter.

For the analytical phase, quality assurance was maintained throughout the gravimetric analysis process in accordance with the requirements of Australian Standard 3580.9.9 Methods for sampling and analysis of ambient air Determination of suspended particulate matter PM_{10} low volume sampler Gravimetric method (Australian Standards, 2006). This involved maintaining the appropriate laboratory environment conditions throughout the filter conditioning and weighing phases, both prior to and following field sampling.

3. Limitations - sampling methodology

One of the potential problems identified in previous studies (NERDCC, 1988; US EPA, 1998b) is a need to ensure that sufficient mass increase was obtained on gravimetric filter samples to ensure that valid test data could be calculated. A significantly lengthened sample time for each test phase was adopted in this study to overcome this limitation. Mass increases for the TSP filters were significantly higher than for the PM_{10} filters, greater than 100 μg in most of cases. The mass increases obtained for the PM_{10} filters were lower and there was sig nificant variability in the individual filter weighing results, due to in stability in the mass increases. This introduces greater statistical variability into the PM_{10} results. On this basis, the sampling results are presented for TSP only and results are not presented in this paper for PM_{10}.

The wind tunnel was operated with saltation particulates to provide for maintenance of surface erosion rates at an equilibrium throughout the test cycle. Therefore, the extended sampling duration test time did not affect the validity of the measured particulate emission rates.

4. Results

Table 1 presents the corrected uncontrolled emission rates for the tested wind speeds for each of the surface types without surface wa tering. These data represent the continuous emission rate for wind speeds in excess of 5 m s^{-1} 100% of the time, and where rainfall has not occurred in the preceding 14 days. The highest measured average emission rate of 7.11 kg ha^{-1} hr^{-1} was for the dragline walkroad at QLD South. For this source, the emission rate at 10.8 m s^{-1} is more than

Table 1
TSP Emission Rates (Without surface Watering).

Mine	Source	Emission Rate (kg ha ^{1} hr ^{1}) for each wind speed				Average Emission Rate (kg ha ^{1} hr ^{1})
		5 m s ^{1}	6.7 m s ^{1}	8.9 m s ^{1}	10.8 m s ^{1}	
Central QLD	Haul Road	2.78	5.51	8.15	12.00	7.11
Central QLD	Haul Road (after truck passed)	2.39	1.46	4.78	4.27	3.23
QLD South	Haul Road	1.97	–[a]	3.28	3.93	3.06
Hunter Valley	Haul Road	–[a]	3.31	8.43	8.97	6.91
QLD South	Dragline Walk Road	5.41	6.91	–[a]	9.24	7.19
QLD South	Spoil (Uncrusted)	2.90	6.78	6.46	8.69	6.21
Hunter Valley	Spoil (Uncrusted)	1.57	3.81	3.84	6.07	3.82
Central QLD	Spoil (Crusted)	1.48	1.77	5.11	12.14	5.12
Central QLD	Spoil (Uncrusted)	6.06	–[a]	5.57	–[a]	5.82
QLD South	Rejects	3.57	1.38	2.09	6.34	3.34
Central QLD	Rejects	4.33	3.40	4.07	7.07	4.72
Central QLD	Tailings	–[a]	–[a]	–[a]	0.02	0.02
QLD South	Dragline stockpile	–[a]	–[a]	–[a]	6.09	6.09
Average		**3.25**	**3.81**	**5.18**	**7.07**	**4.83**

[a] Valid sample data is not available.

4 times the rate at 5 m s^{-1}. The highest emission rate at a specific wind speed is for crusted spoil at 10.8 m s^{-1}. At this wind speed the emission rate of 12.14 kg ha^{-1} hr^{-1} is 8 times the emission rate at 5 m s^{-1}.

The lowest emission rate at a specific wind speed is also for crusted spoil, with an emission rate of 1.48 kg ha^{-1} hr^{-1}. This indicates that at the lower wind speeds surface crusting is reducing the emissions to below typical surface emission rates, and at the higher wind speeds the surface crust must be compromised, thus allowing significant emissions to occur. The emission rates at the intermediate wind speeds are consistent with this, and indicate that the surface crusting is beneficial at a wind speed of 6.7 m s 1, and no longer provides a benefit at 8.9 m s^{-1} as emission rates at this wind speed were similar to emission rates for other surfaces at this wind speed.

The results of the surface emission rate testing with surface watering controls are presented in Table 2. The emission rates at each wind speed, and the overall average, with surface watering are lower than the emission rates without watering as presented in Table 1.

Summaries of the average emission rates by region and by activity are presented in Tables 3 and 4. The overall average emission rates by region are similar, and range from 4.7 kg ha^{-1} hr^{-1} to 4.82 kg ha^{-1} hr^{-1}. There is greater variability in the emission rates for the different activities, with the lowest emission rates measured for the tailings dam (0.02 kg ha^{-1} hr^{-1}), which is low due to the surface being in a moist state, and the rejects stockpile (4.03 kg ha^{-1} hr^{-1}). The highest activity emission rate is for the dragline stockpile, at 6.09 kg ha^{-1} hr^{-1}.

5. Discussion

5.1. Emission rate variability

Overall, the measured emission rates demonstrate that particulate emission rates increase as the wind speed increases (Tables 1 and 3). This trend is less apparent for the watered surfaces (Table 2), indicating that the erosion threshold wind velocity increases where the surface moisture content is higher. This is consistent with the action of the cohesive forces of the adsorbed water film surrounding the soil particles reducing the potential for wind erosion and entrainment of the particulate matter (Wiggs et al., 2004). The variation in emission rates between different types of surface is in the range 2.9 4.67 kg ha^{-1} hr^{-1} (Table 4). Comparison of the emission rates without surface watering (Table 1) and with surface watering (Table 2) demonstrates that surface watering resulted in an overall average control efficiency of 25%.

The average emission rates without watering (Table 3) are consistent for the three mine regions, with similar emission rates determined as an average across all wind speeds tested (a range of 4.70 4.82 kg ha hr^{-1}). Review of the emission rates by mine region at specific wind speeds confirms that there was limited variability between the emission rates at 5 m s ^{1} and 6.7 m s^{-1} with average emission rates of 3.25 and 3.81 kg ha hr^{-1} respectively. At 8.9 m s^{-1} there is a 36% increase in average particulate emission rates relative to emissions at 6.7 m s^{-1}, and an 86% increase to 7.07 kg ha hr^{-1} at 10.8 m s^{-1}, relative to the emission rate at 6.7 m s^{-1}. These results indicate that at wind speeds at or below 6.7 m s^{-1} the action of the wind and saltation particulate erosion processes on the mine surface is insufficient to result in significant surface erosion. At wind speeds of 8.7 m s^{-1} and above there is a significant increase in emissions, indicating that the threshold friction speed has been reached at which both particle suspension and saltation processes occur (Gillies and Lancaster, 2013). These threshold wind speeds are slightly lower than reported for wind erosion of open surfaces in the Athabasca oil Sands Region in Alberta, Canada, where the lowest threshold wind speed for saltation processes was determined as 9.44 m s^{-1}, and more typically occurred at 10.3 m s^{-1} (Wang et al., 2015).

Based on the measured emission rates with and without watering of the test surface, watering control efficiencies for surface particulate emissions are presented in Table 5. Surface watering was completed as a single pass by a mine watercart except in the case of Central QLD, where the watercart passed over the surface twice prior to the controlled emission rate test being completed (which was the standard watering regime adopted at that mine). For the single water cart pass, the application rate was in the range 0.3 0.7 litres/m^2, and for the double pass the water application rate was 0.8 0.9 litres/m^2. These water application rates are lower than a Level 1 watering regime, as defined in the Australian emission estimation manual for mining (Environment Australia, 2012), of 2 litres/m^2/hour or a control efficiency of 50%.

5.2. Evaluation of measured emission rates

The existing published emission rates (Environment Australia, 2012; US EPA, 1998a) define an average 'default' emission rate of 0.4 kg ha^{-1} hr^{-1} for TSP from open erodible surfaces at black coal mines. The measured data represent emission rates at specific wind speeds, during dry conditions. Therefore, based on historic wind speed and rainfall data for the test regions, site specific average wind erosion emission rates for the tested surfaces were determined. Table 5 presents a comparison of the measured emission rate (dry, > 5 m s^{-1} wind speed) with average emission factors adjusted for annual rainfall and wind speed profiles, and the US EPA and Australian NPI default surface erosion emission rate.

The results presented in Table 5 show that the measured, weather corrected average emission rates for SE Queensland and the Hunter Valley are within 15% of the NPI default emission rate. However, the measured emission rate for the Central QLD mine is substantially lower than the default 0.4 kg ha^{-1} hr^{-1} emission rate used for estimating emissions from Australian coal mines and is similar to the default emission rate recommended in the US EPA AP42. For Central QLD, there is a low occurrence of wind speeds in excess of 5 m s^{-1} and a higher annual rainfall than the other test sites, based on observations obtained from the nearest Bureau of Meteorology meteorological monitoring station to the mine (Bureau of Meteorology, 2018). This results in lower calculated emission rates (Table 5) due to the local meteorological conditions. This comparison demonstrates the importance of local wind speed and rainfall to overall annual emission rates from surface erosion. The availability of emission rates at specific wind speeds, as provided in this study, allow for calculation of region

Table 2
TSP Emission Rates with Surface Watering.

Mine	Source	Emission Rate (kg ha ^{1} hr ^{1}) for Specified Wind Speed				Average Emission Rate (kg ha ^{1} hr ^{1})[#]
		5 m s ^{1}	6.7 m s ^{1}	8.9 m s ^{1}	10.8 m s ^{1}	
Central QLD	Haul Road (after truck passed)	2.27	2.15	2.61	4.13	2.79
Hunter Valley	Haul Road	4.6	2.79	6.00	5.75	4.79
Hunter Valley	Stockpile (washed coal)[a]	1.96	1.55	3.74	5.38	3.16
QLD South	Haul Road	1.72	0.67	2.46	2.84	1.92
Average		**2.64**	**1.79**	**4.36**	**5.52**	**3.6**

[a] Coal was wet due to washing, not watering.

Table 3
Average TSP Emission Rates For Each Mine (Without Surface Watering).

Mine	Emission Rate (kg ha $^{-1}$ hr $^{-1}$) for Specified Wind Speed				Average Emission Rate (kg ha $^{-1}$ hr $^{-1}$)
	5 m s $^{-1}$	6.7 m s $^{-1}$	8.9 m s $^{-1}$	10.8 m s $^{-1}$	
l QLD	3.41	3.04	5.54	7.1	4.77
Hunter Valley	1.57	3.56	6.14	7.52	4.70
QLD South	3.46	5.02	3.94	6.86	4.82
Average:	**3.25**	**3.81**	**5.18**	**7.07**	**4.83**

Table 4
TSP Emission Rates by Activity (Without Surface Watering).

Source (No. of Samples)	Emission Rate (kg ha $^{-1}$ hr $^{-1}$) for Specified Wind Speed				Average Emission Rate (kg ha $^{-1}$ hr $^{-1}$)
	5 m s $^{-1}$	6.7 m s $^{-1}$	8.9 m s $^{-1}$	10.8 m s $^{-1}$	
Haul Road (4)	2.38	3.43	6.16	7.29	4.81
Spoil (4)	3.00	4.12	5.25	8.97	5.33
Rejects Stockpile (2)	3.95	2.39	3.08	6.71	4.03
Dragline Walk Road (1)	5.41	6.91	–*	9.24	5.39
Tailings (1)	–*	–*	–*	0.02	0.02
Dragline Stockpile (1)	–*	–*	–*	6.09	6.09

* Valid sample data is not available.

Table 5
Measured Emission Rates Compared to Weather Corrected and Default Emission Rates.

Location	TSP (kg ha $^{-1}$ hr $^{-1}$)			
	Average Emission Rate (Wind > 5 m s $^{-1}$, no rain)	Average Emission Rate Adjusted for Wind Speed and Rainfall	Default Emission Rate NPI[a]	Default Emission Rate AP42[b]
Central QLD	4.8	0.07	0.4	0.1
Hunter Valley NSW	4.7	0.35	0.4	0.1
SE Queensland	4.8	0.42	0.4	0.1
Average	**4.8**	**0.28**	**0.4**	**0.1**

[a] Default emission rates for TSP from open erodible surfaces (Environment Australia, 2012).
[b] Default emission rates for TSP from open erodible (US EPA, 1998a).

Table 6
Measured TSP Emission Rates in Indian Coal Mines.

Coal Mine	Exposed pit surface emission rate, kg ha $^{-1}$ hr $^{-1}$
Sasti	0.55
Lakkhanpur	0.55
Belpahar	2.26
Ananta	3.6
Jagannath	1.84
Block II	0.46
Kusunda	0.56
Rajpura	0.73

specific surface erosion emission rates and adjustment for the local wind speed conditions. This has important implications for dispersion modelling studies, as adoption of the default emission rates could sig nificantly under or over estimate actual emissions for a specific locality such as Central QLD.

The measured emission rates without surface watering are equiva lent to the worst case emission rate under dry, windy conditions, thus representing the upper bound of emission rates likely to occur in the short term. The unadjusted emission rates represent worst case, short term dust erosion events, hence are suitable for use in predictions of short term dust nuisance or short term acute health impacts in the vi cinity of open cut coal mines.

Whilst there is extensive research relating to particulate con centrations in the vicinity of coal mines (Aneja et al., 2012; Gautam et al., 2016a,b; Ghose, 2007a; Huertas et al., 2014; Pokorná et al., 2016), there is limited empirical data in the published literature re lating to surface emission rates from coal mining. Apart from the US EPA (1998a) and NERDCC (1988) studies that form the basis of the emission estimation equations developed for open cut mines in the US and Australia an investigation by Chakraborty et al. (2002) presents

Table 7
Watering Control Efficiency (TSP).

Mine Source	Average Emission Rate (kg ha $^{-1}$ hr $^{-1}$) - no watering	% Moisture – Prior to Watering	Average Emission Rate (kg ha $^{-1}$ hr $^{-1}$) - with watering	% Moisture – After Watering	Measured Control Efficiency %	% Silt Fraction	Moisture Ratio
QLD South Haul Road	3.06	2.0	1.92	9.3	37.3	1.3–2.1	4.7
Central QLD Haul Road (after truck passed)	3.23	1.3	2.79	–	13.6	1.6–2.5	–
Hunter Valley Haul Road	6.91	1.6	4.79	5.7	30.7	16–45	3.6
Average:	**5.7**	**1.6**	**3.2**	**7.5**	**27.2**	-	**4.2**

Table 8
Watering Control Efficiency (TSP).

Mine Source	% Moisture – Prior to Watering	% Moisture – After Watering	% Moisture 1 h After Watering
Dragline walk road	1.7%	12.4%	6.5%
Haul road (high ambient temperature)	1.2%	6.5%	1.8%
Average:	**1.45%**	**9.45%**	**4.15%**

measured emission rates from surface erosion in pits at 8 coal mines in India. These data are presented in Table 6 and have been compared to the empirically derived emissions rates for Australia.

The range of emission rates observed at Indian Mines (Table 6) shows a minimum emission rate of 0.36 kg ha^{-1} hr^{-1}. This emission rate is similar to the default emission rate of 0.4 kg ha^{-1} hr^{-1} adopted in the Australian NPI methodology. The highest emission rate is for Belpahar, at 2.26 kg ha^{-1} hr^{-1}. This is within the worst case surface emission rates (ie, without correction for regional rainfall and wind speed) measured in the Australian coal mines, where an average of 4.8 kg ha^{-1} hr^{-1} was determined. Chakraborty et al. (2002) did not identify whether surface watering controls were adopted at the areas tested in the Indian mines, hence it is possible that the variability in emission rates for the mines in Table 7 may be related to differences in surface moisture. The comparison demonstrates that the average emission rates for Australian mines, when adjusted for local weather conditions, are well within those reported for Indian mines. The worst case emission rates, where on site controls such as surface watering and meteorological conditions have not been accounted for, are more than double the maximum rate measured in the Indian mines.

5.3. Influence of surface watering

The data presented in Table 7 demonstrates an average reduction of 27.2% for watering as a control mechanism, with a range of 13.6% 37.3%. The water application rates result in an average control efficiency lower than the 50% reduction adopted in the Australian NPI for Level 1 watering. This lower efficiency is to be expected, as the water application rate was less than half of the 2 litres/m^2/hour identified in the Australian NPI manual as necessary for achieving a control efficiency of 50%.

Of greater significance is the measured average moisture ratio (% moisture after watering divided by % moisture prior to watering) of 4.2. Based on the US EPA emission factors for unpaved haul roads, this would indicate a control efficiency of > 90% (US EPA, 1998b). The empirical data for Australian conditions indicates that lower control efficiencies are achieved in practice than would be expected based on the US EPA AP42 emission control estimation methods for unpaved haul roads. This indicates that US EPA AP42 watering control efficiency empirical calculation method is not suitable for application to Australian conditions and should not be used for the calculation of control efficiencies for surface watering. Application of this approach could result in over estimation of the benefit of surface watering as a control mechanism, for Australian conditions.

In order to identify the effectiveness of the watering regime over time, for a dragline walkroad and a haul road (under higher ambient temperatures than the walkroad), surface samples were tested prior to watering, immediately following watering and after a period of approximately one hour. The results for these moisture tests are presented in Table 8. These data highlight the significant reductions that occur in surface moisture over a one hour period. In particular, for the higher temperature environment, the surface moisture content reduced to close to the initial level within one hour. This highlights the necessity of surface water application occurring regularly, and at frequencies of one hour or less for haul roads. This conclusion is consistent with the measured watering application rates of 0.3 0.9 litres/m^2 for up to a double water cart pass. Four or more water cart passes per hour would be necessary to achieve the 50% watering control efficiency rate of 2 litres/m^2/hour defined in the Australian NPI emission estimation manual for mining.

6. Conclusions

Using a wind tunnel approach to sampling, this research has identified wind speed specific surface emission rates for a range of open area sources at coal mines in Australia. The empirical data demonstrates that at wind speeds of 5 m s^{-1} 6.7 m s^{-1} there is limited variation in particulate emission rates. At higher wind speeds, emission rates increase significantly, and at a wind speed of 10.8 m s^{-1} particulate emission rates are 86% higher than at a wind speed of 6.7 m s^{-1}. This demonstrates the significance of higher wind gusts in generating particulate emissions, and the necessity of considering local meteorological data when developing particulate emission inventories.

The research has identified an average surface watering control efficiency of 25%. This indicates that lower control efficiencies may be achieved in practice compared to the estimation method based on moisture ratios developed by the US EPA. The research also demonstrates the necessity of completing surface watering at intervals of less than one hour in order to achieve emission reductions of 50% or more under Australian conditions.

The empirical data developed in this study has increased our knowledge of surface particulate emissions in coal mines. Firstly, wind speed specific emission rates have been determined. This allows for calculation of region specific emission rates based on local meteorological conditions. Secondly, emission rates have been determined for a range of open erodible surfaces in open cut black coal mines. Currently, a single generic default emission rate is adopted to represent these sources, hence the research provides for more detailed analysis and calculation of site specific particulate emission rates. The empirical data also provides a basis for prediction of worst case surface erosion emissions under specific wind speeds, and hitherto not available.

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Appendix C: Paper 3 – Open Cut Black Coal Mining: Empirical Verification of $PM_{2.5}$ Air Emission Estimation Techniques

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Open cut black coal mining: Empirical verification of $PM_{2.5}$ air emission estimation techniques

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ABSTRACT

There is a paucity of empirical data relating to $PM_{2.5}$ emissions from open cut coal mines. Availability of high quality emission data is fundamental to the completion of robust analyses of the potential future impacts of particulate emissions from open cut coal mining operations. This study presents $PM_{2.5}$ emission rates determined on the basis of empirical data from open cut coal mining operations located in two regions of Australia. These data are used to validate the currently adopted $PM_{2.5}$ emission estimation methodologies, which are based on total suspended particulate measurements adjusted for $PM_{2.5}$ size fraction. The emission rates determined in this study demonstrate that regional differences between emission rates may be significant for specific activities. Measured average $PM_{2.5}$ emission rates are compared to rates calculated using existing available emissions estimation methodologies. This reveals that for dragline operation and coal dumping there are significant differences between the measured emission rates and emission rates calculated using currently available methodologies. This study identifies that the existing adopted US EPA and Australian emissions estimation methods for $PM_{2.5}$ emissions from coal haul routes, which are one of the most significant sources of $PM_{2.5}$ emissions at open cut mines, are reliable if the surface moisture content and dust control watering regimes are appropriately accounted for. The Australian estimation method is also considered reliable for overburden haul routes, however the US EPA method over predicts the measured $PM_{2.5}$ emission rate by a factor of 3.

Further, the study highlights that there is a high degree of inherent uncertainty in empirically derived emissions estimates for fugitive dust sources of this type. This uncertainty should be considered when completing emissions calculations, particular where these estimations are subsequently the basis for impact assessment and health risk analysis.

1. Introduction

The community health effects associated with exposure to particulate pollution are well documented (Patra et al., 2016; Tiwari et al., 2012; Tsiouri et al., 2014). Furthermore, the World Health Organisation has identified that no safe limit exists for community exposure to particulate pollution (WHO, 2013). This has consequences for industries such as mining, which is one of the primary sources of industrial particulate emissions in countries with significant open cut mining activity (Commonwealth of Australia, 2013; Environment Australia, 2017; Gautam et al., 2016; Huertas et al., 2012b; Sastry et al., 2015).

Where new mines or extensions to existing mines are proposed, atmospheric dispersion modelling of particulates is a key tool that is used to inform environmental policy decisions (Ghose, 2007; Huertas et al., 2012b; Weng et al., 2012). The accuracy of dispersion modelling has been found to be critically reliant on the quality of the input data, with the emission data inputs being a key variable (Tartakovsky et al., 2013). Furthermore, inaccuracies that result in significant over estimates of emissions may lead to requirements for improved mitigation and management techniques for emissions to air. Introducing these controls could pose a commercial risk in terms of the economic viability of a coal mine. Mines that are estimated to have significant particulate emission could also be subject to increased scrutiny by regulators, particularly in relation to proposed expansions to existing facilities and new mines, and the perceptions of the significance of the industry's emissions to the air, particularly by environmental groups, communities and regulators, could be unnecessarily increased. In the case of emissions under estimation, higher environmental particulate concentrations and health risk consequences to the community may arise that were not apparent from dispersion modelling studies.

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The currently adopted particulate emission estimation methods are of varying quality and largely rely on empirical data and estimation techniques developed for the United States (Chakraborty et al., 2002; Commonwealth of Australia, 2012; Petavratzi et al., 2005; Simon et al., 2008). The keynote study that formed the basis for development of mining emissions factors in the United States in the early 1980s was unable to quantify $PM_{2.5}$ emissions directly, and relied upon application of particle size fractionation to estimate the fine particulate emissions (Axetell and Cowherd, 1981). A similar study completed in Australia in the late 1980s adopted a similar approach and estimated $PM_{2.5}$ fractions based on particulate size distributions (NERDCC, 1988). Overall, there is a paucity of empirical data relating to $PM_{2.5}$ emissions, and this a significant gap in our current understanding (Gautam et al., 2016; Richardson et al., 2018).The primary objective of this study is to determine emission rates for $PM_{2.5}$ for a range of activities in open cut black coal mines in Australia. Such empirical data would allow for improved validation of the currently adopted emission estimation techniques, which are currently derived from total suspended particulate measurements adjusted for $PM_{2.5}$ size fraction.

2. Materials and methods

2.1. Selection of sampling methodology

A range of methods are available for the estimation of emissions of particulates: mass balance calculations; engineering calculations; sampling or direct measurements; emission factors; and alternative (approved) techniques (Environment Australia, 2015; US EPA, 2013).

For this study, a direct measurement approach was adopted. Completing direct sampling at an open cut coal mine is challenging due to the variability (temporal and spatial) of the fugitive sources that are encountered in open cut coal mines. Further constraints that applied to this investigation were the absence of mains power in the vicinity of the activities to be sampled, a requirement to maintain a safe separation distance from the mine activity, and the changing nature of the mine activities which necessitated relocating the sampling instrumentation on a daily basis, and, on occasion, during the same sampling day.

In light of these constraints, a downwind sampling method was adopted as the most practicable solution. This method is well documented (Axetell and Cowherd, 1981; Bieringer et al., 2017; Frankell, 1993; Jia et al., 2013; Sastry et al., 2015) and relies on completion of a series of measurements along a downwind transect, with subsequent analysis using Gaussian techniques to calculate the emission rate (NERDCC, 1988; Smith, 1995), based on the concentration and associated dispersion parameters measured at the time.

As the method requires that the sampling transect is downwind of the plume emitted from the activity of interest, the wind speed and direction were measured prior to commencement of sampling at each location to allow selection of a downwind sampling transect position. The wind direction was subsequently monitored throughout the sampling to allow adjustment of the transect position where changes in wind direction resulted in the transect being > 45° from the transect alignment. A typical wind rose for a transect sampling period is presented in Fig. 1. An estimation of initial plume height and horizontal spread was also completed for each sampling location, based on direct observation of the visible particulate plume and review of video footage of the sampling. The plume height and width was estimated by the sampling team based on these visual observations. The largest plume heights and widths were measured for the dragline activities, with an average width of 133 m and height of 58 m observed immediately downwind of the source. This is to be expected given the size and height of a dragline. For point sources, the typical initial plume height was 6 m, and the initial plume width an average of 7.5 m. Narrower widths were typically observed where site features constrained the plume size, for example proximity to earth berms. For line sources only the plume height is relevant, as the source width is continuous. The plume height was generally defined by the turbulent wake to the rear of the vehicle. As the mine haul vehicles were generally a similar size, the plume heights for the various haul activities are very similar (average height of 5 m).

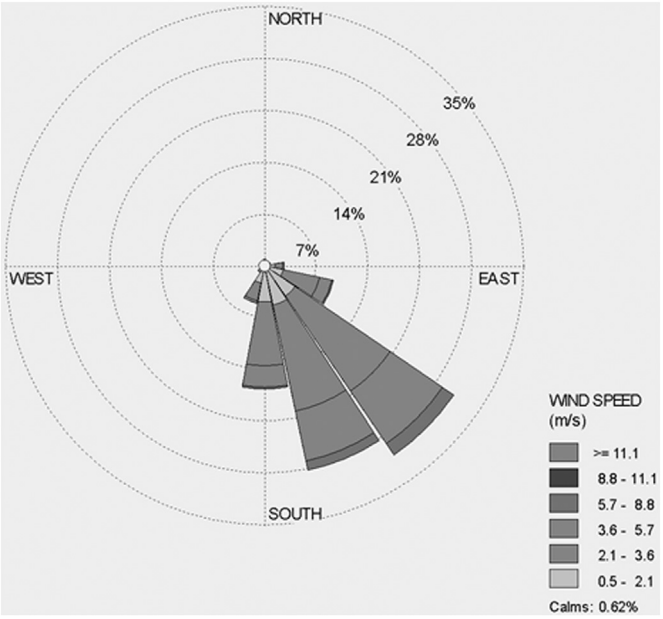

Fig. 1. Wind Rose for Downwind Sample Transect – Dragline Operations (Central Queensland). (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

Additional data collected during the air sampling procedures undertaken for different mining activities included plant and equipment types; material throughputs; duration of each activity cycle; number of vehicle movements; and particulate control methods adopted (eg. haul route watering). These data were required as inputs to the subsequent calculation of emission rates specific to vehicle movement rates or unit of material throughput.

2.2. Instrumentation

A portable beta attenuation sampling method was adopted for the downwind transect measurements. Beta attenuation is an approved reference sampling method in numerous jurisdictions including USA (US EPA, 2016), Europe and Australia. Portable E Bam samplers (Met One Instruments Inc., 2008), were utilised in the study. The E Bam instrument has a data resolution of $1\,\mu g\,m^{-3}$, a detection threshold of $< 6\,\mu g\,m^{-3}$ and an overall accuracy of +/− 10%. This was considered acceptable for the purposes of the study, as these data quality parameters conform with the requirements of AS/NZS 3580.9.12 (Standards Australia, 2013).

The E Bam sampling was completed under ambient temperature and humidity. For beta attenuation sampling, it has been demonstrated that there is potential for relative humidity in excess of 60 80% to result in over estimation of particulate concentrations relative to gravimetric sampling methods, particularly for temperatures below 16 °C (Chang et al., 2001; Takahashi and Sakamoto, 2008; Triantafyllou et al., 2016). Relative humidity and temperature data were recorded for each sampling period to allow the potential influence of these parameters on the measured particulate concentrations to be determined.

Particulate sampling was also completed at an upwind location to allow correction of the emission data for ambient background. A portable real time optical particulate sampler was adopted for this purpose. As this sampling method differs significantly for the beta attenuation method, a second optical sampler was co located with the E Bam at the first transect position for a number of samples at each mine, to allow a

Table 1
Measurement instrumentation utilised in the study.

Measurement Parameter	Instrumentation	Details
$PM_{2.5}$ at 4 positions on a downwind transect from the emission source	Met-One E-Bam, fitted with US EPA approved reference $PM_{2.5}$ cyclone separator	Data recorded for 1 min averages, validated at the end of each hour to provide hourly data. Data resolution: 1 μg m ^{3}, detection threshold: < 6 μg m ^{3}, accuracy: +/ 10%.
$PM_{2.5}$ background concentrations at an upwind position	TSI DustTrak model 8530	Data resolution: 1 μg m $^{3.}$ Particle size range: 0.1 to 10 μm, fitted with $PM_{2.5}$ impactor for sampling. Flow accuracy: +/ 1%.
Wind speed and direction.	Davis weather station for initial stage of sampling.	Davis weather station: 1 min data resolution.
	Met-One 034B Windset coupled with a Campbell Scientific CR800 data logger	Met-One 034B: 1 s data resolution.
Soil moisture and silt content.	Calibrated laboratory oven and soil sieve set	Gravimetric analysis method adopted for determination of soil moisture. Manual sieve method adopted for determining silt fraction.
Flow Calibration	Bios Defender 510 DryCal	Used for flow calibration for the E-BAM instruments.

correction coefficient to be determined. The correction coefficient was applied to adjust the optical sampler data to an E Bam equivalent concentration. To confirm the validity of the method adopted for de termining the calibration coefficient, additional testing was completed using collocated optical and E Bam samplers at the background location and also at transect position 1 for the same test. The data from this additional testing resulted in the same average correction coefficient for the background position and Transect 1 position, thus confirming the suitability of this approach.

Details of the instrumentation adopted in the study are presented in Table 1.

Prior to commencement of sampling transect at each location, a series of quality assurance checks were completed to ensure the accu racy of the E Bam data in accordance with the manufacturer's re commendations. These involved an air leak check, flow rate calibration, particulate concentration zero and span check and an instrument self check in accordance with the documented manufacturer's procedure. The Dustrak optical particulate samplers were calibrated by an external approved laboratory prior to the sampling, and field checks of zero and flow were completed for each sampling phase to confirm the on going calibration status of the instruments.

2.3. Sampling locations within the mines

Research has shown that mining particulate emission rates may vary on a regional basis or even within mines where significant geological differences occur (Ghose, 2007; Huertas et al., 2012b). The sampling programme was completed in two Australian mining regions: Queens land (QLD) Bowen Basin and the New South Wales (NSW) Hunter Valley. Sampling was completed at one mine in Queensland and three mines in New South Wales. Each mine operated a number of pits and working faces, hence allowing for sampling of the same activity at a number of different locations within the mine subject to local conditions and work schedules. Table 2 summarises the mining activ ities and number of test sites per activity for the sampling programme. For each activity, the number of sampling hours completed for the full transect (i.e. 4 positions × 1 h sample per position) is provided. In addition, the total number of hours sampled at each individual transect position is presented.

When identifying the suitability of specific sampling locations for the study, the following parameters were considered:

- absence of significant rainfall in the previous month. Sampling was only completed where materials or surfaces were in a dry condition, except where watering was a standard control technique (e.g. for haul routes);
- light to moderate winds. Sampling could not be completed during zero or very light wind conditions, as there was no downstream plume of emissions. Sampling was not completed under average wind speeds of > 5 m/s which would result in rapid dispersion of emissions leading to difficulties in accurately characterising the width and height of the downwind plume. Typical average wind speeds for the measurements were in the range 2 3 m/s, with < 1% of 1 min weather observations exceeding 8 m/s.
- logistical and safety issues. The availability of target activities (such as dragline operation) during the proposed sampling period was a specific issue, and close liaison with the mine operator was neces sary to ensure fieldwork coincided with target activities.

The initial sampling position on the transect was typically located 20 60 m from the source of emissions. For some operations, such as the dragline, larger separations were necessary for safety reasons and the initial transect Position 1 was located 100 m or more from the source. The intermediate and final positions on the transect were largely de fined by localised constraints such as proximity to earth berms and pit edges. The average distance for Position 4 on the transect was 103 m

Table 2
Summary of Sampling Programme.

Mine Activity	Source Type	QLD – Bowen Basin			NSW – Hunter Valley		
		Sites Tested	1-h Data Points[a,b]	Transect Hrs	Sites Tested	1-h Data Points[a]	Transect Hrs
Coal Haul Road	Line	2	27	7	3	13	3
Overburden Haul Road	Line	2	16	4	4	24	6
Drilling	Point	3	24	6	3	23	6
Dragline	Point	3	29	7	1	20	5
Overburden Loading	Point	2	21	5	8	72	18
Overburden Dumping	Point	2	16	4	2	8	2
Coal Loading	Point	3	37	9	2	9	2
Coal Dumping	Point	2	21	5	1	14	3.5

[a] This is the total number of 1 h measurement data points that were considered in determining the average overall emission factors for each activity.
[b] The total number of hours is < 4 per transect in some instances. This is due to instruments failing post sampling quality assurance checks or due to site constraints limiting the number of samplers that could be positioned along the transect to < 4 positions.

Fig. 2. Dragline Measurement Transect.

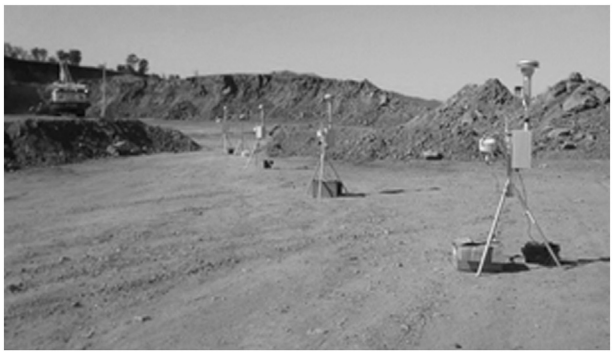

Fig. 3. Overburden Loading Measurement Transect.

from the initial (Position 1) sampling location, with the intermediate positions equally spaced between these points.

Figs. 2 4 present examples of the sampling instrumentation in po sition for three different mining activities.

2.4. Determination of emission rates

2.4.1. Overview

The empirical data obtained during the field trials was used as the basis for determining emission factors for the activities considered. A Gaussian approach was adopted based on the methodology for devel opment of a number of the current emission factors utilised in the Australian National Pollutant Inventory Emission Estimation methods (Commonwealth of Australia, 2012; NERDCC, 1988). Emission factors were determined separately for line and point sources using simplified Gaussian dispersion equations.

2.4.2. Calculation of line source emissions

Due to the nature of line sources, the spread of the plume is cal culated in the vertical direction only as the whole plume will move

Fig. 4. Drilling Measurement Transect.

Table 3
Dimensionless Constants Used to Calculate Plume Spread Parameters.
Source: Zimmerman, J R and Thompson, R S National Environmental Research Centre, Users Guide for HIWAY. A Highway Air Pollution Model, 1975.

Stability Class[a]	Description[a]	a	b
A	Very unstable	0.180	0.945
B	Unstable	0.145	0.932
C	Slightly unstable	0.110	0.915
D	Neutral	0.085	0.870

Note: Night time stability classes E and F are not considered, as all sampling was completed during the daytime.

[a] Stability class and descriptions as per Pasquill, F The estimation of the dispersion of windborne material. Meteorological Magazine. 90: 33–49. February 1961.

across the downwind transect in the horizontal direction. The vertical plume spread parameter incorporates a virtual distance term which accounts for the initial size of plume in the vertical axis. Firstly, the vertical height of the plume was estimated during field observations. This height is divided by 2.15 (Turner, 1994) to provide the σ_{z0} para meter. The virtual distance (x_0) is then defined as:

$$x_0 = (\sigma_{z0})^{1/b} \tag{1}$$

where, b is a dimensionless empirical parameter, and σ_{z0} is an initial vertical plume spread parameter measured in meters. The vertical plume spread parameter (σ_z) is a function of the stability class and the downwind distance of the sampler, defined as follows:

$$\sigma_z = a(x + x_0)^b \tag{2}$$

where, a and b are dimensionless empirical parameters, and x is a downwind distance. The two empirical parameters based on the stabi lity class are presented in Table 3.

The emission rate q in $gs^{-1}\,m^{-1}$ is defined as follows:

$$q = \frac{1}{2}\chi sin(\theta)(\sqrt{2\pi})\sigma_z u \tag{3}$$

where, χ is a plume centreline concentration at a distance x downwind from the source (g/m3), θ is an angle between wind direction and line source (degrees), and u is mean wind speed (m/s).

2.4.3. Calculation of point source emissions

As described above, several assumptions can be made about the location of the plume centreline for line sources which result in the simplification of the line source equation. This is not the case for point sources, as the transect could be located off the plume centreline in both the vertical and horizontal directions due to varying winds and emis sion heights. Therefore, two adjustment factors for the horizontal and vertical axes are introduced. These factors are functions of both the effective distance between the sampler and the plume centreline, and the plume spread parameter for the relevant axis.

The vertical plume spread parameter (σ_z) is the same as defined for the sources while the horizontal plume spread parameter is defined as:

$$\sigma_y = \frac{\sigma_\theta x}{57.3} + \sigma_{yo} \tag{4}$$

where, σθ is the standard deviation of mean wind direction, and σy0 is observed plume width at the source divided by 4.30 m (Turner, 1994).

To calculate the distance from the plume centreline to the sampler in the horizontal direction (y), the resultant wind direction relative to the samplers is determined and the distance is then calculated by tri gonometry. The distance from the plume centreline to the sampler in the vertical direction (z) is defined as the difference between the height of the emission point and the height of the samplers. The reduction factors in the horizontal and vertical direction (R_y and R_z respectively) are defined as:

Table 4
Parameters for Determining Atmospheric Stability Class – Sigma Theta Method.

σ_θ	Stability Class	P_θ
$\sigma_\theta > 24.8^\circ$	A	0.06
$22.3^\circ < \sigma_\theta < 24.8^\circ$	B	0.15
$16.4^\circ < \sigma_\theta < 22.3^\circ$	C	0.17
$10.9^\circ < \sigma_\theta < 16.4^\circ$	D	0.23
$\Sigma_\theta < 10.9^\circ$	E	0.38

$$R_y = exp\left(-0.5\frac{y^2}{\sigma_y{}^2}\right) \text{and } R_z = exp\left(-0.5\frac{z^2}{\sigma_z{}^2}\right) \quad (5)$$

Then, the emission rate (Q) for a point source in gs^{-1} is defined as follows:

$$Q = \frac{\chi\pi\sigma_y\sigma_z u}{R_y R_z} \quad (6)$$

The Pasquill Gifford stability class during the measurement was determined by the Sigma Theta method described by Slade (1968). This involves observing each 20 min interval of the meteorological data during the measurement period and determining the most extreme wind direction in degrees during each interval.

The Sigma Theta value is then determined as:

$$\sigma_\theta = \frac{\Delta\theta_{max}}{6} \quad (7)$$

where $\Delta\theta_{max}$ is the extreme range of wind direction over a 20 min interval.

The stability class is then determined from the value of σ_θ from Table 4. The stability class boundary values have been adjusted for a measurement height of 2 m by the following method (US EPA, 2000; Yamartino, 1984):

$$(Z/10)^{(P_\theta)} \quad (8)$$

where Z is measurement height in metres and Pθ is determined from Table 4.

2.4.4. Calculation of source specific emission rates

The emission rates were determined in terms of g/s for point sources and $gs^{-1}\,m^{-1}$ for line sources. These emission rates were further analysed to determine emission rates specific to the tested activity, using the activity rates for each test as recorded at the time of sampling. Emission rates from haul roads were expressed in kilograms per vehicle kilometre travelled (VKT). For point sources, throughput specific emission rates were determined by dividing the emission rate in grams by the unit throughput, such as the number of holes drilled or the quantity of material loaded or dumped.

3. Results

The emission factors determined from the sampling data for each activity are presented for NSW (Hunter Valley) and QLD (Bowen Basin) coal mines in Table 5. The calculated standard deviation is presented for each emission factor. These results have been corrected for measured background concentrations, hence represent activity specific emission rates.

The standard deviations for the dataset are relatively high at 1.0 to 2.3 times the overall average sample result. This indicates a significant degree of variability associated with the results. The US EPA has noted that standard deviations of up to 5 are typical for measured emission rates (US EPA, 1995). Hence this is a recognised feature of empirical emission sampling for fugitive sources. To improve data quality, in accordance with the draft US EPA emission data calculation methodology (US EPA, 2013), the raw data has been further analysed to remove samples below the method detection threshold (MDL) and to remove data outliers. The resultant emission rates are presented in Table 6. The standard deviations of the average data set are improved, as a result of this data processing, now ranging from 0.7 1.7 times the calculated emission rate.

The results of the temperature and humidity sampling completed for each test confirm that the particulate samples were collected at temperatures > 16 °C and the maximum average relative humidity for individual sampling periods was 55.1%. On this basis, the potential for over estimation of particulate concentrations due to particulate bound water is considered to be negligible (Takahashi and Sakamoto, 2008). Water was applied to the road surface during sampling as a standard operating practice for management of dust, hence the potential for increased particulate bound moisture to be present for these sources, despite the absence of a significant influence from relative humidity, has been considered. Takahashi and Sakamoto (2008) indicated that the differences in particulate concentrations measured by beta attenuation sampling relative to gravimetric sampling methods is not significant at estimated particle moisture concentrations of 12% or less. The maximum measured moisture content for all samples collected in this study was 6%, therefore particulate bound moisture is well within the threshold estimated by Takahashi and Sakamoto (2008), as resulting in over estimation of particulate concentrations. Therefore, no adjustment of the measured concentrations to account for particle bound moisture was necessary.

4. Discussion

4.1. Spatial variability

The variability between the two Australian coal mining regions considered in this study is consistent with sampling studies completed overseas (Chakraborty et al., 2002; Huertas et al., 2012a) that identify that mining activity emission rates can vary on a regional basis. Chakraborty et al. (2002) present particulate emission data for 10 mining regions of India, and these data indicate an average variability by a factor of 2.2 between the regions in India for the emission sources considered for the QLD and NSW mines. The maximum variability exhibited was a factor of 3 for mineral loading operations for the ten regions of India included in the Chakraborty et al. study. Table 7 presents a comparison of the emission rates for the Queensland Bowen Basin and the New South Wales Hunter Valley. The comparison demonstrates that the emission rates for the Bowen Basin and the Hunter Valley vary by more than a factor of 2 except for overburden haul and overburden dumping. Overall, the NSW emission rates are higher, except in the case of coal loading and dumping where they are lower than the average QLD emission rates. Even in the case of coal loading and dumping, the emission rate variation is well within the range of up to two orders of magnitude observed in the US EPA empirical particulate emissions analysis study for sample datasets with 10 or more data points (Axetell and Cowherd, 1981).

To determine whether region specific features were the cause of these differences, Table 8 presents a comparison of silt and moisture content data for the mining activities tested.

Review of the % silt content data indicates that the data for QLD is typically at the lower range of the measured silt contents for the majority of parameters. This indicates potential for particulate emission rates to be lower than the NSW mines. Moisture content are also higher for QLD for the coal haul roads, overburden haul roads and overburden dumping, however the moisture contents for the dragline and coal dumping are lower than those measured in NSW. These features may be the cause of the higher calculated emission rates for coal haul, overburden haul and overburden loading for the NSW data. In the case of drilling, the silt fraction is significantly higher than all other samples for both regions, with a range of 4.05 8.86%. The higher silt content is associated with the nature of the drilling process, which is likely to result in the breakdown of the natural geological material that comes in

Table 5
Average Emission Rates (Background Corrected).

Mine Activity	Overall Average		QLD – Bowen Basin		NSW – Hunter Valley		Units
	Emission Rate	Standard Deviation	Emission Rate	Standard Deviation	Emission Rate	Standard Deviation	
Coal Haul Road	0.0803	0.1260	0.0321	0.0283	0.1912	0.1869	kg/VKT
Overburden Haul Road	0.0357	0.0371	0.0199	0.0220	0.0470	0.0417	kg/VKT
Drilling	0.4727	1.0747	0.0801	0.1265	0.8383	1.4025	kg/hole
Dragline	0.0135	0.0237	0.0149	0.0295	0.0113	0.0069	kg/BCM[a]
Overburden Loading	0.0012	0.0015	0.0004	0.0004	0.0014	0.0016	kg/t
Overburden Dumping	0.0005	0.0007	0.0004	0.0007	0.0007	0.0007	kg/t
Coal Loading	0.0039	0.0087	0.0046	0.0093	0.0003	0.0003	kg/t
Coal Dumping	0.0046	0.0064	0.0083	0.0070	0.0004	0.0006	kg/t

[a] Bank Cubic Metre – one metre cubed corrected for the density of the material.

Table 6
Sample Average Calculated Emission Rates (background, MDL and outlier corrected).

Mine Activity	Overall Average		QLD – Bowen Basin		NSW – Hunter Valley		Units
	Emission Rate	Standard Deviation	Emission Rate	Standard Deviation	Emission Rate	Standard Deviation	
Coal Haul Road	0.0715	0.1215	0.0272	0.0286	0.1912	0.1868	kg/VKT
Overburden Haul Road	0.0336	0.0275	0.0224	0.0221	0.0411	0.0286	kg/VKT
Drilling	0.1691	0.2885	0.0437	0.0414	0.3000	0.3704	kg/hole
Dragline	0.0074	0.0073	0.0048	0.0065	0.0113	0.0069	kg/BCM[a]
Overburden Loading	0.0009	0.0008	0.0004	0.0002	0.0011	0.0008	kg/t
Overburden Dumping	0.0003	0.0005	0.0003	0.0004	0.0005	0.0006	kg/t
Coal Loading	0.0006	0.0004	0.0007	0.0003	0.0002	0.0002	kg/t
Coal Dumping	0.0041	0.0063	0.0067	0.0071	0.0003	0.0003	kg/t

[a] Bank Cubic Metre – one metre cubed corrected for the density of the material.

to contact with the drill bit to the finest particle sizes possible. This increases the potential for emissions of particulate matter from drilling, hence drilling sources are an important component of a mine particu late emission inventory despite the relatively localised nature of emis sions from this activity.

4.2. Comparison with existing emission estimates for $PM_{2.5}$

Directly comparable empirically derived emission datasets for $PM_{2.5}$ are not currently available. While a range of empirical datasets are available for Total Suspended Particulates and PM_{10} (Chakraborty et al., 2002; Chaulya, 2006; Lal and Tripathy, 2012), calculation of $PM_{2.5}$ emission rates relies primarily on application of $PM_{2.5}$ fractio nation estimates from particulate size analysis data to empirically de rived total suspended particulate emission rates. Therefore, to de termine whether the measured $PM_{2.5}$ emission rates differ significantly from those calculated using the currently adopted emission estimation methods, a comparison is made to emission rates estimated using the currently adopted US EPA AP42 (US EPA, 1998a, 1998b) and Aus tralian National Pollutant inventory (Environment Australia, 2012) methods in Tables 9 and 10 respectively The calculated emission rates

Table 8
Comparison of Silt and Moisture Content.

Mine Activity	Silt Content %		Moisture Content %	
	QLD – Bowen Basin	NSW – Hunter Valley	QLD – Bowen Basin	NSW – Hunter Valley
Coal Haul Road	0.87 (1)	0.08–3.22 (2)	4.75 (1)	1.24–2.18 (2)
Overburden Haul Road	0.9 (1)	3.2 (1)	4.8 (1)	2.2 (1)
Drilling	6.90 (1)	4.05–8.86 (2)	3.44 (1)	2.90–6.00 (2)
Dragline	0.82 (1)	1.18 (1)	2.45 (1)	3.19 (1)
Overburden Loading	0.59 (1)	0.13–3.92 (4)	5.05 (1)	0.83–2.62 (4)
Overburden Dumping	2.93 (1)	N/A	5.01 (1)	N/A
Coal Loading	0.05–0.47 (3)	N/A	1.32–1.58% (3)	N/A
Coal Dumping	0.03 (1)	0.04 (1)	1.31 (1)	2.51 (1)

Note: values in brackets are number of measurements for that parameter.

Table 7
Comparison of Specific Regional Emission Rates.

Mine Activity	QLD - Bowen Basin	NSW – Hunter Valley	Ratio – QLD:NSW	Units	QLD – Transect hours[b]	NSW – Transect hours[b]
Coal Haul Road	0.0272	0.1912	0.14	kg/VKT	7	3
Overburden Haul Road	0.0224	0.0411	0.55	kg/VKT	4	6
Drilling	0.0437	0.3000	0.15	kg/VKT	6	6
Dragline	0.0048	0.0113	0.42	kg/hole	7	5
Overburden Loading	0.0004	0.0011	0.36	kg/BCM[a]	5	18
Overburden Dumping	0.0003	0.0005	0.60	kg/t	4	2
Coal Loading	0.0007	0.0002	3.50	kg/t	9	2
Coal Dumping	0.0067	0.0003	22.33	kg/t	5	3.5

[a] Bank Cubic Metre – one metre cubed corrected for the density of the material.
[b] Number of hours that measurements were completed using a transect of 4 instruments downwind for that mine activity.

Table 9
Comparison of Measured, Default and Calculated Haul Route $PM_{2.5}$ Emission Rates.

Mine Activity	NPI Emission Estimate $PM_{2.5}$[a]	Average Study - Measured $PM_{2.5}$[b]	Difference (measured minus estimate)	Ratio (measured to estimate)	Units
Coal Haul Road:					
Calculated (accounting for surface moisture)	0.0511	0.0715	0.0204	1.39	kg/VKT
Calculated, 75% control	0.0128	0.0715	0.0587	5.59	kg/VKT
Default (uncontrolled)	0.2538	0.0715	0.1823	0.28	kg/VKT
Default, 75% control	0.0635	0.0715	0.0080	1.13	kg/VKT
Overburden Haul Road:					
Calculated (accounting for surface moisture)	0.0303	0.0336	0.0033	1.11	kg/VKT
Calculated, 75% control	0.0076	0.0336	0.0260	4.43	kg/VKT
Default (uncontrolled)	0.2538	0.0336	0.2202	0.13	kg/VKT
Default, 75% control	0.0635	0.0336	0.0299	0.53	kg/VKT

[a] NPI Default TSP emission rate multiplied by NERDDC 0 μm – 2.5 μm size fraction.
[b] With haul route watering controls occurring during the measurements. The haul route watering is approximately equivalent to 75% control as defined in the NPI Mining Manual (Commonwealth of Australia, 2012), based on observations during the sampling.

Table 10
Comparison of Average Measured and Calculated (Australian Method) $PM_{2.5}$ Emission Rates.

10Mine Activity	Australian Method $PM_{2.5}$[a]	Average Study $PM_{2.5}$	Difference	Ratio	Units
Coal Haul Road	0.0511	0.0715	0.0204	1.398	kg/VKT
Overburden Haul Road	0.0303	0.0336	0.0033	1.107	kg/VKT
Drilling	0.0531	0.1691	0.1160	3.185	kg/hole
Dragline	0.0042	0.0074	0.0032	1.762	kg/BCM
Overburden Loading	0.0013	0.0009	0.0004	0.720	kg/t
Overburden Dumping	0.0005	0.0003	0.0002	0.625	kg/t
Coal Loading	0.0015	0.0006	0.0009	0.414	kg/t
Coal Dumping	0.0004	0.0041	0.0037	10.250	kg/t

[a] NPI Default TSP emission rate multiplied by Australian SPCC 1986 0 μm – 2.5 μm size fraction.

have accounted for the particulate control techniques applied during the time of the sampling, to provide a direct comparison. Average silt and moisture content values for the relevant activity as measured in the research study are adopted for the purposes of the calculations, thus the findings account for key local variables.

The US AP42 provides particle size fractionation corrections for $PM_{2.5}$, and these have been adopted for the purposes of predicting the emission rates presented in Tables 9 and 10. $PM_{2.5}$ fractionation factors are not published in the current Australian NPI Mining manual for fugitive emission sources at open cut coal mines. Therefore, an estimated $PM_{2.5}$ particle size fraction has been applied for the Australian NPI method based on research completed in the Hunter Valley (State Pollution Control Commission NSW, 1986).

The comparison of emission rates confirms that the Australian NPI calculation methods estimate emissions are within a factor of 2 of the average measured emission rates for coal haul roads, overburden haul roads, draglines, overburden loading and overburden dumping identified in this study. The US EPA emissions estimates are also similar for coal haul roads, overburden loading and dumping, and greater than a factor of 2 for the remaining activities.

The activities with the most significant differences for both calculation methodologies are coal dumping, drilling and coal loading and for the US EPA emission factors only, draglines and overburden haul. In the case of coal dumping, the average measured dataset is skewed by high emission rates for the Queensland Bowen Basin; the data for the New South Wales Hunter Valley (0.0003 kg/t) is equivalent to the average of the US EPA and Australian calculated emission rates (0.0002 kg/t and 0.0004 kg/t respectively). The coal dumping operations sampled in Queensland included stockpile (dozer) and nearby work area maintenance (grader) activities, and this is reflected in the higher emission rates. Whilst stockpile and work area maintenance activities are a regular feature of the coal stockpile activities, the NSW emission rates represent more typical emission rates for coal dumping only.

Similarly, the drilling emission rates measured for the Queensland Bowen Basin (0.0437 kg/hole) are within 18% of the calculated emission rates using the US EPA and Australian NPI methods (0.0531 kg/hole). This indicates the higher emission rate determined for the NSW Hunter Valley may relate to specific local conditions. The measured coal loading emission rate is significantly lower than the estimated emission rate, indicating that the current emission estimation methods may over estimate $PM_{2.5}$ from this activity.

For draglines, the US EPA emission rate is > 8 times lower than the measured emission rate. This indicates that the US EPA AP42 emission estimation method for draglines may significantly underestimate emissions when applied in other regions of the world and, possibly, in US coal mines.

Finally, in the case of overburden haul, the US EPA emission rate (0.0935 kg/VKT) is > 3 times higher than the measured emission rate (0.0336 kg/vkt) and the emission rate calculated using the Australian NPI calculation method (0.0303 kg/vkt). This indicates a potential for the US EPA calculated emission rate to significantly overestimate $PM_{2.5}$ emissions from over burden haul.

4.3. Haul route $PM_{2.5}$ emissions sensitivity analysis

In terms of the significance of the differences between measured and estimated $PM_{2.5}$ emission rates, haul road emissions are generally the most significant source of mass emissions of particulates in open cut coal mines at > 70% of total particulate emissions (US EPA, 1988). The dominance of mine haul roads as the primary source of particulate releases is also confirmed by emissions estimates for a range of mining proposals in Australia, as presented in the Environmental Impact Statements for these projects for example the Foxleigh Plains (Katestone Environmental, 2012) and Bylong Coal (Pacific Environment Ltd., 2015) air quality assessments. Given the significance of haul activities as a source of particulate emissions, further analysis has been completed to assess the significance of selection of different variables and assumptions in the calculation methods. Table 11 presents a comparison of the measured emission rates for coal and overburden haul routes, with the default emission rates and calculated emission rates as defined in the Australian NPI Emission handbook for Mining (Environment Australia, 2012).

The comparison presented in Table 11 demonstrates that, for coal

Table 11
Comparison of Measured, Default and Calculated Haul Route $PM_{2.5}$ Emission Rates.

Mine Activity	NPI Emission Estimate $PM_{2.5}$[a]	Average Study - Measured $PM_{2.5}$[b]	Difference (measured minus estimate)	Ratio (measured to estimate)	Units
Coal Haul Road:					
Calculated (accounting for surface moisture)	0.0511	0.0715	0.0204	1.39	kg/VKT
Calculated, 75% control	0.0128	0.0715	0.0587	5.59	kg/VKT
Default (uncontrolled)	0.2538	0.0715	0.1823	0.28	kg/VKT
Default, 75% control	0.0635	0.0715	0.0080	1.13	kg/VKT
Overburden Haul Road:					
Calculated (accounting for surface moisture)	0.0303	0.0336	0.0033	1.11	kg/VKT
Calculated, 75% control	0.0076	0.0336	0.0260	4.43	kg/VKT
Default (uncontrolled)	0.2538	0.0336	0.2202	0.13	kg/VKT
Default, 75% control	0.0635	0.0336	0.0299	0.53	kg/VKT

[a] NPI Default TSP emission rate multiplied by NERDDC 0 μm – 2.5 μm size fraction.
[b] With haul route watering controls at an average rate of 2.7 water cart passes per hour occurring during the measurements.

haul routes, using the Australian NPI method, the calculated emission rate (accounting for surface moisture) is 29% lower than the measured emission rate determined from the empirical data. Similarly, using the NPI default emission rate with 75% watering control efficiency, results in a predicted emission rate that is 11% lower than the measured emission rate. Whilst these emission rates would result in a degree of underestimation, given the variability associated with fugitive dust emissions estimates, this represents a reasonable correspondence between the measured and calculated datasets. Where the calculated emission rate is also adjusted for emission control in the form of watering, the resultant emission rate is 82% lower. This approach would result in significant underestimation of potential emissions, hence would not be suitable for use in emissions estimations and modelling studies.

For overburden haul routes, as with the coal haul routes, there is closest correspondence between calculated (with moisture control) and measured emission rates. The default emission rate both with and without controls over estimates emissions, and the calculated emission rate with an adjustment for 75% control significantly under estimates emissions by a ratio of > 4.

Review of the watering rates occurring when the coal and overburden haul route emission testing was completed confirms that, on average, a water cart traversed the test surface and applied water 2.7 times per hour tested. Therefore, it is concluded that water application at this rate is not sufficient to achieve the 75% reduction in $PM_{2.5}$ emissions from the haul route emission source assumed in the NPI emission estimation calculation method.

This analysis confirms that for coal haul and overburden haul routes, application of calculated emission rates that account for measured surface moisture contents results in emissions estimates that are closest to the measured emission rates. It is also concluded that, where an emission calculation accounts for material moisture content, then application of additional corrections to take account of watering for particulate control may result in significant underestimates of emissions in practice. This has significant implications for dispersion modelling of proposed mines, as assumptions relating to a 75% reduction in haul route emissions where watering is adopted as a control technique is likely to over estimate the reductions achievable in practice based on current haul route watering practices.

5. Limitations of the method

The key limitation of the empirical sampling and emission estimation method as described in this paper is the ability to accurately characterise the plume characteristics during the sampling programme. Improved data quality and a reduction in the standard deviation for the individual activity emission rates may have been possible if additional particulate samplers and wind sensors had been installed at additional horizontal and vertical positions in the downwind plume. Due to cost limitations, this was not practicable for this study.

6. Conclusions

The study has determined empirically derived emission rates for $PM_{2.5}$ for a range of open cut mining activities. The results of the study confirm that there may be significant variability in emission rates for different mines, and differences in silt fractions and material moisture contents may be the cause of this variability for a number of the empirically derived emission rates.

Average coal haul route emission rates for $PM_{2.5}$ as derived in this study are consistent with the currently adopted site specific calculation methodologies except where control in the form of watering is considered in the calculation. Where the calculation method is applied, the control efficiency of haul route watering may be significantly over estimated. The Australian NPI default emission estimation method for overburden haul is also consistent with the empirically derived emission rates and the US EPA calculation method results in over estimation by a factor of three. As with coal haul, where over burden haul emission rates are calculated with watering controls, the resultant emission rate is a significant under estimate of emissions in practice.

Based on the empirical data, the existing US EPA $PM_{2.5}$ emissions estimation techniques for draglines underestimates Australian emissions significantly. This indicates the need to adopt region specific emission rates for dragline emissions.

For coal dumping, significant variability was observed, and this was related to differences in ancillary activities occurring at the time of sampling. In the case of drilling, the average measured emission rates were higher than the calculated emissions, however the dataset was skewed by the significantly different emission rates observed between the three mine test sites.

For coal loading, measured $PM_{2.5}$ emission rates are significantly lower than the calculated emissions, and the current calculation methods may over estimate emissions from this source.

The research has demonstrated that where calculated emission rates account for site specific variables such as moisture content, applying further corrections for watering controls may result in significant underestimates of $PM_{2.5}$ emission rates.

The emissions dataset presented in this paper provides $PM_{2.5}$ emission rates for open cut coal mining activities, based on empirical data. This represents an advance over current emissions estimation techniques which rely on application of $PM_{2.5}$ correction factors to TSP or PM_{10} emission rates to allow estimation of $PM_{2.5}$ emissions.

The study has also highlighted the significant variability associated with empirical measurements of fugitive particulate emissions. This is

of particular relevance when risk assessments of existing or proposed future mines are being completed. This feature of the emission data may introduce a greater degree of uncertainty to predictive modelling, and this should be considered in any decision making process that relies on these data.

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