Comparison of the performance of two atmospheric dispersion models (AERMOD and ADMS) for open pit mining sources of air pollution

By

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Submitted as partial fulfilment of the requirements

For

Masters of Science in Applied Science: Environmental Technology

In the

Department of Chemical Engineering

Faculty of Engineering, Built Environment and Information Technology

University of Pretoria

2012

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Synopsis

The performance of the AERMOD and ADMS dispersion models was tested using PM_{10} (thoracic dust) emissions from Rössing Uranium Mine open pit in Namibia. The performance of the two models was evaluated against the observations and also against each other using various statistical measures. The models were tested under different case scenarios (cases explained in chapter 4) with the aim of evaluating their performances as well as their inter model variability.

The study was undertaken from the 13 July 2009 – 14 August 2009. The results from the study showed that the performance of ADMS was superior to that of AERMOD. In general, the performance of AERMOD was very poor and simulated extremely high concentration values. AERMOD performed even more poorly during calm conditions. ADMS performance was superior to AERMOD as was evident from the values of various performance statistical measures and a conclusion reached was that ADMS is likely to be a better model to use in cases where prolonged calm conditions are experienced.

Keywords: air dispersion modeling, fugitive emissions, open cast mining, particulate matters

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Acknowledgements

First and foremost, I would like to thank and praise the Almighty God for giving me the knowledge and energy to complete this study. Next, I thank Rössing Uranium Limited for funding my studies. My sincere gratitude goes to the Department of Chemical Engineering at the University of Pretoria for partly funding my research.

I wish to extend my heartfelt gratitude to my supervisor, Dr. Gerrit Kornelius for his scholarly guidance and support.

I am indebted to Neel Breitenbach and Reneé von Gruenewaldt of Airshed Planning Professionals (Pty) Ltd for their assistance in the preparation of the topographical and meteorological files. I also would like to thank Coleen De Villiers of the South African Weather Services for providing me with the meteorological data.

I am grateful to the Rössing Uranium Limited employees for the support they rendered to me during the time of my study. Some of them include Aina Kadhila Amoomo, Pedru Shamba, Rabanus Shoopala, Besser Rowhan and Jacklyn Mwenze. I also would like to thank Prof Jairos Kangira for his editorial work. Last but not least, I want to thank my family and friends for their patience and moral support from the beginning of my study to the end.

Thank you all. I would not have completed this study without your support.

Dedication

This thesis is dedicated to my late mother, Mrs Diina Neshuku. Despite her physical absence, she continues to be my inspiration in every respect.

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Abbreviations and Acronyms

^{222}Rn	Radon-222
^{226}Ra	Radium-226
^{85}Kr	Krypton
ACGIH	American Conference of Governmental Industrial Hygienists
ADMS	Atmospheric Dispersion Modelling System
AERMAP	AERMIC terrain pre-processor
AERMET	AERMIC Meteorological pre-processor
AERMIC	**A**MS/**E**PA **R**egulatory **M**odel **I**mprovement **C**ommittee
AERMOD	**AER**MIC **MOD**EL
AMS	American Meteorological Society
ANFO	Ammonium nitrate, fuel oil
BNFL	British Nuclear Fuels Ltd
BR	Basil Read
Ca	Calcium
CBL	Convective boundary layer
CERC	Cambridge Environmental Research Consultants
CFD	Computational fluid dynamic
CGM	Chinese Guideline Model
CGS	Cordierite Gneiss Schist
CIX	Continuos Ion Exchange
CSIR	Council for Scientific and Industrial Research
CTDMPLus	Complex Terrain Dispersion Model-Plus
DAS	Dust-a-Side
DEM files	Digital Elevation Mapping files
EF	Emission Factor
EMA	Environmental Management Act
EPA	Environmental Protection Agency
EPRI	Electric Power Research Institute
ER	Emission Reduction
EU	European Union
FB	Fractional bias

FDM	Fugitive Dust Model
FFP2	Filter Face Piece 2 (medium)
g/s	gram per second
GD	Gardner Denver
h	Boundary layer height
h_c	height scale
HEF	High Explosive Fuel
HPDM	Hybrid Plume Dispersion Model
IDC	Industrial Development Corporation
IOA	Index of Agreement
ISC3	Industrial Source Complex Model Version 3
ISCST3	Industrial Source Complex Short Term 3
LIDAR	Laser Imaging Detection and Ranging (system)
LG7	Low grade (dump) 7
LG5	Low grade (dump) 5
L_{MO}	Monin-Obukhov length
MMRS	Mine Management Reporting System
MP	Monitoring point
MSHA	Mine Safety and Health Administration
NMSE	Normalized Mean Square Error
NPI	National Pollutant Inventory
NPRI	National Pollutant Release Inventory
NRPB R91	National Radiological Protection Board model
NTP	Normal Temperature and Pressure
NSW	New South Wales
OEL	Occupational Exposure Limit
OSHA	Occupational Safety and Health Administration
PBL	Planetary Boundary Layer
pdf	probability density function
PM_{10}	Particulate matter with aerodynamic diameter less than 10 microns
$PM_{2.5}$	Particulate matter with aerodynamic diameter less than 2.5m
PTM	Particle Trajectory Model

PHII	Phase 2
PHIII	Phase 3
P1	Stockpile 1
P2	Stockpile 2
P3	Stockpile 3
P4	Stockpile 4
Q–Q plots	quantile quantile plots
RTDM	Rough Terrain Diffusion Model
RUL	Rossing Uranium Limited
SAWS	South African Weather Services
SBL	Stable Boundary Layer
SF_6	Sulphur Hexafluoride
SKM	Sinclair Knight Merz
SO_2	Sulphur Dioxide
SW	South West
SX	Solvent extraction
Ti	Titanium
Tl	Thallium
TLV	Threshold Limit Value
TSP	Total Suspended Particulates
U	Uranium
UK	United Kingdom
US	United State
U_3O_8	Uranium Oxide
UM	Unified model
USEPA	United State Environmental Protection Agency
UTM	Universal Transverse Mercator
VKT	Vehicle Kilometer Travelled
W6	Waste (dump) 6
W7	Waste (dump) 7
WHO	World Health Organisation
WRF	Weather Research Forecasting (model)

NOMENCLATURE

Symbol	**Description**
C_o	Observed Concentration in µg/m^3
C_p	Predicted Concentration in µg/m^3
ρ	Density in kg/m^3
µg/m^3	micro gram per cubic meters
m	meter
kg/m^3	kilogram per cubic meters

Chapter 1: Introduction

1.1 Background

Mining operations from opencast mines generate a considerable quantity of dust through various activities such as blasting, unpaved road haulage, loading and stockpiling (Silvester *et al.,* 2009; Chaulya, 2004). The generated dust is an environmental hazard that can negatively impact on human health as well as the surrounding environment. The dust generated from uranium mines contains radionuclides, primarily ^{222}Rn and its short-lived decay daughters which can seriously affect human health (Fernandes *et al.,* 1995). In addition, it may contain heavy metals such as manganese, vanadium and arsenic in relatively small quantities, further exacerbating the impacts of uranium mining on human health and environment (Fernandes *et al.,* 1995).

The determination of emission rates of various mining activities and prediction of pollutants concentration is necessary to assess the impacts of mining on air quality (Chakraborty *et al.*, 2002). The study of the transport and dispersion of dust in the atmosphere is crucial for managing and improving the current controls. It also determines the occurrence and frequencies of worst scenarios of weather and in the end, it enables people to avoid or minimise emissions during these adverse conditions (Cooper and Alley, 2002).

Atmospheric dispersion modelling is one of the tools that can be used to investigate dust emissions and dispersion. Atmospheric dispersion modelling is the mathematical simulation of the dispersion of pollutants primarily in the boundary layer of the atmosphere. It is undertaken by making use of computer programmes that solve mathematical equations and algorithms which simulate the dispersion of pollutants (El-Harbawi *et al.*, 2009).

Dispersion models have been developed for ground level concentration prediction, in most cases for regulatory purposes (Harsham and Bennett, 2008). Dispersion models, however, differ in their assumptions and structure as well as in the algorithms they use; as a result, predictions vary from model to model.

The performance of a model is assessed by comparing the predicted results to measured results during validation studies. Validation studies using one model are good only for model development. However, they do not assess the differences between models. It is therefore necessary to compare results from different models tested under the same conditions in order to assess the inter-model variability.

In this study, two atmospheric dispersion models, US AERMOD and UK ADMS models, were used to predict the dispersion and ground level concentration of PM_{10} at an opencast mine, with specific reference to Rössing Uranium Mine as a case study. AERMOD and ADMS are well validated dispersion models used worldwide and they are often applied to opencast mining in Southern Africa.

The performances of these two models have been tested in several studies involving stack emissions under various meteorological and topographical conditions (Sidle *et al.*, 2002; Dunkerley *et al.*, 2001; Harsham and Bennet, 2008). However, few studies have been conducted on fugitive dust emissions from low-level or in-pit sources from opencast mining. In the few studies that have been conducted, predictions were done using other models such as the Fugitive Dust Model (FDM) (Chaulya, 2002; Singh *et* al., 2006; Trivedi *et al., 2008*).

Furthermore, the results from AERMOD and ADMS have been tested against measured results of gaseous emissions both in flat and complex terrain (Riddle *et al.*, 2004; Hall *et al.*, 2000: Hanna *et al.*, 1999). However, few studies have been conducted on emissions of particulates. AERMOD and ADMS have fundamentally different algorithms; thus their treatment of dispersion in the terrain is different (Dunkerley *et al.*, 2001). It is therefore useful to compare the two models in order to evaluate their variability.

This study therefore serves as a validation study for the two models using PM_{10} emissions from an opencast mine, where fugitive dust sources, including those located in pits, predominate.

1.2 Objectives of the study

Under different meteorological conditions, terrain and source types, models behave differently. The primary aim of this study was to investigate how the two models predict PM_{10} dispersion and resulting ground level concentrations under the meteorological conditions that prevailed during the study. Other factors taken into consideration were the complex terrain around Rössing mine and the nature of the source, which is opencast mining in a deep pit.

The secondary objectives of the study were:

- To determine the emission rates for various PM_{10} sources at Rössing Mine
- To investigate the inter model variability between the two models.
- Evaluate the performance of the models, when the simulation was done with in-pit sources treated as if they were located on the surface rather than at the bottom

1.3 The outline of the dissertation

The following chapters are presented in this dissertation:

Chapter 1: Introduction

The research topic is introduced in this chapter.

Chapter 2: Literature survey

This chapter focuses on the review of literature pertaining to the subject of the study. This includes

- background information on Rössing Uranium Mine, the location of the mine, topography, climate and sources of dust around the mine.
- descriptions of dust, particle size and the impact of dust on health, environment and operational cost and maintenance
- information on regulations pertaining to dust and a review on the dispersion models used in the study
- a general description of dispersion modelling, basic mathematical algorithms and factors affecting the dispersion of pollutants in the atmosphere .

Chapter 3: Methodology

This section describes all methodologies and research instruments that were used for data collection. Further, the modelling process is explained.

Chapter 4: Results and discussion

The results of the research are presented in this chapter. The chapter also provides a discussion of the results of the research.

Chapter 5: Conclusions and recommendations

A summary of the research findings is presented in this chapter and areas to be investigated in further studies are outlined.

Chapter 6: References

A list of references is given in this chapter.

2.1 Background information on Rössing Uranium Mine

Rössing Uranium Mine is the third largest uranium oxide producer in the world and its production accounts for 8% of the total world production (RUL, 2009). It mines a high tonnage deposit of low grade uranium in a granite mineral called Alaskite (Moeller, 2001). The majority of shares in the mine (69%) are owned by Rio Tinto, followed by an Iranian Foreign Investment company which owns 15 percent (figure 2.1.) (RUL, 2008).

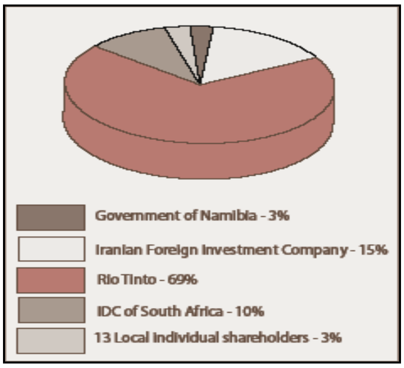


Figure 2.1: Rössing Uranium shareholders (Courtesy: RUL, 2008)

The mining operations are conducted through open pit mining also known as opencast or surface mining. The Rössing pit, known as the SJ pit, measures 3km by 1.2km and is presently 345m deep (figure 2.2) (Leggatt, 2009).

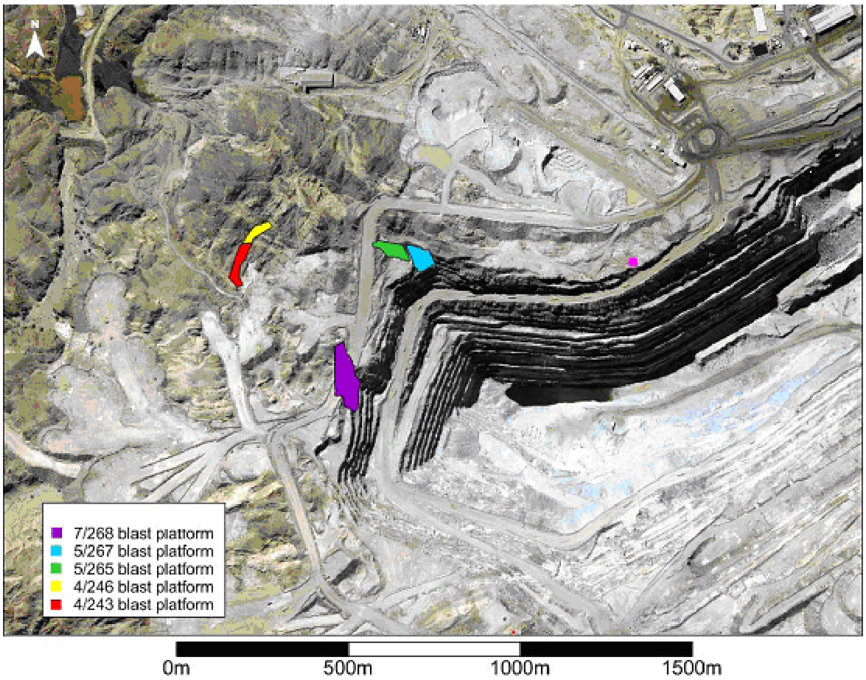


Figure 2.2: Part of Rössing open pit (Courtesy: RUL, 2008)

2.1.1 Location and topography

Rössing Uranium Mine is located approximately 70 kilometres north-east of Swakopmund (see figure 2.3) in Namibia. The geographical location of the mine is 15º 02’30'' East latitude and 22º 27’50'' south longitude in the Namib Desert.

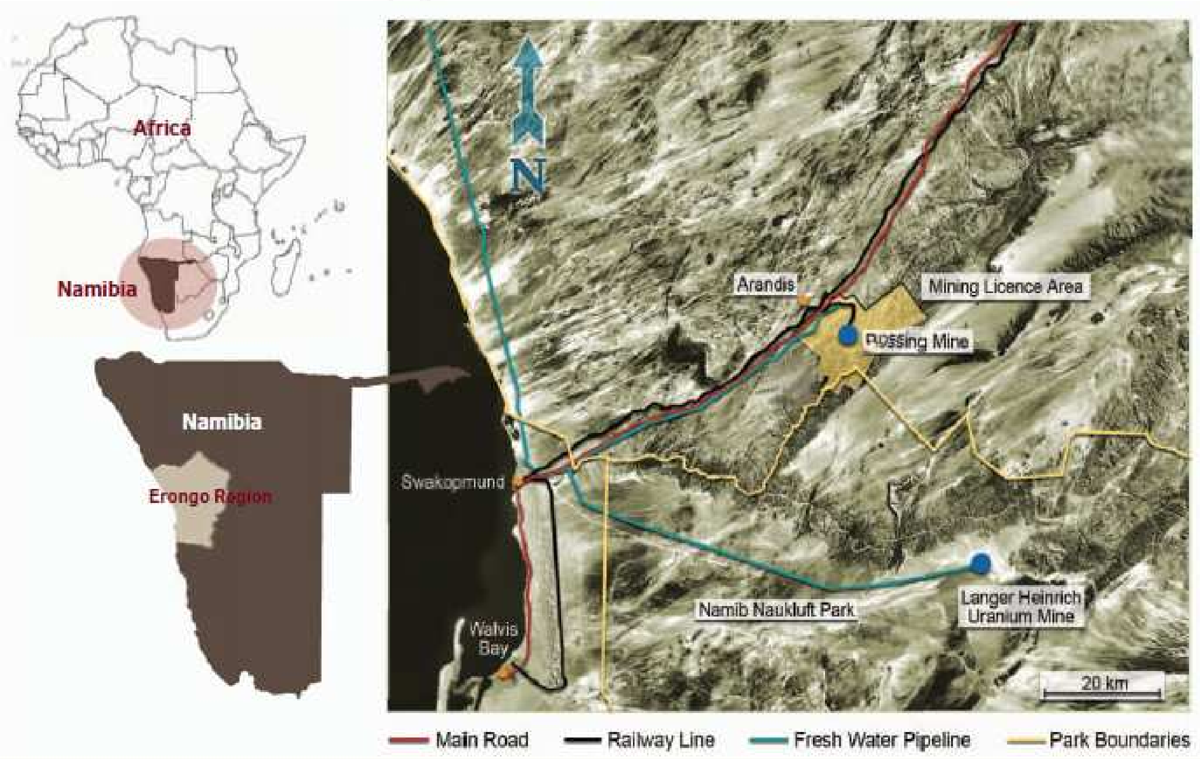


Figure 2.3: Location of Rössing Uranium Mine (Source: www.rul.com)

The mine area is characterised by low relief on the west, north and north east. The south-west part is characterised by shallow drainage lines and storm-wash gullies which drain towards the Khan River. The Khan River area is more hilly and craggy and the drainage lines combine and deepen to form a gully to the east which transverses the mine area and discharges into the Khan River.

2.1.2 Climate

Climatic condition variations play a major role in determining the diffusion, direction, distribution and transportation of atmospheric pollutants (Ninham Shand, 2008). It is therefore vitally important to understand the climatic features of an area under study. Rössing Mine is located in a desert, hence the amount of rainfall received is very low and its distribution is extremely inconsistent. On average, the mine receives 30-35 mm of rainfall annually. Mostly rainfall occurs during late summer and autumn as showers of sometimes a high intensity lasting for a short period. Practically no rainfall is received during summer months. In exceptional cases falls of up to 1mm per month are recorded.

Winds predominantly experienced at Rossing are north-easterly, westerly and south–westerly. A strong north-east to easterly wind called 'berg wind' occurs at Rössing Mine around 50 times per year mostly from the month of April to September. During the study period, berg winds were experienced. The berg winds occur when air displaced from the plateau to the coast becomes heated adiabatically due to a drop in altitude. These wind conditions are characterised by high temperature and wind speeds. Peak wind speeds can reach 125km/hr in extreme cases. Due to the high wind speed associated with these wind conditions, a large quantity of dust, sand and fine gravel are emitted and transported leading to dusty conditions.

Large variations in day to day air temperatures are experienced at Rossing, though seasonal variations are not well marked. Mean diurnal temperature ranges from 23.8 ºC in late autumn (May) to 15.4 ºC in spring (October). Minimum temperatures are recorded during the early morning hours and range from 2.0 ºC in August to 12 ºC in March. Maximum diurnal temperature ranges from 31.8 ºC in July to 39 ºC in January.

2.1.3. Mining operations

The ore at Rössing Mine is mined through four major stages: drilling, blasting loading and hauling. A short description of each stage, together with the potential for dust generation, is given below.

The first stage of the mining operation is drilling. After charging the drilled holes with a heavy ANFO explosive, the rocks are blasted to fragment them into required sizes. The ore is then loaded onto haul trucks by means of hydraulic and rope shovels.

(a) (b) (c)

Figure 2.4: Different stages of mining operations at Rössing Mine (a) drilling in progress (b) blasting (c) loading of ore into a truck by means of hydraulic shovel (Pictures courtesy of RUL)

After loading, the trucks pass through the radiometric scanners that measure the radioactivity level of each load. This scanning exercise determines whether the truck must proceed to the primary crushers; to a low grade stockpile; high grade stockpile, or to the waste dumps depending on the grade, calc index (Calcium carbonate content (increases acid consumption at the processing plant)) and CGS (Cordierite Gneiss Schist (reduces separation efficiency)) content of the material.

Figure 2.5.: Radiometric scanning of the ore loaded on the haul truck

Table 2.1: Stockpiles grouping according to the properties of the ore

Stockpile	Material properties
P1	High grade, low calc. index
P2	Low grade, low calc[A]. index
P3	low grade, high calc index
P4	Low grade, high CGS[B] content
LG	Low grade material

[A] Calcium carbonate content (increases acid consumption at the processing plant)

[B] Cordierite Gneiss Schist (reduces separation efficiency) (Aipanda, 2010)

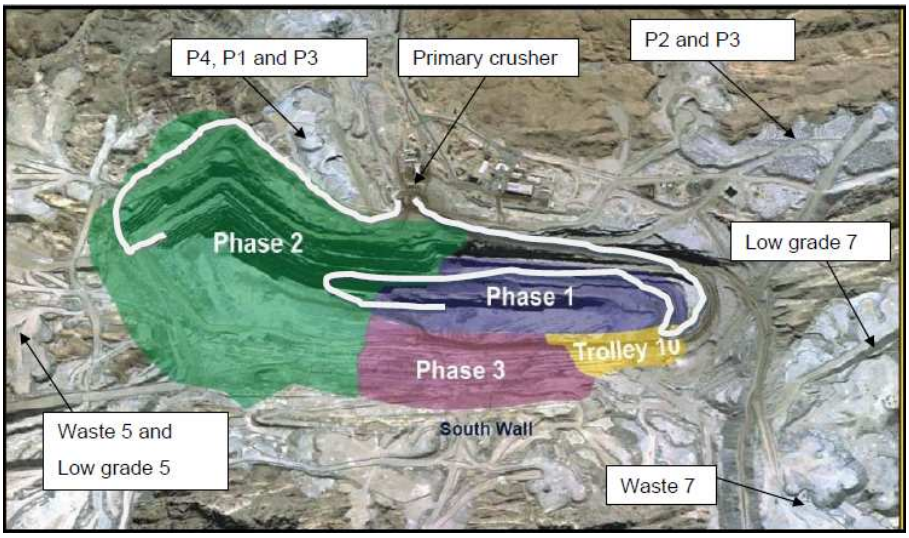


Figure 2.6: Rössing Mine open pit layout (W-Waste dump; LG-Low Grade stockpile; P- Ore stockpile) (Aipanda, 2010) (not to scale)

During the mining operations a vast amount of dust is generated through various activities. Dust is generated when blast holes are drilled and when blasting activities are carried out. During blasting, dust clouds are generated as a result of material fracture, energy release and the air volume displacement and translation due to the slumping of the fractured rock material to the ground (Silvester *et* al., 2009). Material handling is another source of dust at Rössing Mine. This happens when the ore is being loaded into the haul trucks by means of shovels. In addition, the dumping of both the ore and waste at stockpiles and waste dumps emit a significant quantity of dust into the atmosphere.

However, the primary source of dust from mining activities is the wind-borne dust generated during hauling and mine transportation on unpaved roads. When heavy vehicles travel on the unpaved roads at the mine site, the force of the wheels on road surface pulverizes the surface material and dust particles are lifted and dropped from the rolling wheels. The road surface is exposed to strong air currents in turbulent shear with the surface and dust is generated (Chakraborty, 2002).

2.1.4 Other sources of dust at Rössing Mine: Processing plant and tailings

The processing operation consists of the following main stages: crushing; grinding; leaching; slime separation; thickening; continuous ion exchange (CIX); solvent extraction (SX); precipitation; filtration; drying and roasting. A detailed description of the processing operations is beyond the scope of this thesis. Crushing and wind erosion of tailings have been identified as some sources of dust at the processing plant, (Moeller, 2001). A short description of the two sources, including the mechanisms of dust generation, is given below.

The processing plant contributes a relatively low quantity of dust in comparison to the pit and tailings emissions. In 1999 the processing plant emitted dust of 4 g/s as compared to 18 g/s and 46 g/s from the open pit and tailings respectively (Moeller, 2001). Sources and activities generating dust at the processing plant are given in Table 1.2 below.

Table 2.2: Sources of dust at the processing plant

Source	***Activity***
Primary crushing	Loading, crushing and reclaim
Coarse ore	Dumping and reclaim
Fine crushing	Loading, crushing and reclaim
Fine ore stockpile	Dumping and reclaim
Coarse ore stockpile	Wind erosion

Adopted from Moeller, 2001

Loading and crushing of the ore at both primary and fine crushing plants generate dust which ends up being emitted in the atmosphere. Another source of dust at the processing plant is wind erosion from coarse and fine ore stockpiles especially during windy conditions. Wind erosion of tailings generates a high quantity of dust at Rössing Mine. It topped the top ten dust generation sources list (with regard to TSP) in 1999 with an emission rate of 91.2 g/s (Moeller, 2001). The magnitude of the problem becomes larger during east wind (berg wind) conditions.

2.2 Dust theory

OSHA (2008) defines dust as finely divided solids that may become airborne from the original state without any chemical or physical change other than fracture. Dust is also defined as small solid particles conventionally below 75µm in diameter, which settle out under their own weight but which remain suspended for some time (Petavratzi *et al.*, 2005).

Dust is generated in a range of particle sizes. This study focused on particulate matter with an aerodynamic diameter less than 10 micron (PM_{10}). Aerodynamic diameter is defined as the diameter of a hypothetical sphere of density $1g/cm^3$ having the same terminal velocity in calm air as the particle of concern, regardless of its geometric size, shape and density (Petavratzi *et al.*, 2005).

2.2.1 Dust classification

Dust can be classified according to its environmental, occupational health and physiological effects. Through environmental effects, dust is classified as: generated, totally suspended dust, nuisance and fugitive dust. The physiological effect classes are: toxic dust, carcinogen, fibrogenic, explosive and nuisance dust. Occupational health effect classes are inhalable, thoracic and respirable dust (Petavratzi *et al.*, 2005). For the purpose of this thesis, dust is classified with regard to the occupational health effect.

Total inhalable dust is the fraction of airborne material, which enters the nose and mouth during breathing and is therefore available for deposition anywhere in the respiratory tract. Thoracic dust is defined as the fraction of inhaled particles that penetrate beyond the larynx (Petavratzi *et al.*, 2005). Respirable dust represents the fraction of dust particles that are small enough to penetrate the nose, upper respiratory system and deep into the lungs (OSHA, 2008). Particles can also be classified according to their sizes (Table 2.3).

Table 2.3: Dust classification according to particle sizes

Fraction	Size range
PM_{10} (thoracic fraction)	≤10 μm
$PM_{2.5}$ (respirable fraction)	≤2.5 μm
PM_1	≤1 μm
Ultrafine (UFP or UP)	≤0.1 μm
PM_{10}-$PM_{2.5}$ (coarse fraction)	2.5 μm – 10 μm

Wikipedia 2010(b)

PM_{10} is classified as thoracic dust while $PM_{2.5}$ is classified as respirable fraction. Particles that penetrate deep into the respiratory system are not removed by the natural clearance mechanisms of cilia and mucus and are more likely to be retained (OSHA, 2008).

2.2.2 Impacts of dust

Dust has a potential to cause negative effects particularly on human health and the environment. Besides, it can also affect the productivity of the mining operations and the safety of the workers.

Impacts on human health

Dust has been documented through the years as one of the biggest occupational "killers" (Petavratzi *et al.*, 2005). A wide range of occupational diseases may develop in mine workers depending on the physical, chemical and toxicological properties of the inhaled dust. The effects of exposure to dust are more serious when silica is a component of respirable dust. Silica in dust causes a disease called silicosis. Crystalline silica in respirable dust causes the death of more than 250 workers in the US each year (Reed, 2005). Exposure to dust containing respirable quartz can lead to lung emphysema and cancer (Inyang and Bae, 2006).

Workers at uranium mines are at a risk of inhaling respirable dust which is rich in silica, radionuclides and their decay progeny which can lead to chronic diseases. Crystalline silica is known to have an effect of decreasing the active life of macrophage resulting in less controlled accumulation of dust in alveoli. This decreases the oxygen exchange capability of the lung's alveoli due to a reduction in the lung tissue's elasticity (Moeller, 2001).

Another health implication caused by the inhalation of respirable dust associated specifically with uranium, thorium and vanadium ores is excess lung cancer (Petravatzi *et al.,* 2005). Several studies have concluded that short term increase in the concentration of PM_{10} by 10µg m^{-3} is associated with 0.5 to 1.5 percent increase in daily mortality, higher hospitalisation and health-care visits for respiratory and cardiovascular disease and enhanced outbreaks of asthma and coughing (Jacobson, 2002).

Impacts on environment

The effects of dust on agriculture and ecology of an area depends on the size distribution, the deposition rate and the concentration of dust particles in the ambient air. The effects of particulates matter (PM) on vegetation further depends on the constituents of PM (Grantz *et al.*, 2003). Other effects of PM_{10} on vegetation are: reductions in growth, yield, flowering and reproduction of plants. Dust can also have an effect on natural communities by altering the competitive balance between species in a community (Farmer, 1993). Heavy dust coating on vegetation can abrade plant surfaces; bury organisms and photosynthetic organs (Grantz *et al.*, 2003). In addition, heavy metals and other constituents of PM can reach the soil affecting the nutrient cycling important for plant growth and health of biota. Particulate emission can also contribute to climate change since the small particles in the atmosphere can absorb and reflect radiation from the sun affecting the cloud physics in the atmosphere (Reed, 2005).

Impacts on safety and productivity

Small particles in the air are known to reduce visibility. Small particles scatter and absorb light as it travels to the observer from the source. This action results in extraneous light from the sources other than the observed object being detected by the observer, hence impairing visibility (Reed, 2005).

Poor visibility caused by high levels of dust in the air from the pit, can affect the safety of employees. This impact is more serious at night due to low light and during windy conditions. However, this impact is usually due to short term high emissions episodes such as blasting (NSW, 2006). Dust can also reduce productivity and cause equipment and machinery damage. When dust is deposited on machinery and equipment, it reduces their life cycle and increases regular cleaning (Kotze, 1999).

Rössing Mine maintains a safe working environment at the mine and treats the safety of its workers and contractors with high priority. In 2008, a total of 2.9 million hours free of lost time injury incidents were achieved (RUL, 2008). In an attempt to offset and minimise the effects of dust emissions, various dust control measures have been put into place at the mine and they are discussed in section 2.7.

Impacts on operational cost

Dust can affect the haul cycle time by influencing the haul truck operator efficiency through unsafe and unfavourable working conditions that may be caused by dust emissions. This affects the overall productivity of mining operation and eventually reduces the money generated. As mentioned above, dust increases the frequency of maintenance of equipment; it thus increases the equipment maintenance and replacement costs (Moeller, 2001). High dust generation rates can slowly remove the wearing course of the haul road, thus increasing the rolling resistance between the haul truck wheel and the haul road increasing the cost of road maintenance. Further, the increase in rolling resistance increases fuel consumption (Kotze, 1999).

2. 3. Regulations and air quality standards for PM_{10}

The aim of air dispersion modelling is to quantify the impact of a certain facility or activity on the atmosphere. The impact is quantified by comparing the predicted concentration of the pollutant at ground level to a reference level. The most commonly used reference for comparison is the ambient air quality standards and limits (Thomas 2008).

2. 3.1 Ambient Air quality standards for PM_{10}

Air quality standards and limits have been developed worldwide with the aim of protecting the health of employees and the general public. There is no international air quality standards for PM_{10}, hence a number of countries have developed their own standards. The World Health Organisation (WHO) has established air quality guidelines (Cooper and Alley, 2002). However, standards of most countries are less stringent than those of the WHO. For example, the PM_{10} 24-hour average for the US and China is 150µg/m^3 and 100 µg/m^3 respectively as compared to 50 µg/m^3 for

WHO (Table 2.4). In few cases, national air quality standards for some countries are stricter than those established by WHO. For instance, the annual mean standard for PM_{10} in Scotland is 18 µg/m^3 as compared to 20 µg/m^3 for WHO.

In addition, organisations such as the European Union (EU) have established air quality standards and limits on particulate matter and other substances which apply to all member states (Petavratzi *et al.*, 2005). Air quality legislation and standards in the US are well established; consequently many countries have adopted its standards and practices.

The Namibian environmental legislation is still at an infancy stage and no national air quality standards have been developed yet. The first Environmental Management Act (EMA) was enacted in 2007 as the Environmental Management Act (Act No. 7 of 2007) (Government Gazette No.3966). The EMA describes various rights that citizens have, including the right to an environment that does not pose threat to human health. Rössing Uranium Mine has established its own air quality standards which are equivalent to the South African standards, as Namibian national standards have not yet been established.

Table 2.4: Air quality standards of various organisations and countries

Country/organisation	**Limit concentrations**	**Averaging times**
UK	20 µg/m^3	annual mean
Australia[2]	50 µg/m^3 -	24-hour mean annual mean
Scotland	18 µg/m^3	annual mean
EU	50 µg/m^3 20 µg/m^3	24-hour mean Annual mean
US	150 µg/m^3 revoked	24-hour mean Annual mean

China[1]	100 µg/m^3 150 µg/m^3	24-hour mean Annual mean
WHO	50 µg/m^3 20 µg/m^3	24-hour mean Annual mean
South Africa[2]	180 µg/m^3 60 µg/m^3	24-hour mean annual mean
Rössing Uranium mine[3]	180 µg/m^3 60 µg/m^3	24-hour mean annual mean

CERC, (2007), [1]Inyang and Bae, (2006), [2]Thomas (2008), [3]Kadhila-Amoomo, (2009)

2. 3.2 Occupational exposure limits for PM_{10}

The aim of occupational exposure limits (OEL) is to prevent or limit the exposure of workers to dangerous substances at workplaces as well as to protect them from such substances (Petavratzi *et al.*, 2005). A number of countries have developed their own occupational exposure limit systems, while some have adopted well established systems like the American Conference of Governmental Industrial Hygienists (ACGIH) limits which are called threshold limit values (TLVs).

In order to evaluate the hazard of exposure to mineral dusts, the content of quartz or other crystalline form of free silica must be considered. The TLVs will therefore vary depending on the percentages of free silica in dust (OSHA, 2008). OELs for different countries for both respirable inert and quartz dust are listed in Table 2.5.

Table: 2.5. OELs for different countries for both respirable inert and quartz dust

Country	Respirable inert dust (mg/m^3)	Respirable quartz dust (mg/m^3)	OEL type
Denmark	5	0.1	TLV
Finland	0.2	0.1	OES
United kingdom	4	0.1	Workplace exposure limits
Italy	3	0.05	TLV (based on ACGIH)
Portugal	5	0.05	TLV

Source: IMA-Europe, 2009

For the USA, different organisations like OSHA and NIOSH have developed different occupational exposure limits and they all differ from each other (table 2.6).

Table: 2.6. US occupational exposure limits

Organisation and OEL type	Quartz-TWA
OSHA PEL	$(10mg/m^3)/(\% SiO_2+2)$
NIOSH PEL	$0.05mg/m^3$)
ACGIH TLV (recommended guideline not enforceable)	$0.05mg/m^3$

Source: Fung, 2005

Rössing operations have to meet Rio Tinto (RT) occupational health standards. The RT standards for dust are $10mg/m^3$ for Inhalable dust; 3 mg/m^3 for Respirable coal dust and 5 mg/m^3 for Respirable dust (other) (Rio Tinto, 2003).

2. 4. Air dispersion modelling theory

Dispersion modelling uses mathematical equations describing the atmosphere, dispersion, chemical and physical processes influencing a pollutant released from sources of a given geometry to calculate concentrations at various receptors as a result of the release (Holmes and Morawska, 2006).

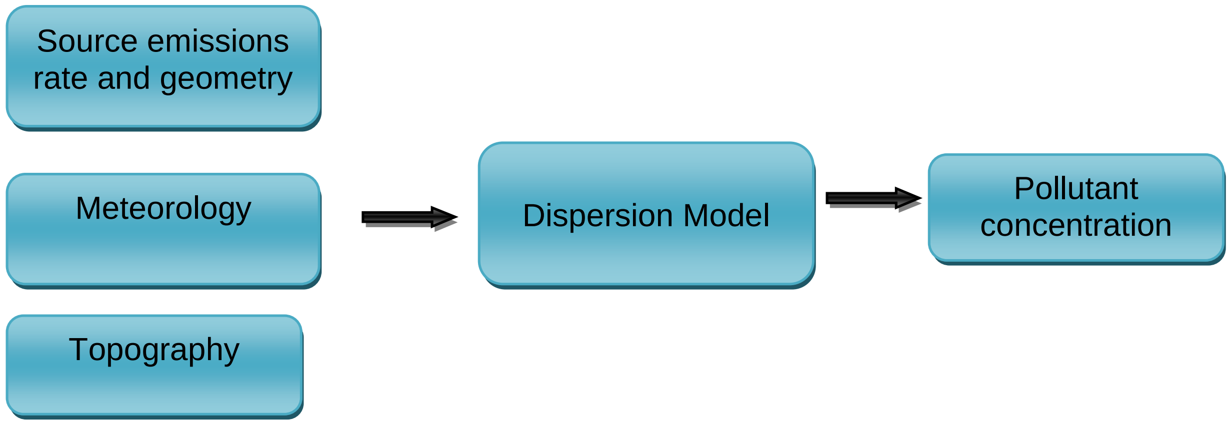


Figure 2.7: A schematic representation of the input-output of an air dispersion model

Dispersion models require two types of data inputs: information on the source or sources including pollutant emission rates, and meteorological data (Kanevce and Kanevce, 2006). In addition, they also need information on the topography of the study area (Figure 2.7). The models then use this information to simulate mathematically the pollutant's transport and dispersion. The output is air pollutant concentrations, for a particular time period, usually at specific receptor locations (Kanevce and Kanevce, 2006).

2.4. 1. Mechanisms of pollutants dispersion in the atmosphere

A simple example of pollutants dispersion in the atmosphere is through molecular diffusion, when matters move from a region of high concentration to a region of low concentration. However, apart from molecular diffusion, plumes spread due to other complex processes. These processes are mechanically and thermally generated turbulence and wind fluctuations (Cooper and Alley, 2002).

2. 4.1.1 Turbulence

Molecules of pollutants in the air are transported from one point to another by means of turbulence. Turbulence is defined as a collective random motion involving a group of many molecules (Turner, 1994). Turbulence is made up of both thermal and mechanical eddies. Eddies are macroscopic random fluctuations from the "average" flow (Cooper and Alley, 2002). These turbulent eddies are responsible for the dispersion of pollutants in the atmosphere. Eddies disperse pollutants by intercepting the plume, replacing a batch of concentrated pollutants in a plume with a batch of clean air from a distance away from the plume, consequently diluting the plume and spreading it in both vertical and lateral directions (Cooper and Alley, 2002).

Mechanical turbulence

Mechanical turbulence is created through the interaction between the horizontal force exerted by one layer on an adjacent layer and the gradient of the mean velocity with height (Venkatram, 2008). The stronger the wind or the larger the roughness elements , the greater the mechanical turbulence hence rough surfaces such as forests or trees produce more eddies than smooth surface such as ice (Cooper and Alley, 2002) Buildings, trees and other obstacles increases mechanical turbulence because these obstacles increase the horizontal forces that slow down the mean wind (Venkatram, 2008).

Thermal turbulence

The thermal energy generated from the sun is absorbed by the ground. The absorbed heat is transferred into the lower atmosphere by means of conduction and/or convection thus generating thermal eddies. More eddies are created when there is strong insulation than when the energy from the sun is weak (Cooper and Alley, 2002).

2.4.1.2. Wind fluctuations

Plume dispersion can also be caused by random shift in the wind. Pollutant concentrations are measured over a certain period of time called averaging time, for example, an averaging time of an hour. The wind direction and speed change during this period and more or less pollutant is blown towards the receptor. As a result, these random fluctuations cause the spread of the plume over a large area downwind of the source (Cooper and Alley, 2002). As the plume travels downwind of the source, the pollutant spreads further in the y and z directions, and the maximum concentration eventually decreases.

2.4.2. Types of models

The modelling of pollutants dispersion in the atmosphere is carried out by using mathematical algorithms. There are several basic mathematical algorithms some of which include: Box models, Gaussian model, Lagrangian and Eulerian model (Reed, 2005). These models differ in the type of pollutant accommodated, pollutant source type and whether they use plume or puff approach.

2. 4. 2. 1. Box model algorithm

The Box model is the simplest of all modelling algorithms which is based on the conservation of mass. The airshed is treated as a box into which pollutants are emitted and where they undergo chemical and physical processes. The air inside the box is assumed to have a homogenous concentration. The model uses that assumption to estimate the average pollutant concentration anywhere within the airshed (Wikipedia, 2010). The following equation represents the Box model:

$$\frac{dCV}{dt} = QA + uC_{in}WH - uCWH \qquad (2.1)$$

Where:

Q = pollutant emission rate per unit area
C = homogenous species concentration within the airshed
V = volume described by the box
C_{in} = species concentration entering the airshed
A = horizontal area of the box (L x W)
    L = length of the box
    W = width of the box

u = wind speed normal to the box

H = mixing height

Although this model is useful, it is unsuitable for the modelling of particle concentrations since it simulates the formation of pollutants within the box without providing any information on the local concentrations of the pollutants (Holmes and Morawska, 2006).

2. 4. 2.2. Lagrangian model algorithm

Lagrangian models are similar to Box models in a sense that they define an airshed as a box containing an initial concentration of pollutants. However, the Lagrangian model then follows the trajectory of the box as it moves downwind. The Lagrangian model then calculates the air pollution dispersion by computing the statistics of the trajectories of a large number of the pollution plume parcels. The Lagrangian model uses a moving frame of reference (Wikipedia, 2010). The Lagrangian equation has the following form (Reed, 2005):

$$\langle c(r,t)\rangle = \int_{-\infty}^{t}\int p\left(r,t\middle|r,^{'}t^{'}\right)S\left(r,^{'}t^{'}\right)dr^{'}dt' \qquad (2.2)$$

Where:

‹c(r, t)› = average pollutant concentration at location *r* at time *t*

S (r’, t’) = source emission term

P (r, t| r’, t’) = the probability function that an air parcel is moving from location r’ at time t’ to location r at time t.

The disadvantage of Lagrangian model is that they are limited when results from its prediction are compared with actual measurements, because measurements are made at stationary points, while the model predicts pollutant concentration based upon a moving reference grid (Reed, 2005).

2. 4. 2. 3. Eulerian model algorithm

The Eulerian model is similar to a Lagrangian model because it also tracks the movement of a large number of pollution plume parcels as they move from their initial location. However, they differ in as sense that the Eulerian model uses a fixed three dimensional Cartesian grid as a frame of reference rather than a moving frame of reference. The Eulerian models solve an equation of conservation of mass for a given pollutant. The equation generally follows the following form (Reed, 2005):

$$\frac{\partial \langle c_i \rangle}{\partial t} = -\overline{U}.\nabla\langle c_i \rangle - \nabla.\langle c_i' U' \rangle + D\nabla^2\langle c_i \rangle + \langle S_i \rangle \qquad (2.3)$$

Where:

$U = \bar{U} + U'$

U = windfield vector U(x, y, z)

$\bar{U}$ = average wind field vector

U' = fluctuating wind fields vector

c = ‹c› + c'

c = pollutant concentration

‹c› = average pollutant concentration; ‹› denotes average

c'= fluctuating pollutant concentration

D = molecular diffusivity

S_i = source term

The term $-\overline{U}.\nabla\langle c_i \rangle$ is hyperbolic, the turbulent diffusion is parabolic and the source term is generally defined by a set of differential equations making it difficult to solve. This type of equation can be computationally expensive to solve (Reed, 2005).

2. 4. 2. 4. Gaussian plume model

Gaussian type models are the most common dispersion models used in atmospheric dispersion modelling. The term "Gaussian" refers to the statistical concept in which a group of arranged values follows a bell-shaped curve distribution (Cora and Hung, 2003).This type of model assumes that the pollutant disperses according to the normal statistical distribution (Holmes and Morawska, 2006). At the point of release, the pollutant concentration is at maximum and decreases in both lateral and vertical directions following the normal distribution. The two models used in this comparative study were developed based on Gaussian plume. The Gaussian model uses a Gaussian equation which is used for point source emissions in general (Cooper and Alley, 2002):

$$C(x,y,z,H) = \frac{Q}{2\pi u\ \sigma_y \sigma_z} \exp\left(-\frac{y^2}{2\sigma_y^2}\right) \exp\left(-\frac{(z-H)^2}{2\sigma_z^2}\right) \exp\left(\frac{(z+H)^2}{2\sigma_z^2}\right) \qquad (2.4)$$

Where:

C = steady-state concentration at a point (x, y, z), µg/m^3

Q = pollutant emission rate, µg/s

U_s = mean wind speed at release height

$\sigma_y \sigma_z$ = standard deviation of lateral and vertical spread parameters, n

y = horizontal distance from plume centreline, m

H = effective stack height (H = h +Δh) where h = physical stack height and

Δh = plume rise,

z = vertical distance from ground level, m

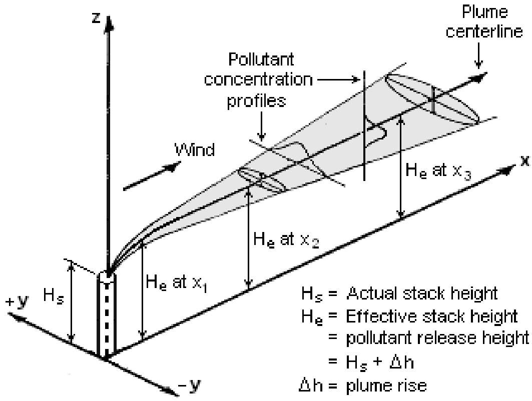


Figure 2.8: Graphical representation of double Gaussian distribution in the plume. (Vannucci *et al*, 2008)

The first exponential term represents the lateral dispersion and vertical dispersion is described by the second exponential term. The terms σ_y and σ_z in equation 2.4

represent the standard deviation of the horizontal and vertical distributions of the plume of the pollutant. High standard deviation values would result from an unstable, turbulent atmosphere, whereas low values would occur in less turbulent atmospheric conditions (Tshukudu, 2003). In older models, these coefficients are defined by stability classes created by Pasquill and Gifford and they increase as the downwind distance increases (Holmes and Morawska, 2006).

The Gaussian model is based on the following assumption: the emission must be constant and uniform; the wind direction and speed are constant; net downwind diffusion is negligible compared to vertical and crosswind diffusion; the terrain is relatively flat; there is no deposition or absorption of the pollutant and the vertical and crosswind diffusion of the pollutant follow a Gaussian distribution (Reed, 2005).

Gaussian plume models have a limitation when they are applied to particle dispersion modelling. This limitation is a result of the use of steady state approximations without taking into account the time required for the pollutant to travel to the receptor and the vertical particle movement due to gravity during this time (Holmes and Morawska, 2006). However, in recent years, advanced Gaussian models have been developed that overcome most of the limitations in Gaussian models developed earlier. AERMOD and ADMS are the new generation models developed with advanced algorithms to overcome the early Gaussian model limitations.

2.4.3. Factors affecting dispersion of pollutants in the atmosphere

There are a number of factors that can affect the dispersion of pollutants in the atmosphere and these include: meteorology, topography and atmospheric stability.

2.4.3.1 Meteorology

Meteorology is a vital element of atmospheric dispersion modelling because it determines the diluting effects of the atmosphere (Kanevce and Kanevce, 2006). The dispersion, transformation and removal of pollutants in the atmosphere depend on the meteorological conditions of the site. Hence, good and appropriate meteorological data preferably from a weather station within the area of interest is needed in order to achieve the best results from modelling (D'Abreton, 2009). The important meteorological data needed for modelling are: temperature, wind speed, wind direction, cloud cover and atmospheric stability.

Air temperature

Temperature affects the buoyancy of the plume since the higher the temperature difference between ambient air and the plume, the higher the plume will rise (D'Abreton, 2009). This in turn reduces the ground level impact of pollutants. Temperature is also important for the development of the mixing and inversion layer (Thomas, 2008).

Wind speed

Wind speed is one of the important meteorological parameters in dispersion modelling. Wind speed influences initial dilution of the plume leaving a source, hence the stronger the wind speed, the more rapid the dilution of the pollutants and thus the lower the concentrations at the ground level and vice versa (Thomas, 2008). Mechanical turbulence that increases mixing and dilution is created by the wind and the higher the wind speed the stronger the mechanical turbulence (Colls, 2002).

Wind direction

Wind direction determines the direction in which the pollutants released in the atmosphere are transported (Turner, 1994). In this study it played a major role since it was used to determine the monitoring point where the monitor was set. Only monitoring points or receptors downwind of the source are affected by the plume emitted. Wind direction together with other meteorological parameters determines the spatial pattern of average ground level concentration.

2.4.3.2 Atmospheric stability

Atmospheric stability influences the vertical movement of particles in the atmosphere which is also influenced by the temperature effect of the air (Cora and Hung, 2003). Atmospheric stability is defined as the atmospheric tendency to resist or enhance vertical motion or alternatively suppress or augment existing turbulence (Zoras *et al.*, 2006). Over 40 years ago, Pasquill introduced a method of estimating atmospheric stability accounting for both mechanical and thermal turbulence. Atmospheric stability was classified into six categories ranging from A (very unstable) to F (very stable).

The categories were developed based on the wind speed, solar radiation (daytime) and cloud cover (at night). Strong insulation leads to the heating of the ground increasing the temperature of the lower part of the atmosphere, creating an unstable condition. If the wind speed rises, the vertical mechanical mixing becomes stronger than the buoyancy effects, leading to neutral stability. During the night the ground cools creating stable conditions (Colls, 2002). Table 2.1 below shows the stability classes developed by Pasquill.

Table 2.7 Pasquill-Gifford stability classes

Wind speed(m/s)	**Daytime incoming insolation (Wm^{-2})**			**Night cloudiness**	
	Strong (>590)	Moderate (300-590)	Slight (<290)	Cloudy (≥4/8)	Clear (≤3/8)
<2	A	A-B	B	E	F
2-3	A-B	B	C	E	F
3-5	B	B-C	C	D	E
5-6	C	C-D	D	D	D
>6	C	D	D	D	D

Adopted from Copper and Alley, 2004

The Pasquill stability classes have some disadvantages. The six distinct stability classes, A to F, do not account for the continuous nature of turbulent intensities. It does also not take into consideration the variations in surface properties, such as roughness, length and albedo, which are important in determining the relation between meteorological observations and the turbulence properties that control dispersion (Venkatram, 2008).

However, the two models used in this study use other parameters to estimate atmospheric stability. The atmospheric stability in ADMS is described based on boundary layer height *h* and the Monin-Obukhov length L_{MO} and atmospheric dispersion is estimated from these two parameters (CERC, 2007). AERMOD makes use of three parameters to describe stability. The parameters are: Albedo (the fraction of total incident solar radiation reflected by the surface); Bowen ratio (The ratio between sensible heat (due to conduction and convection) and latent heat (due to phases changes)); and surface roughness length (the height at which the mean horizontal wind speed is zero). In the end, AERMOD uses these parameters in the calculations of *h* and L_{MO} (Venkatram, 2008). Discussion on parameters used for atmospheric stability estimation for the two models are discussed in subsequent sections in this chapter.

2.4.3.3 Topography

Topography also influences the dispersion of air pollutants. The term “topography” refers to the surface features of land, including the configuration and elevation of both man-made and natural features (Cora and Hung, 2003). Topographical features may impede the dispersion of pollutants, especially when the pollutants are released in low-lying areas (Cora and Hung, 2003). Surface roughness, buildings, hills, trees and obstructions are some of the topographical features that can affect pollutant dispersion in the atmosphere. The effect of surface roughness on dispersion is further discussed briefly in the next paragraph.

Surface roughness

When wind flows over a surface, objects on that surface will have frictional effects on the wind speed close to the surface. A parameter called a surface roughness length z_0 is used to show the magnitude of this effect (Turner, 1994). Surface roughness length is defined as the height at which wind speed goes to zero (0), based on theoretical logarithmic profile (Brode, 2006). It ranges from less than 0.001m (1mm) over water to 1.0m or higher for forests and urban areas. Table 2.4 gives values of surface roughness length for various land use categories at different seasons of the year.

Table 2.8: Surface roughness length by land use and season (in meters)

Land use	Spring	Summer	Autumn	Winter
Water (fresh and sea)	0.0001	0.0001	0.0001	0.0001
Deciduous forest	1.00	1.30	0.80	0.50
Coniferous forest	1.30	1.30	1.30	1.30
Swamp	0.20	0.20	0.20	0.20
Cultivated land	0.03	0.20	0.05	0.01
Grassland	0.05	0.10	0.01	0.001
Urban	1.00	1.00	1.00	1.00
Desert shrub land	0.30	0.30	0.30	0.15

(Li, 2009)

2.5 Review of models used in the study: AERMOD and ADMS

2.5.1 AERMOD

AERMOD is a steady-state Gaussian plume model that incorporates air dispersion based on planetary boundary layer (PBL) turbulence structure and scaling concepts. It includes treatment of both surface and elevated sources and both simple and complex terrain (EPA, 2004). It is applicable to rural and urban areas, and multiple sources including point, area, and volume sources (Vora, 2010).

The concentration distribution in the stable boundary layer (SBL) is assumed to be Gaussian in both vertical and horizontal planes. The American Meteorological society (AMS) defines SBL as a cool layer of air adjacent to a cold surface of the earth, where temperature within that layer is statically stably stratified. In convective boundary layer (CBL), the horizontal distribution is assumed to be Gaussian while the vertical distribution is described with bi-Gaussian probability density function (pdf) (Cimorelli *et al*, 2004). AMS defines CBL as a type of atmospheric boundary layer characterized by vigorous turbulence tending to stir and uniformly mix, primarily in the vertical, quantities such as conservative tracer concentrations, potential temperature and momentum or wind speed.

AERMOD modelling system comprises a meteorological pre-processor (AERMET), a terrain pre-processor (AERMAP) and the dispersion model (AERMOD) (see figure 2.9).

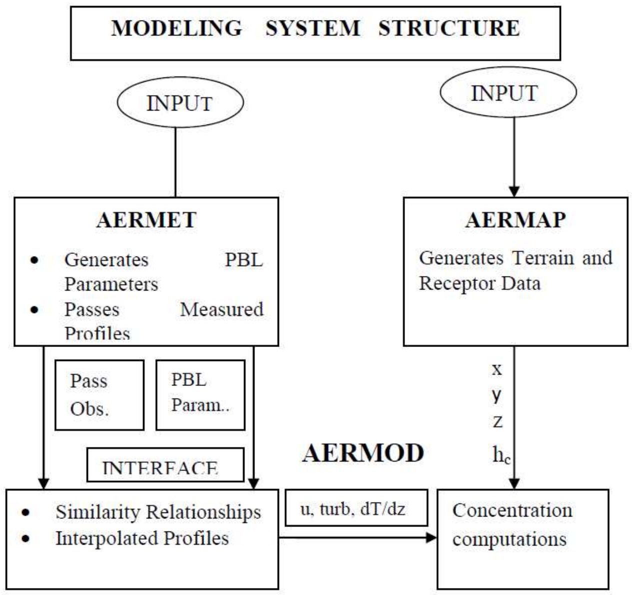


Figure 2.9: The flow and processing of information in AERMOD (Vora, 2010)

AERMET provides AERMOD with the meteorological information needed to characterise the PBL. AERMET requires standard meteorological observations such as wind speed, wind direction, temperature and cloud cover. It also needs the surface characteristics parameters of albedo, surface roughness and Bowen ratio. It then makes use of this data for the calculations of planetary boundary layer (PBL) parameters such as: Mixing height (z), Monin – Obukhov length (L), temperature scale, convective velocity scale (w) and surface heat flux (H) (Cimorelli *et al.*, 2004).

The information from AERMET is passed on to AERMOD where similarity theories are used to calculate lateral and vertical turbulent fluctuations (v, w), vertical profiles of wind speed (u) and potential temperature gradient (dθ/dz).

AERMAP is used to calculate the terrain height scale (h_c) for each receptor location, which is used to calculate the dividing streamline height. AERMAP also generates receptor grids for AERMOD. The input to AERMAP is the topographical data in a format of Digital Elevation Mapping (DEM) files. The information generated from AERMAP is then passed on to AERMOD as the location of receptors, the receptor's height above mean sea level and the receptor specific terrain height scale (h_c) (Cimorelli *et al.*, 2004).

AERMOD then uses this information from the two pre-processors to compute concentrations of pollutants, taking into account the changes in dispersion rate with height and making use of non-Gaussian plume in convective conditions (Paine *et al.*, 1998).

2.5.2 ADMS

ADMS is a short-range dispersion model that simulates a wide range of buoyant and passive releases to the atmosphere either individually or in combination. It is a new generation dispersion model using two parameters, namely, the boundary layer height *h* and the Monin-Obukhov length L_{MO} to describe the atmospheric boundary layer and using a skewed Gaussian concentration distribution to calculate dispersion under convective conditions (CERC, 2007).

ADMS has been developed to simulate the dispersion of buoyant or neutrally buoyant gases and particulate emissions to the atmosphere (Carruthers *et al.*, 1994). The model has a fully integrated meteorological pre-processor. The ADMS suite also contains ADMS Mapper which enables users to visualise model set up and to create and edit sources, receptor and buildings (CERC, 2007). The model is applicable up to 60 km downwind of the source and provides useful information for distances up to a 100 km (CERC, 2007).

ADMS characterises the boundary layer using the Monin-Obukhov length L_{MO} and boundary layer height *h* and not by a Pasquill-Gifford stability class. Stability in ADMS corresponds to:

Table 2.9: stability categories in ADMS

Stable	$h/L_{mo} \geq 1$
Neutral	$-0.3 \leq h/L_{mo} < 1$
convective	$h/L_{mo} < -0.3$

Kanevce and Kanevce, 2006

Monin-Obukhov length

The Monin-Obukhov length is a measure of the depth of the near-surface layer in which shear effects are likely to be significant under any stability condition (Kanevce and Kanevce, 2006). It is defined by:

$$L_{mo} = \frac{-u_*^{\,3}}{\left(\dfrac{\kappa g F_{\theta 0}}{\rho c_p T_0}\right)} \qquad (2.5)$$

Where:

u_* = friction velocity at the earth surface,

□ = is the von Karman constant (0.4)

g = gravitational acceleration

$F_{\theta O}$ = is the surface sensible heat flux

c_p= specific heat capacity of air

ρ= density of air

T_O = near-surface temperature

During unstable conditions, the Monin-Obukhov length is negative and it is measured as the height above which turbulent motions caused by thermal turbulence is more important than mechanical turbulence (CERC, 2007). In stable conditions, the Monin-Obukhov length is positive and it is then measured as the height above which stable stratification inhibits vertical turbulent motion (CERC, 2007). Figure 2.10

shows the ADMS representation of various Monin-Obukhov lengths with corresponding Pasquill-Gifford stability categories.

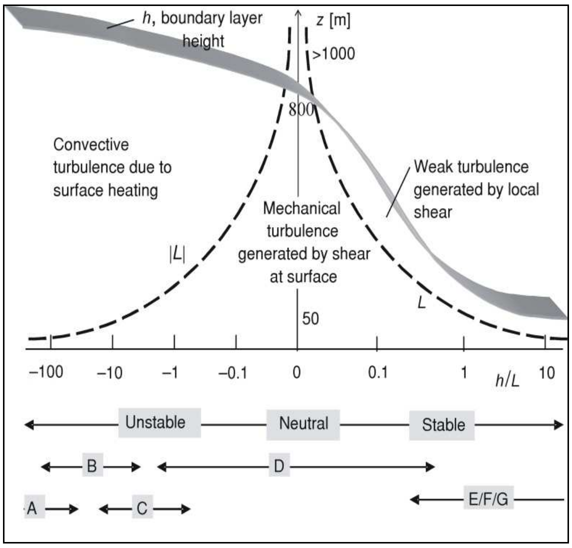


Figure 2.10: Variation of Monin-Obukhov length and boundary layer height with atmospheric stability (Kanevce and Kanevce, 2006).

2.6. Model and inter-model validation studies

2.6.1 AERMOD studies

AERMOD has undergone a wide-ranging evaluation for its performance, in order to evaluate how well the model estimates the concentration by comparing it against various independent databases and field data. Some of the validation studies are described below.

Kesarkar (2006) evaluated the performance of AERMOD using gaseous pollutants in the study that was conducted to understand the dispersion of PM_{10} over Pune in India. In this study AERMOD was coupled with a regional weather prediction model (WRF). The planetary boundary layer and surface layer parameters required by

AERMOD were computed using the WRF model. The result from the study showed that the concentrations were under predicted in the modelling process over the city.

Sivacoumar *et al.*, (2009) used AERMOD together with FDM and ISCST3 model in the study involving the simulation of fugitive dust emissions and control measures in stone crushing industry. In this study the performances of the models were evaluated against distance of the impact zone. The impact zone for measured concentration varied from 211 to 1350 m with a mean of 784 m. The impact zone from measured concentration was compared to that of predicted concentrations of FDM, ISCST3 and AERMOD and they varied 153–2650 m, 143–1056 m, and 135–1225 m with a mean of 1335 m, 501 m and 679 m respectively. The study concluded that AERMOD showed a better performance over the other two models.

There is a database that has been developed for model validation containing data from dispersion model validation experiments that were conducted using the ^{85}Kr released from the BNFL Sellafield site as a tracer. This database has been used to provide a validation of the regulatory models: ADMS, AERMOD and NRPB R91. The statistical tests showed a general trend of improvement in model performance when building and terrain modules were used (Hill *et al.,* 2001).

2.6.2 ADMS studies

During its development, ADMS has been validated against datasets including wind tunnel datasets. The performance of ADMS together with AERMOD and ISC3 was evaluated using three datasets: Kincaid, Indianapolis and Praire Grass (CERC, 2005).

Kincaid Power Plant - The Kincaid dataset consists of 171 studies that were performed at the Kincaid power station where SF_6 was released from a 187m tall stack. This power station is surrounded by flat terrain. AERMOD showed the predicted mean of 50% of the observed mean. ADMS showed better performance than other models.

The Indianapolis Power Plant – This dataset consists of 170hours of SF_6 tracer experiments carried out for EPRI (Electric Power Research Institute) in 1985 at the Perry Power plant on the outskirts of Indianapolis. The predicted concentrations from the models were compared to the observed data from the experiment. The results show that ADMS slightly overestimated the mean and standard deviation of the data. However, AERMOD under-estimated the mean by 57% and over estimated the standard deviation by 45%. ISCST3 predicted a mean approximately one and half times the observed.

Prairie Grass – A project called Prairie Grass which was designed by Air Force Cambridge Research Centre personnel was carried out in North Central Nebraska in 1956. This site was located on flat land covered with natural prairie grasses. Small quantities of SO_2 tracer were released over 10 minutes period from near ground level. About 35 trials out of 70 were conducted during convective condition (daytime) and the rest were done at night with temperature inversions present (stable conditions). The mean concentration predicted by AERMOD is identical to the observed mean; this can however be due to fact that Prairie grass results have been used directly in the AERMOD model formulation. ADMS under-estimated the mean concentration slightly predicting approximately 82% of the observed mean, however the correlation of all models was good.

Dunkerley *et al.*, (2001) – ADMS has been used in the inter comparison study between AERMOD, ADMS and ISC for the purpose of assessing the effects of terrain on dispersion. The terrain selected was that of Porton Down in UK. The performance of the three models was compared in six cases under different meteorological conditions. The results showed that under neutral stability conditions, ADMS prediction was constantly lower than AERMOD under flat terrain. Under stable conditions, the ADMS maximum ground level concentration predictions were much smaller than AERMOD's and also smaller than ADMS corresponding results under neutral and unstable conditions. The study concluded that the three models use different methods to account for the effect of the terrain on dispersion which generates correspondingly diverse results.

Carruthers *et al.*, (1994) carried out study that compared ADMS to the Chinese Guideline Air Dispersion Model (CGM). The comparison focused on how the two models predict the dispersion of pollutants from a source where no initial buoyancy and momentum was considered. Sources near ground level, 50m and 200m above the ground were considered for very unstable, neutral and stable conditions. Cases where no plume rise was modelled, the models tended to show the greatest difference for low sources with GCM showing much greater concentrations for unstable, neutral and stable flows. Differences are smaller but still significant for the elevated sources but ADMS show maximum concentrations considerably nearer to the source than CGM especially for unstable and neutral conditions. The latter can be attributed to the fact that ADMS generally exhibits faster mixing spreading of the plumes which are elevated for elevated sources resulting in plume reaching the ground more quickly. However, the study concluded that ADMS produces more accurate concentration predictions than CGM.

Harsham and Bennett (2008) conducted a sensitivity study for the validation of three regulatory dispersion models: ISC3, UK-ADMS and AERMOD. In this study lidar measurements were made for the dispersion of the plume from a coastal industrial plant over three weeks between September 1996 and May 1998. Where possible, each model was run according to choices between urban or rural surface characteristics; wind speed measured at 10 m or 100 m; and surface corrected for topography or topography plus buildings.The outputs from each model were compared to the results from the lidar measurements. All models underestimated dispersion in the near field and underestimated it beyond a few hundred. ISC3 showed the smallest dispersion while AERMOD gave the largest values for the lateral spread and ADMS gave the largest values for the vertical spread.

Broke *et al.* (2007) conducted a comparison study involving two versions of ADMS (3.1 and 3.3) and two versions of AERMOD (999351 and 04300). The results from the two models were compared to SO_2 measurements around groups of power stations in Yorkshire and Lower Trent Valley for the year 1998 and 1999. In addition, comparisons between the two models were also done for the area around Iron

Bridge Power Station (where terrain effect requires consideration) for the year 2003 and 2004. The results showed that most recent versions of the two models, that is, ADMS 3.3 and AERMOD 04300, agreed with the measured 1-hour SO_2 concentration statistics to within a factor of 2. In all the three study areas, both models showed a tendency to over-predict values for 1-hour concentrations at lower percentiles. At Yorkshire and Lower Trent Valley (in flat terrain) AERMOD tended to under-estimate these values. Around Iron Bridge (including terrain effects) both models tended to under predict the 1-hour concentrations above the 99.73 percentile.

2.7. Emission estimation

The emission of particulates is dependent on parameters such as meteorological conditions, emission control efficiency as well as on the material characteristics. In order to account for the amount of pollutant discharged into the atmosphere, an emission inventory has to be compiled. An emission inventory is an estimate of the quantity of emissions discharged to air for a given area. It includes a variety of contaminants and should include estimates for all major sources of those contaminants (NPI, 2001).

Estimation of emissions from various sources is facilitated by emission factors (USEPA, 1998). An emission factor "is a representative value that attempts to relate the quantity of a pollutant released to the atmosphere with an activity associated with the release of that pollutant" (USEPA, 1998). Emission factors are always expressed as a function of the weight, volume, distance or duration of the activity emitting the pollutant. The general equation used for estimation of emission is (USEPA, 1998):

$$E = A \times EF \times \left(1 - \frac{ER}{100}\right) \qquad (2.6)$$

Where:

E = emissions

A = activity rate

EF = emission factor

ER = overall emission reduction efficiency %

Particulate emissions from mining operations originate from various sources, and for each source an equation for estimating emissions has been developed. The USEPA has done extensive work on developing techniques and equations for emission estimation and they are available on their website, AP- 42 (http://www.epa.gov/ttnchie1/ap42/). In addition, the Australian NPI (National Pollutant Inventory) has manuals on emission estimation techniques including mining and their available on their website (www.npi.gov.au). However, the Australian work is not as comprehensive as the USEPA (NPI, 2001). There are other organisations and countries that have developed emission estimation equations and emission factors, for example, the EU. Further information on the EU emission estimation techniques and emission factors can be accessed by visiting this website: www.eea.europa.eu. Environment Canada through the National Pollutant Release Inventory (NPRI) also has well established emission factors and guidelines on how to report them. More information on this can be obtained from www.ec.gc.ca/inrp-npri.

The equations for estimating PM_{10} emitted from different sources used in this study were derived from the USEPA and NPI websites. Since there are no equations specifically developed for the mining of uranium, the equations used were adopted from coal mining due to the readily availability of this information.

2.7.1. Drilling and Blasting (EPA, 1998)

Emissions from drilling and at the open pit mine are considered to be insignificant contributors to the overall particulate emissions. No equation has been developed for this source except the total suspended particulate (TSP) emission factor of 0.59kg/hole drilled for uncontrolled emissions (USEPA, 1998). However, there is a weakness in this emission factor since it does not take into consideration the moisture content of the material drilled, the diameter and the depth of holes drilled (NPI, 2001).

Besides the emission factor for TSP, the USEPA (1998) does not provide the emission factor for PM_{10}. However, the NPI provides an emission factor of 0.31kg/hole for PM_{10} estimated from the PM_{10}/TSP mean fraction obtained from the Hunter Valley studies (NPI, 2001).

The equation that estimates emissions from blasting is given below (USEPA (1998):

$$E = 0.00022 \times A^{1.15} \tag{2.7}$$

Where:
E = emission factor (kg/blast)
A = area blasted in square metres

US EPA provides another equation which is used to calculate emissions from blasting and it is:

$$EF = 344 \times A^{0.8} \times M^{-1.9} \times D^{-1.8} \text{ (TSP)} \tag{2.8}$$

Where:
EF = emission factor kg/blast
A = Area blasted in m^2
M = moisture content in %
D = depth of blast holes in metre

In order to get the emission values for PM_{10}, the value obtained from the equation above should be multiplied by factor of 0.52 (conversion factor used to convert TSP emission factors to PM_{10} emission factors, (USEPA 1998)). Blasting and drilling were not considered in this study because their contributions to overall dust emission are generally low.

Control measures

Dust emissions from drilling can be reduced by using water (wet drilling) and 70% reduction efficiency can be achieved by this method (NPI, 2001). At Rössing Mine, 0.5m^3 of water is used per production hole and 0.25 m^3 of water per pre – split hole (Ihuhua, 2009) in an attempt to reduce dust emissions from drilling. Besides, dust emissions from drilling can be controlled by means of fitting each drill with a dust collector to extract the generated dust. This control method can achieve a control efficiency of up to be 99% (NPI, 2001).

2.7.2. Aggregate handling

Aggregate handling includes operations such as loading and offloading of materials. The main operation that handles materials in aggregate form is stockpiling. A considerable amount of dust is emitted at several points during the stockpiling of materials. These points include: material loading onto the pile, loads out from piles and emissions from movement of trucks and loading equipment in the stockpile area (USEPA, 2006). The equation used to estimate emissions from aggregate handling is given below (USEPA, 2006):

$$E = k \times 0.0016\left(\frac{U}{2.2}\right)^{1.3} \times \left(\frac{M}{2}\right)^{1.4} \qquad (2.9)$$

Where:

E = emission factor in kg/t

k= particle size multiplier (dimensionless) (the value of k for PM_{10} is 0.35)

u= mean wind speed, metre per second (m/s)

m = material moisture content (%)

This equation is applicable to the following source conditions:

Silt content (%)	**Moisture content (%)**	**Wind speed (m/s)**
0.44-19	0.25-4.8	0.6-6.7

Control measures

Loading of material into the trucks has no documented method to control dust emissions (NPI, 2001). Dust from dumping and tipping can be minimised by wet dust suppression using water sprays. However, the use of water spray with chemical agents such as surfactants provide more extensive wetting making it a more effective technique than water alone (USEPA, 1998). Table 2.5 shows the control methods used for dust suppression during aggregate handling activities with corresponding control efficiency.

Table 2.10 Control efficiency for different dust control methods

Activity	**Control measure**	**Control efficiency (%)**
Loading trucks	No control	-
Unloading trucks	Water sprays	70
Loading stockpiles	Water sprays Telescopic chute with sprays	70 -75

(NPI, 2001)

At Rössing Mine, there is no control measure of dust emitted when loading materials into the trucks and offloading at piles except at primary crushers where water sprays are used while the truck is tipping. There are a number of dust collectors installed at several points at the coarse ore stockpile and crushers that are aimed at collecting the dust emitted from these sources.

2.7.3. Unpaved road

The amount of dust emitted from a certain portion of an unpaved road varies linearly with the speed a vehicle travels. It also varies directly with the silt content of the surface material on the road. Silt content of the road material is the fraction of particles smaller than 75µm in diameter (USEPA, 1998). At mines where heavy duty vehicles and other heavy equipment travel on unpaved roads, emissions vary directly with the vehicle weight. In addition, the moisture content of the surface material on the road also affects the quantity of the dust emitted from the road, since dry materials are more susceptible to be blown up by the wind and the moist materials tend to conglomerate into big particles, thus reducing the emissions (USEPA, 1998). For heavy vehicles travelling on unpaved surfaces at industrial sites, emissions are estimated from the following equation (SKM, 2005):

$$E = k\left(\frac{s}{12}\right)^{0.8}\left(\frac{W}{2.7}\right)^{0.4}\left(\frac{M}{0.2}\right)^{-0.3} \qquad (2.10)$$

Where:

E = emission factor in kg/VKT

k = empirical factor

s = surface material silt content (%)

W = mean vehicle weight (tons)

M = Moisture content (%)

This equation was developed using the following source conditions (USEPA, 2006):

Surface silt content (%)	**Mean vehicle weight (ton)**	**Surface moisture content (%)**
1.8 – 25	2 -290	0.03 -13

The equation was later revised in 2001 removing the parameter of moisture content of the road surface. In 2003, a new equation was published (SKM, 2005):

$$EF_{10} = 0.423 \times \left(\frac{s}{12}\right)^{0.9} \times \left(\frac{W}{2.7}\right)^{0.45} \qquad (2.11)$$

Where:

EF = emission factor for PM_{10} (kg/VKT)

The revised equation (2.11) was used on the calculation of emission factors for the roads in this study.

Haul roads dust suppression

Dust from haul roads can be controlled by using the following methods (USEPA, 1998):

(a) Vehicle restrictions – limits on speed, weight or number of vehicles on the road.
(b) Surface improvement – paving, adding of slag to a road
(c) Surface treatment – watering or using chemical dust suppressants (such as tar and bitumen products; hygroscopic salts; petroleum resins etc. (Moeller, 2001).

Reducing the vehicle speed is an unattractive measure because it will decrease the overall mine productivity. Paving is not an economically attractive measure since most of the industrial roads are not permanent. Using materials that have low silt content like placing gravel on roads requires regular maintenance such as grading (USEPA, 1998).

Watering increases moisture content which agglomerates particles thereby decreasing the likelihood of particles becoming suspended when vehicles travel on the road surface (USEPA, 1998). The efficiency of watering depends on the amount of water added during each application, the application frequency, the weight, speed and number of vehicles travelling on the watered road, and meteorological conditions (USEPA, 1998). A control efficiency of 50% can be achieved for level one watering (2l\m^2\hr) and 75% for level two watering (>2l\m^2\hr) respectively (NPI, 2001).

Chemical dust suppressants reduce emissions by changing the physical characteristics of the existing road surface material (USEPA, 1998). Many chemicals form hardened surface that bind particles together. The disadvantage of using chemical suppressants is that it is costly (Petravatzi *et al.,* 2005) and has adverse effects on plant and animal life (USEPA, 1998), but they have less frequent reapplication requirements. A control efficiency of about 80% can be achieved when applied at a regular interval of 2 weeks to 1 month (USEPA, 1998).

At Rössing Mine, dust from haul roads is controlled by Dust-a-Side (DAS) on the main haul roads. Additionally, water is used at bench intersections and on roads leading to waste dumps and ore stockpiles. DAS is made up of an aqueous bituminous emulsion, which is used after it is diluted with water. The solution has a product-to-water ratio of 1:39 (Ihuhua, 2009). This product works by binding the wearing course material thus reducing the dust emitted from the haul roads (Moeller, 2001).

2.7.4. Wind erosion from active stockpiles

Wind erosion is defined as the movement of material by the wind and occurs when the lifting power of moving air is able to exceed the force of gravity and the friction which holds an object to the surface (Wiki, 2010). There are various factors that affect the extent of wind erosion. Some of the factors are aridity of climate, soil texture, soil moisture, soil structure and vegetation.

The texture of the soil affects the extent of wind erosion, for instance, coarse sand and gravelly or rocky soils are more resistant to wind erosion since the particles are too heavy to be removed by wind erosion. The soil moisture increases cohesion thus temporarily preventing the soil to be eroded by wind. Little structure improving matter on the soil makes the soil susceptible to wind erosion. Vegetation acts as a wind break by cutting the speed of wind at ground level (Roose, 1996).

Several field experiments that have been conducted using portable wind tunnels concluded that the threshold wind speeds exceeds 5m/s at 15cm above the surface. They have also indicated that erosion potential is directly proportional to the wind speed, that is, the high the wind speed, the high the erosion potential. Erosion potential is defined as the finite availability of erodible material (mass/area) (USEPA, 1998). The emission factor for wind generated particulates emissions resulting in erodible and non erodible surface material subjected to disturbance is calculated using this equation (USEPA, 1998):

$$EF = k \sum_{i=1}^{N} P_i \qquad (2.12)$$

Where:

EF = emission factor (g/m^2)

k = particle size multiplier

N = number of disturbances per year

Pi = erosion potential corresponding to the observed (or probable) fastest mile[1] of wind for the period between disturbances, g/m2

The erosion potential function for dry exposed surface is:

$$P = 58\ (u^* - u_t^*)^2 + 25\ (u^* - u_t^*) \qquad (2.13)$$

$$P = 0\ for\ u^* \le u_t^*$$

Where:

P = erosion potential function (g/m2)

u* = friction velocity (m/s)

u_t = threshold friction velocity (m/s)

Another equation used to estimate emissions from active stockpiles (adapted from coal mining) is as follow:

$$E = 1.9\left(\frac{S}{1.5}\right)365\left(\frac{365 - P}{235}\right)\frac{f}{15} \qquad \text{(NPI, 2001)} \qquad (2.14)$$

Where:

E = emission factor in kg/ha/year

S = silt content %

P = number of days when rainfall is greater than 0.25mm

[1] The fastest mile represents the wind speed corresponding to the whole mile of wind movement that has passed by the 1 mile contact anemometer in the least amount of time (USEPA, 1998).

f= percentage of time that wind speed is greater than 5.4m/s at the mean height of the stockpile

Control measures

The use of water sprays, wind breaks and enclosure are some of the control measures that can be used to reduce dust emissions from stockpiles. Using water alone provides a temporal slight reduction on emissions; however, using water mixed with chemical agents improves the wetting process (EPA, 1998). Water sprays, wind breaks and total enclosure can achieve control efficiencies of about 50%, 30% and 99% respectively (NPI, 2001).

Wind erosion from mining contributes very little to the overall dust emissions at Rössing Mine, hence it was left out from the calculations of emission factors in this study.

Chapter 3: Methodology

3.1 Data collection

3.1.1 Monitoring

PM_{10} was monitored at Rössing pit using Trackpro 3.6.0 SidePak aerosol monitor AM510 from 13 July 2009 to 14 August 2009. Depending on the wind direction, the monitor was set downwind of the pit, that is, if the wind was blowing from the north east, the monitor was set on the south west direction of the pit. Due to the unstable nature of the wind, the monitor was moved from one monitoring point to another twice a day depending on how the wind direction changed. The sampling time ranged from 8 – 16 hours a day depending on the battery life used for the monitor. The coordinates of the monitoring points were given in UTM and are shown in the Table 3.1 below. Figure 3.1 shows the locations of these points.

Table 3.1: Coordinates of the monitoring points

Monitoring point	x	y	z
MP1	507019.9	7517331	561.13
MP2	508248.8	7515218	532.03
MP3	507612.1	7514688	523.26
MP4	503528.9	7513532	497.08
MP5	504446.8	7512585	480
MP6	504389.4	7512711	486.11

MP – Monitoring Point

Figure 3.1: Location of the monitoring points around the Rössing open pit

3.1.2 Data processing

The measured data were processed to remove data that were not recorded on the downwind direction of the pit. This data processing was done by selecting the data that were recorded when the wind direction was continuously blowing from one direction and the monitor was set downwind from the pit within 30 ° of that direction.

Rössing Mine operates 24 hours a day, with three shifts consisting of 8 hours. These shifts are the day shift (08H00 to 16H00), the afternoon shift (16H00 to 12 midnight) and the night shift (12 midnight to 08H00). The data recorded during hours of shift change, which are 08H00, 16H00 and 00H00, were not included in the data for modelling. The data collected an hour after blasting were also excluded since blasting was left out from the modelling process. Other data falling out of the above described conditions were discarded. The concentration data were noted at five minutes intervals (the PM_{10} monitor takes a reading after every 5 seconds) and were averaged to hourly values for the model runs.

3.2. Modelling methodology

Breeze AERMOD pro 7 was run using Trinity Consultants interface software. ADMS 4.2 was run using CERC interface software. Surfer 9 was used as the mapping and contouring software.

As outlined in the literature survey, the models need meteorological data, topographical data as well as source information including the geometry and emission rate. In addition they also need information about the receptor location and height.

3.2.1 Meteorological data

The meteorological data required for the model input files were obtained from the surface onsite weather station at the mine site known as Bill point. The Bill point weather station is located at 22º 28’.007 south longitude and 015º 02’.563 East latitude with an elevation of 567m above sea level (see figure 3.1). The meteorological data were recorded at five minutes intervals and average hourly values were computed for the model input files.

3.2.1.1 ADMS meteorological input data

A meteorological file .MET was used as an input file for the model. The following parameters were included:

- ✓ Julian day (e.g. Dec 31 =365 or 366)
- ✓ Local time (0-24)
- ✓ Wind speed (m/s)
- ✓ Wind angle (degree)

Cloud cover (min = 0 and max = 8) (the cloud cover data was obtained from the SAWS where it was generated from the Unified Model (UM). The Unified Model is a Numerical Weather Prediction software suite originally developed by the United Kingdom Met Office. Data are provided by observations from satellites, from the ground, from buoys at sea, radar, radiosonde weather balloons, wind profilers, commercial aircraft and a background field from previous model runs (Wikipedia, 2010).

3.2.1.2. AERMOD

AERMET requires meteorological data for the surface data, upper air data and data from an onsite weather station. There is no upper air monitoring station located in areas close to Rössing Mine, hence Unified Model (UM) data obtained from the SAWS were used. The upper air data set consisted of the following data: Atmospheric pressure in millibars; height above the ground level (m); dry bulb temperature (°C); wind direction (degrees from the north) and wind speed (m/s). The data were given at 7 pressure levels: 500, 550, 600, 650, 700, 750 and 800mb.

The onsite data consisted of single surface hourly data measured at Rössing onsite Davis weather station. The data included wind speed, wind direction, temperature, humidity, pressure and solar radiation. The meteorological data were processed using the Met Pre-Processor in order to get it in the correct format for model input files.

3.2.2. Topographical data

3.2.2.1 ADMS topographical data

ADMS can be run with or without the hill option selected. The hill option was selected in this study because the study site is located in an area with hills. A .ter file (.ter file is a pre-formatted file consisting of terrain data), containing terrain data consisting lines with N, X, Y, Z, was used as an input file (CERC, 2007).

Where:

N = is an incrementing counter for each line

X = x coordinate of the data point

Y = y coordinate of the data point

Z = z is the height of the terrain

The topographical terrain file has a grid of 20km x 20km with a resolution of 70m x 70m (4900 points) (see figure 3.2, pit indicated by the red arrow).

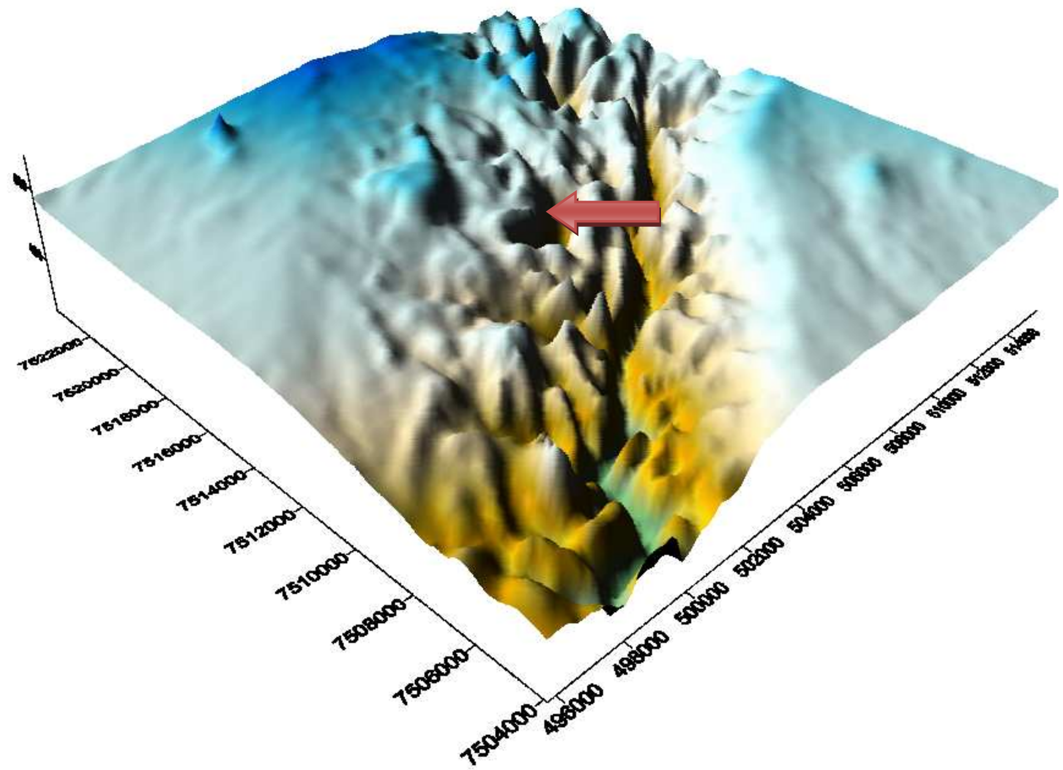


Figure 3.2: A 3D image of the pit generated from the topographical data used in the modelling files.

3.2.2.1 AERMOD topographical data

AERMOD requires terrain file in a form of DEM files as an input file. This DEM files used for running AERMOD were prepared with assistance from consultants from Airshed Planning Professionals (Pty) Ltd.

In an attempt to find out whether the performance of the models would improve, the models were also run with the option of simulating the in-pit sources as if they were located on the same altitude as the surface surrounding the pit (taken to be 480 m above sea level (ASL)) rather than at the bottom of the pit (see figure 3.3).

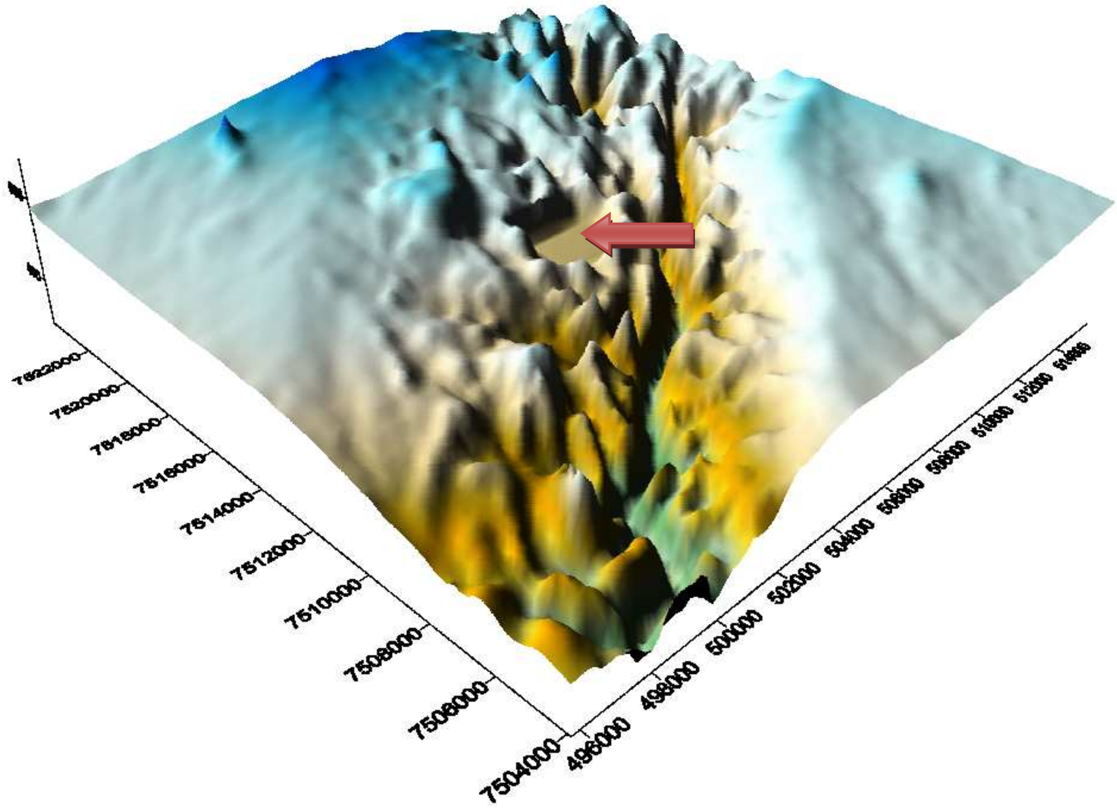


Figure 3.3: A 3D image of the pit with in-pit sources taken to be sitting at 480m ASL.

3.2.3 Source parameters and geometry

Table 3.2 and Table 3.3 give a summary of input parameters of the two models. The parameters differ according to the source type, that is, the input parameters for point source differ from those of volume source.

Table 3.2: Summary of input parameters of AERMOD

Source type	**Input parameters**
Point source	- Point emission rate in g/s - Release height above ground in meters - Stack gas exit temperature in degrees K - Stack gas exit velocity in m/s - Stack inside diameter
Volume source	- Volume emission rate in g/s - Release height above the ground, in meters - Initial lateral dimension of the volume in meters - Initial vertical dimension of the volume in meters
Area source	- Area emission rate in g/(s-m^2) - Release height above ground in meters - Length of X side of the area - Length of Y side of the area - Orientation angle for the rectangular area in degrees from North clockwise (optional) - Initial vertical dimension of the area source plume in meters
Area for polygon	- Area emission rate in g/(s-m^2) - Release height above ground in meters - Number of vertices of the area source polygon

	- Initial vertical dimension of the area source plume in meters

Source: USEPA, (2004)

Table 3.3: Summary of Input parameters of ADMS

Input parameters	Point	Area	Volume
Specific heat capacity of source material in J/ ° C/kg	✓	✓	×
Moral mass of the release material	✓	✓	×
Temperature or density of the release T (constant), RHO (density) and A (ambient)	✓	✓	×
Actual or NTP: emission parameter given at normal temperature and pressure (NTP) or at the actual release temperature and pressure	✓	✓	×
Efflux: exit velocity, volumetric flow rate or mass flow rate	✓	✓	×
Height (m): height above the ground. For volume source it is the mid-height of the volume above ground.	✓	✓	×
Diameter (m)	✓	×	×
Velocity (m/s)	✓	×	×
Volumetric flux (m^3/s) if actual was selected	✓	✓	×
Temp.(°C): temperature of the release	✓	✓	×

Xp (m), Yp(m): X and Y coordinates of the centre of point source in UTM	✓	×	×
L1 (m): width of a line source or vertical dimension of a volume source	×	×	✓
Mass flux (kg/s): mass flux of the emission if mass flux was selected	✓	✓	×
Emission rate: point (g/s), area (g/m^2/s), volume (g/m^3/s)	✓	✓	✓
Source geometry	X and Y coordinate	X and Y coordinates of 3-50[a] vertices of the sources	X and Y coordinates of 3-50[a] vertices of the sources

Source: CERC, 2007 [a] the polygon must be in convex shape

The concentrations of PM_{10} from overall emissions as a result of various sources at the pit were simulated for each modelling period. The input parameters of each source were entered into the models as outlined in the two tables above. The following sources were identified and were treated as specified sources as outlined in table 3.4 (refer to figure 2.6 in chapter 2 for the location of these sources on the pit). The dimensions of volume and area sources were computed from the Google earth image.

Table 3.4: Sources of PM_{10} at Rössing pit

Source	Source type
Loading of materials at the pit (Phase II, III & Trolley 10)	Volume source
Loading of materials at the ore (P) stockpiles (P1,P4, P2_200 & P2_100)	Volume source
Unloading of materials at P stockpiles (P2_100; P2&P3; P1 & P4)	Area source
Unloading of materials at low grade (LG) piles	Area source
Unloading of materials at waste dumps (Waste 2, 6 and 7)	Area source
Loading at the coarse ore stockpile	Point source
Roads	Area source

Loading of material at the pit and various stockpiles was treated as volume source and the dimensions were the length, width and height of the haul trucks as shown in Figure 3.4.

3.2.4. Source geometry and location

Table 3.5: Dimensions of the haul trucks

Truck model	L (m)	W (m)	H (m)	Volume (m^3)
Komatsu 465 (BR)	6.45	6.63	5.77	246.7
Caterpillar 785 (BR)	7.65	5.89	5.77	300.0
Komatsu 730E (RUL)	8.43	7.25	5.61	342.9

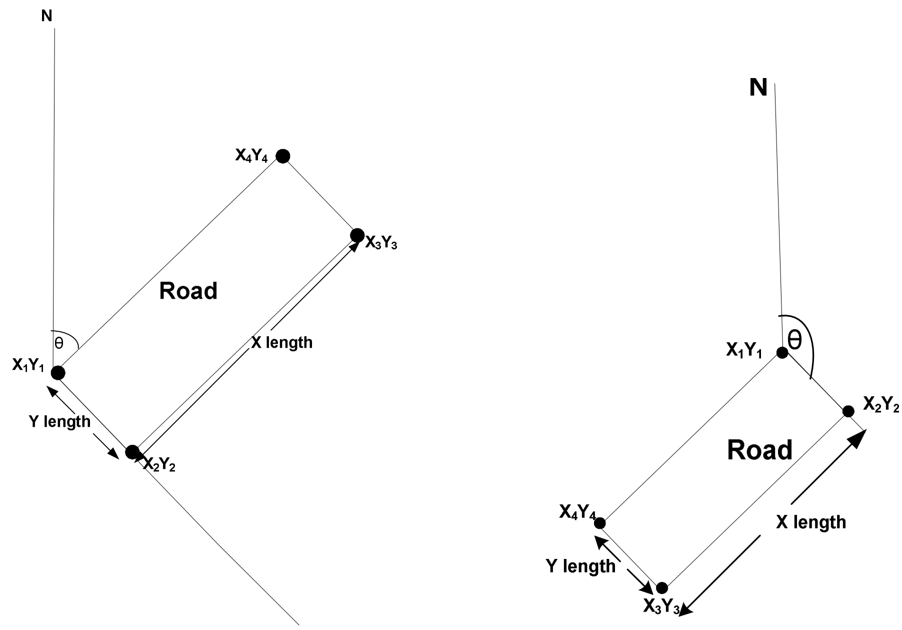
Figure 3.4: The dimensions of the haul truck

The roads were divided into a number of sections (Figure 3.5) in order to get the angle of each section from the true north (a parameter required by AERMOD). For each section, the length, width, angle and an area, were determined. Another reason why the roads were divided into sections is because the models do not model the curves, that is, all sources must be in convex shapes. All sections of the roads were assumed to be straight roads to facilitate the measurement of the angle from the north. Only roads where haul trucks travel were included in the modelling process.

Figure 3.5: Some of the roads around the Rössing open pit (scale: 1:2500)

The number of loads per road section was computed from the Mine Management Reporting System (MMRS) reports. The number of loads is used to calculate the total vehicle kilometre travelled (VKT) which is required for the calculation of the emission factors (see equation 2.11).

In case of AERMOD, the road requires the angle in degrees from the north and the X and Y coordinates of the south west (SW) corner of the source. Figure 3.6 shows the relationship of the area source parameters for the rotated rectangle. In case A, the Y length is equal to the width of the road and X length equals to the length of the road. In case B, the Y length is equal to the width of the road and X length is equal to the width of the road. The X and Y coordinates, that is, x_1y_1 in Figure 3.6, the X and Y lengths as well as the angle from the north (angle θ) for each area source, were determined. ADMS requires only the X and Y coordinates of the 3-50 vertices (corners) of each area source. For each area and volume source, the X and Y coordinates of four vertices (x_1y_1; x_2y_2; x_3y_3; x_4y_4 in figure 3.6) were input into the model.



Case A Case B

Figure 3.6 Sketches of roads as an example of area sources

3.2.4. Modelling grids and receptor locations

The modelling domain of 8km by 8km was selected. The domain included all sources and receptors. A regular Cartesian grid with 31 points on each direction was used for modelling. Specified receptor points, discrete receptors as known in AERMOD, which represented the monitoring points, were also input into the models in order to facilitate the simulation of concentrations at those points for the purpose of comparing them to the measured concentration (for the location of monitoring point i.e. specified points refer to table 3.1).

3.2.5 Emission inventory methodology

An emission inventory was compiled using the emission estimation equations from the USEPA emission factors website AP-42 and Australian NPI Emission Estimation Technique manuals as outlined in section 2.7 of the literature survey. Hourly emissions rates were calculated for each source for the duration of the study period.

3.2.5.1 Aggregate handling

Loading and unloading of material (ore or waste) was grouped under aggregate handling category and equation 2.9 was used to estimate emissions from this source category.

(d) Loading

(b) offloading

Figure 3.7: showing haul truck (a) loading at the pit and (b) offloading at the waste dump

As required for the emission estimation equations, the moisture content of the materials was obtained from laboratory analysis results provided by the Land Management section at Rossing Uranium Mine. The amount of material loaded at the pit, ore stockpiles and the amount of material unloaded from the trucks onto the waste dumps and/or stockpiles were obtained from Mine Management Reporting System (MMRS) reports from Rössing pit operations for the shifts during which ambient measurements were carried out.

3.2.5.2 Unpaved roads

There is a large network of unpaved roads at Rossing. In order to reduce the amount of dust generated and emitted from the roads, there are control systems in place. The haul roads (main ramps) are treated chemically using a chemical binder called Dust-a-Side (Figure 3.8b) (DAS). Dust on other sections of the roads is controlled by means of water spraying using water carts as explained in the literature survey. An emission efficiency of 50% and 80% was used for the roads treated with DAS and the roads sprayed by water respectively (NPI, 2001).

(a) (b)

Figure 3.8 Unpaved roads (a) with no Dust-a-Side (b) treated with Dust-a-Side

The two types of roads were treated as different sources because each road type has its own silt and moisture content. The silt and moisture content were also obtained from laboratory analysis results provided by the Land Management section at Rössing Uranium Mine. The weight of the haul trucks was obtained from the mine maintenance workshop.

Chapter 4: Results and Discussion

4.1 Results on emissions calculation

As discussed in the methodology chapter, an emission inventory was compiled for the different sources of PM_{10} at the Rössing Uranium Mine open pit. The results from the emission estimation are discussed in the following sections.

4.1.1 Emissions from material (ore and overburden) handling

Tables 4.1 to 4.2 show the emission rates from various activities during material handling. Activities emit PM_{10} at different rates depending on the magnitude of the activity as explained below.

Table 4.1: PM_{10} emissions as a result of material loading at the Rössing pit

Source	Emission rate (g/s)
PHII	6.4
Tr10	5.9
PHIII	2.8
P2_100	0.201
P4	0.125
P2_200	0.054
P1	0.026

Table 4.2: PM_{10} emissions as result of unloading material at the Rössing pit

Source	Emission rate (g/s)
W6	5.3
LG7	1.0
LG5	0.74
W7	0.28
P stockpile	0.26
P2	0.12
P3100_top	0.097
P4	0.028
P3100	0.018

The loading of material Phase 2 (PH II) generates the highest amount of PM_{10} recording the highest emission rate of 6.4g/s in the category of material aggregate handling (refer to Table 4.1). Tipping at waste dump 6 topped the group of material offloading with an emission rate of 5.3m/s, with P4 showing the lowest of 0.03g/s (refer to Table 4.2).

The differences in the emissions rates at various locations can be attributed to the difference in the amount of material loaded and offloaded at various loading and unloading points. The more the material loaded/unloaded the more dust is emitted as can be deduced from the units of the emission factors of material aggregate handling (kg/ton of material handled refer to equation 2.9). This is very apparent in this case since more material is loaded at trolley 10 than at other loading points. Similarly, more material is offloaded at waste 6 than at any other dump.

4.1.2 Emissions from unpaved roads

Unpaved roads associated with material movement in and from the pit as was discussed in the methodology chapter were classified into two categories and the PM_{10} emission rate from the two categories are given in Table 4.3.

Table 4.3: PM_{10} emissions from unpaved roads

Road type	Emission rate ($g/s/m^2$)
Dust-a-side roads	0.0000285
Non dust-a-side roads	0.000146

Unpaved road treated with Dust-a-Side (DAS) solution contributed less to the PM_{10} emissions in comparison with the sections of the roads which are not treated with DAS. The difference in emissions can be attributed to the fact that DAS has better palliative action than water that is applied on other sections of the road. Another reason could be the fact that there are few roads treated with DAS compared to roads not treated with DAS.

4.1.3 Overall emission rate at the pit

On average, during the measurement period the emission rate of PM_{10} from unpaved road was the highest at the pit for the duration of the study as compared to the emissions from material handling (see Table: 4.4).

Table: 4.4. The overall (on average) emission rate of PM_{10} at the pit

Source	Emission rate (g/s)
Unpaved road	7.6
Material handling	3.5

4.2. Summary of results for the meteorological data

The predominant wind directions recorded during study period were the westerly and west-south-westerly (refer to figure 4.1). However, during the morning hours, the north – easterly and east-north-easterly wind directions predominated. The wind speed recorded during the study ranged from as low as 0.07m/s (mostly experienced during night hours) to 6.98m/s (25.13km/hr). Around 31% of hours included in the modelling process recorded wind speeds below 1m/s. The ambient temperature experienced was from 9.66 °C to 30.95 °C.

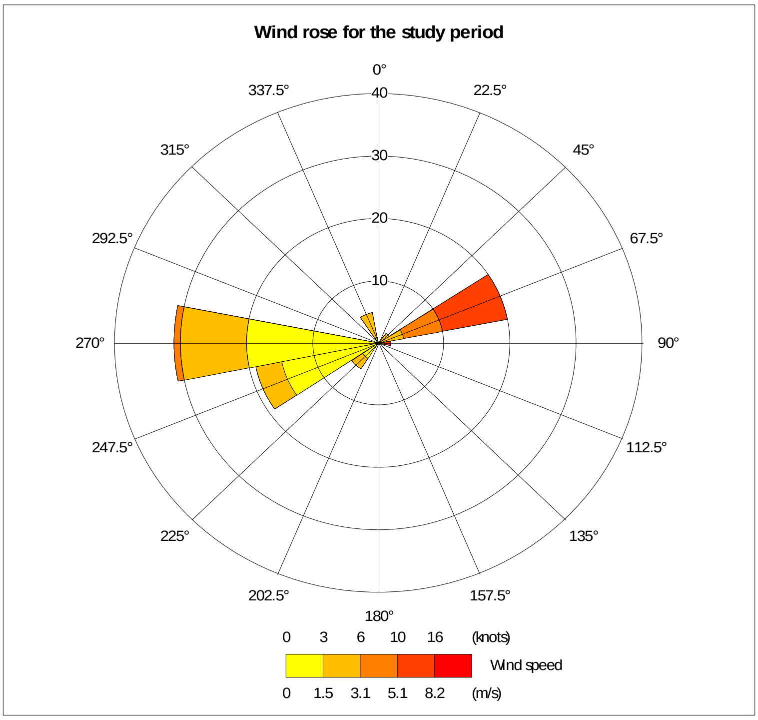


Figure 4.1: A wind-rose showing the summary of meteorological data

4.3. Evaluation of ADMS and AERMOD for the dispersion of PM_{10} using field data from Rössing Uranium Mine

4.3.1. Model performance measures

There is a number of performance measures used for the evaluation of dispersion models. These include the mean, standard deviation, fractional bias (FB), geometric mean bias (MB), Index of Agreement (IOA), coefficient of correlation (r) and normalized mean square error (NMSE) (Singh *et al.*, 2006; Kumar *et al.*, 2006). The performance of the model is evaluated by comparing the mean, standard deviation or any other performance measures of the observations to that of the predicted values. In this study, the model evaluation was done using the following statistical approaches: the mean, standard deviation, NMSE, IOA, and Max_R (ratio of predicted to observed concentration). In addition, Quantile-Quantile plots (Q-Q plots) were also used to evaluate the performance of the models.

NMSE is a measure of the overall deviation between the observed and predicted values, smaller values of NMSE reveal that the model is performing well both in time and space (Kumar *et al.*, 2006; Hirtl and Baumann-Stanzer, 2007). The expression for NMSE is given by:

$$\text{NMSE} = \frac{\overline{\left(C_o - C_p\right)^2}}{\overline{C_p} \times \overline{C_o}} \qquad (4.16)$$

The IOA is a measure of the skill of the model in predicting variations about the observed mean; a value above 0.5 is considered to be good (Zawar-Reza *et al.*, 2005). The expression for IOA is given by:

$$\text{IOA} = 1 - \frac{\sum_{i=1}^{N}\left(\overline{Cp - C_0}\right)^2}{\sum_{i=1}^{N}\left(\left|Cp - \overline{C_0}\right| + \left|C_0 - \overline{C_0}\right|\right)^2} \qquad (4.17)$$

The difference between NMSE and IOA is that NMSE is a statistical performance measure that gives information on the actual value of the error produced by the model (Sandu *et al.*, 2005); while IOA measures the agreement between the measured and observed values.

Q-Q plots are cumulative frequency distributions that provide a graphical characterization of the distribution of observed and modelled values over their entire ranges (Danish, 2006). These plots determine if the two sets of data come from populations with a common distribution. A model with a slope similar to that of the 1:1 line and with values close to the 1:1 line indicates a good fit between the simulated results and observed data (Zou *et al.*, 2010). A solid line has been added to the Q-Q plots to indicate an unbiased prediction and two dotted lines have been added to indicate a factor of two under- and overprediction (Paine *et al.*, 1998).

4.3.2 Models performance analysis

The performance of the models was evaluated in four different cases as outlined below.

Case 1 – The models were evaluated with the all the sources sitting at their actual elevation.

Case 2 – The performance evaluation was made when the models were run with the in-pit sources treated as if they were located on surface (taken to be 480 m ASL) rather than at the bottom.This approach was tested to see if the models perform better when all the sources are at the surface (as it was set at 480m ASL) or perform better when some sources are sitting at the bottom of the pit (390m deep pit) while others are at the surface (all sources sitting at their actual elevations).

Case 3 – Everything was the same as in case 1 except that data observed and predicted during periods when wind speed was below 1m/s (in case of AERMOD) and below 0.75m/s (in case of ADMS) were removed because the models do not give accurate or reliable results below those wind speeds due to calm conditions: 1m/s for AERMOD and 0.75m/s for ADMS.

Concentrations simulated by AERMOD may increase unrealistically to large values when wind speeds less than 1m/s are input to the model (USEPA, 2005). As a result the model was tested without the data recorded when wind speed was below 1m/s, to determine whether the performance of the model would improve. ADMS on the other hand automatically skips the hours with an average wind speed below 0.75m/s. although ADMS can be run using a calm condition option, whereby the wind speed can be lower than 0.75 m/s, this does not apply when the model is run with the hill option as it was the case in this study.

Case 4 – Everything was the same as in case 2 except that data observed and predicted during periods when wind speed was below 1m/s (in case of AERMOD) and below 0.75m/s (in case of ADMS) were removed.
However, AERMOD was further evaluated in two more cases using an open pit source type for sources inside the pit.

Case 5 – An open pit as a source type was used for sources inside the pit, with the pit at its normal elevation and all data were used.
Case 6 – same as in case 5 except that data observed and predicted during periods when wind speed was below 1m/s (in case of AERMOD) were removed.

ADMS was not evaluated in these last two cases since it does not have this option.

4.3.3. AERMOD model evaluation results

Calm conditions were experienced during the study especially during late afternoon and evening hours. This was due to very low wind speeds (wind speeds as low as 0.067m/s) which were experienced during these hours (refer to wind rose in Figure 4.1). Stable conditions prevailed during these hours as it was shown by the positive Monin-Obukhov length on the AERMET output data file.

AERMOD has a shortfall when it comes to calm conditions. Concentrations simulated by AERMOD may increase unrealistically to large values when wind speeds less than 1m/s are input to the model (USEPA, 2005). This can be deduced from the steady state Gaussian equation (which is used in the simulation of concentration values during stable conditions) where the concentration is inversely proportional to the wind speed (the lower the wind speed the higher the concentration; refer to equation 2.4 in Chapter 2).

In such cases where calm conditions prevail for extended periods, the hourly concentrations calculated with steady-state Gaussian models should not be considered valid. EPA (2005) recommended that these hours should be discarded and considered to be missing.

In the present study, the model performance was first evaluated with the data recorded when wind speed was less than 1m/s. High concentrations of over 1000 $\mu g/m^3$ of PM_{10} were simulated by AERMOD during these conditions. The US EPA recommendations regarding these conditions were then followed and observations for all the hours with wind speeds less that 1m/s were not used in the evaluation of performance.

The model performance statistical measures of all the six cases are shown and discussed below.

Table 4.5: AERMOD model performance statistics for case 1

	Mean	STDEV	NMSE	IOA	MAXr
Observed	4.42	4.94	0.00	1.00	1.00
Predicted	590.2	881.6	131.5	0.00207	175.4

Table 4.6: AERMOD model performance statistics for case 2

	Mean	STDEV	NMSE	IOA	MAX_r
Observed	4.42	4.94	0.00	1.00	1.00
Predicted	834.9	1105.9	187.8	0.0022	190.2

As can be seen from the statistical measures showed in the Tables 1 and 2 above, AERMOD performed very poorly. The mean, standard deviation, NMSE were all highly over predicted. The agreement between the observed and predicted values is extremely poor as it is evident from the index of agreement depicting the model poor prediction power in this case study.

When the pit was set to a flat plane of 480m, the performance of the model deteriorated further with a standard deviation and NMSE reaching high values of 1105.9 and 187.8 respectively. However, the IOA value increased slightly from 0.0021 to 0.0022.

Table 4.7: AERMOD model performance statistics for case 3

	Mean	STDEV	NMSE	IOA	MAXr
Observed	5.17	5.58	0.00	1.00	1.00
Predicted	272.4	404.7	50.69	0.00402	69.6

Table 4.8: AERMOD model performance statistics for case 4

	Mean	STDEV	NMSE	IOA	MAXr
Observed	5.17	5.58	0.00	1.00	1.00
Predicted	516.2	737.4	97.82	0.0031	107.0

When the concentration values recorded at wind speed lower than 1m/s were removed, the model performance improved as compared to the first two cases, although the model still performed very poorly as can be seen from the statistical measures in Table 3 and 4 above.

Table 4.9: AERMOD model performance statistics for case 5

	Mean	STDEV	NMSE	IOA	MAXr
Observed	4.42	4.94	0.00	1.00	1.00
Predicted	136.2	204.0	23.39	0.0033	31.26

Table 4.10: AERMOD model performance statistics for case 6

	Mean	STDEV	NMSE	IOA	MAXr
Observed	5.17	5.58	0.00	1.00	1.00
Predicted	76.7	112.5	12.89	0.0036	18.1

AERMOD has an option of using an open pit as a source type. When this option was used, the model performance improved. In case 6 where all the data were used, the mean dropped from 590.2 in case 1 to 136.2 and the IOA increased from 0.00207 to 0.0033. The IOA value decreased when the data observed at wind speed below 1 m/s were removed. However, other statistical measures which were very high, like the NMSE and MAX ratio, reduced, although not to acceptable values.

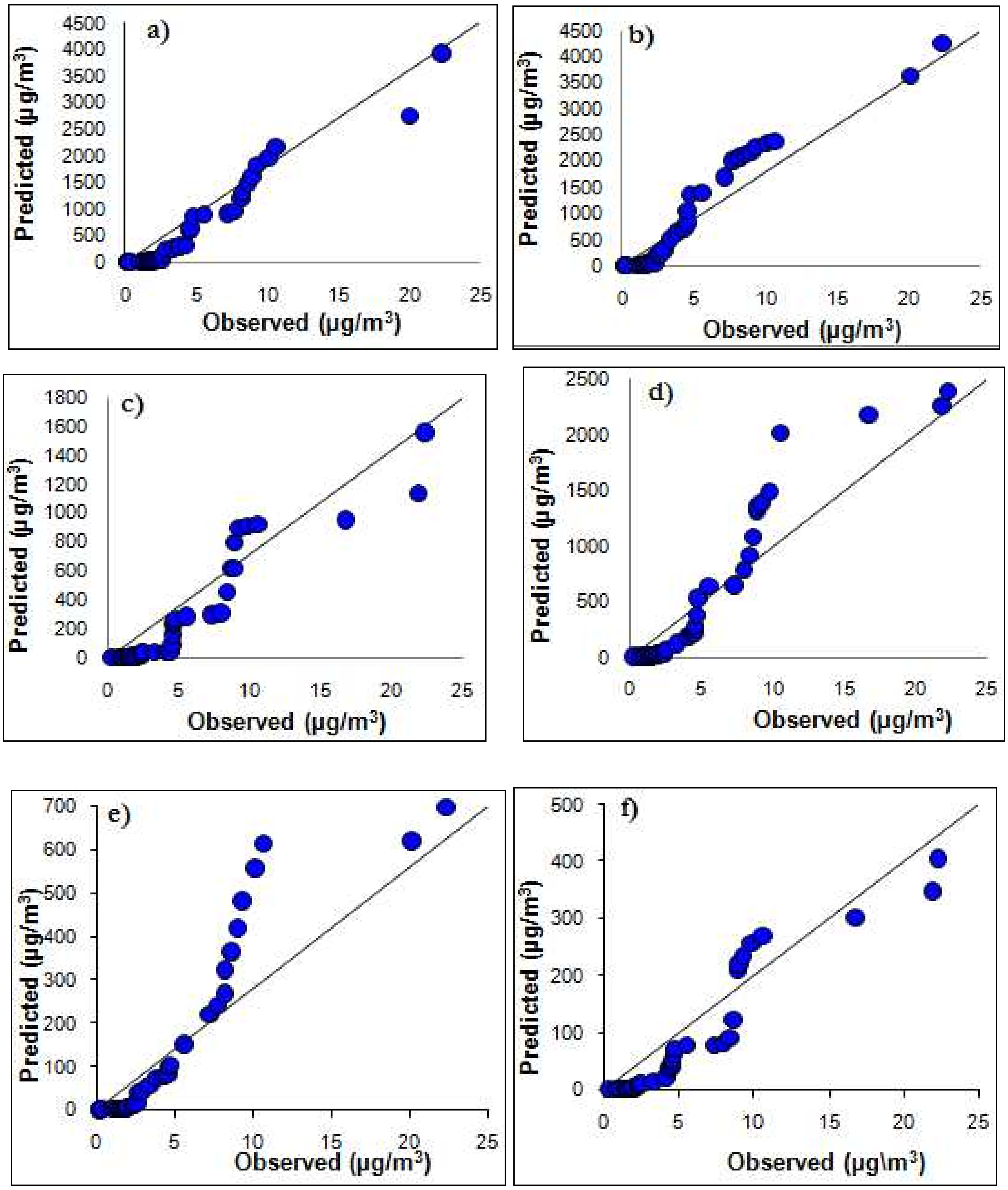


Figure 4.2 Q–Q plots of AERMOD predicted hourly concentration vs. observed hourly concentrations µg/m^3 for (a) for case 1 (b) for case 2 (c) for case 3 (d) for case 4 (e) for case 5 and (f) for case 6

The QQ plots in (a) and (b) surprisingly show that the two datasets are from the same statistical distribution, although there is a huge difference in the predicted and observed data. Figure (c) shows that the two datasets are not from the same statistical distribution, with under prediction visible with low numbers and an over prediction with large values. Figures (d) and (e) show an over prediction which is in agreement with other statistical measures. The model under predicted the concentrations levels at low values as can be seen in all the plots. After the data simulated when the values recorded below 1m/s were removed (Figure f), the distribution of the data slightly improved. However, the model results showed a similar trend to other cases, where at low values the model under predicted while at large values an over prediction was evident.

4.3.4 ADMS model evaluation results

The performance of ADMS was evaluated from case 1 to 4. However, ADMS skips all the meteorological line or hours with wind speeds below 0.75 m/s (as calm conditions). That means the model does not simulate any concentration below that threshold wind speed limit. As a result, the model results in cases 1 and 2 are identical to the results in cases 3 and 4. Hence only the results from cases 1 and 2 are presented here.

Table 4.11: ADMS model performance statistics for case 1

	Mean	STDEV	NMSE	IOA	MAXr
Observed	7.51	9.42	0.00	1.00	1.00
Predicted	5.25	10.84	0.184	0.42	1.16

The model results show an under prediction as it is evident from the QQ plot as well as by the mean values. The NMSE is relatively low showing a small error produced by the model. The performance of the model seemed to be acceptable if the NMSE value is less than 0.5. The model performed well in predicting the high end of the concentration distribution as shown by MAX ratio value close to one. Overall agreement between predicted and observed values is somewhat lower than acceptable as indicated by the IOA of 0.42.

Table 4.12: ADMS model performance statistics for case 2

	Mean	STDEV	NMSE	IOA	MAXr
Observed	7.51	9.42	0.00	1.00	1.00
Predicted	3.73	7.49	0.203	0.48	0.861

In case 2, when the elevation of the pit was set to a flat plane of 480m, the model performance was slightly improved in terms of the agreement between the predicted and the observed values as it is evident from the change in the IOA value from 0.42 to 0.48. However, the error produced by the model increased slightly as shown by the change in the NMSE values from 0.184 to 0.203, but it was still within acceptable values. The high end of the concentration distribution was slightly under predicted with a value lower than one, although the performance is still good as it is evident from the MAX ratio below one but close to one.

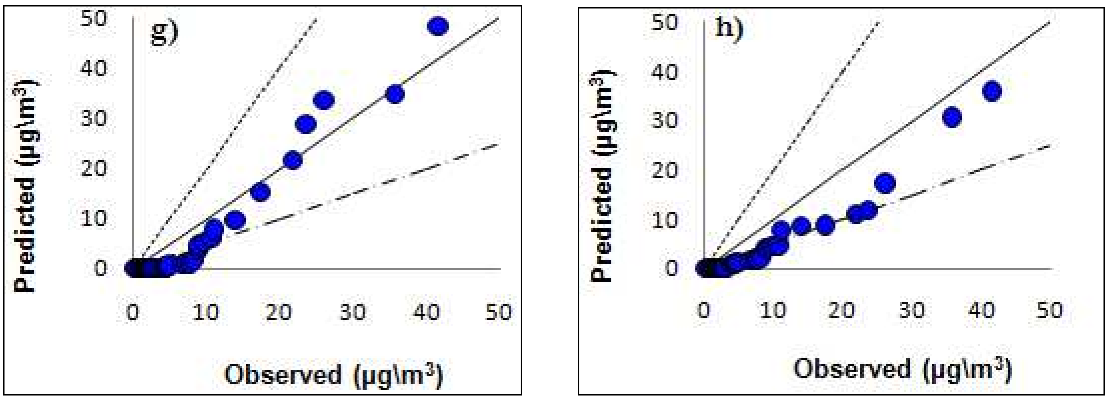


Figure 4.3: Q–Q plots of ADMS predicted hourly concentration vs. observed hourly concentrations μg/m^3 for (g) for case 1 (h) for case 2

Figure 4.3 (g) shows the QQ plot of the model predictions versus the observations for case 1. The graph shows an under prediction at low values. However, the predictions improved with high values on the concentration distribution. Figure 4.3

(h) shows the QQ plot for the model prediction versus observations for case 2. The graph shows an under prediction through the concentration distribution.

4.4 Discussion

The difference between model predictions and observations can be due to the fact that the model cannot include all the variables that affect the observation at a particular time and location (Perry *et al.* 2004). Uncertainties in meteorological data can also cause predicted values to deviate from the observations. The experience of model developers proves that the uncertainty caused by wind direction alone can cause disappointing results from what is viewed as a well performing dispersion model (Paine *et al.*, 1998).

The uncertainties brought in by instrument errors like weather stations can be another factor that can cause the deviations of predicted results from observations (D'Abreton, 2009).

Model underestimation may be possibly due to fact that no background PM_{10} concentration levels were used in the model during the simulation. The results from a study conducted by Kasarkar *et al.* (2005) revealed the same experience when the simulation of PM_{10} with AERMOD over Pune (in India) was done with the absence of background levels. Similarly, the results from the validation study of ADMS with complex terrain at Lovett Power plant showed large numbers of points for which the modelled values were zero and the observations values were non-zero and the same reasoning of the absence of background levels was discussed (CERC, 2007).

The differences in the performance of the two models can be attributed to the fact that ADMS and AERMOD use different algorithms in their predictions of pollutant concentrations. The general form of the expressions for the concentration in AERMOD for both CBL and SBL can be written as follows:

$$C\{x, y, z\} = \left(\frac{Q}{\tilde{u}}\right) P_y\{y; x\} P_z\{z; x\},$$

Where Q is the source emission rate, $\tilde{u}$ is the effective wind speed, and P_y and P_z are probability density functions (pdf) which describe the lateral and vertical concentration distributions, respectively.

As can be deduced from the equation used by AERMOD; the concentration increases as the wind speed decreases. As a result the high concentration simulated by AERMOD can be attributed to the low wind speeds experienced during the study.

Chapter 5: Conclusions and Recommendations

5.1 Conclusions

The study evaluated the performance of ADMS and AERMOD in the prediction of the dispersion of PM_{10} from Rössing Uranium Mine open pit. The performance of the two models was evaluated against the observations and also against each other using various statistical measures.

The study showed that the performance of ADMS was superior to that of AERMOD. AERMOD performed poorly during calm conditions, (wind speed was less than 1m/s). When observations under calm conditions were not taken into account, the performance of the model improved, although not to acceptable values.

An attempt to obtain improvement by setting all pit sources in a flat plane at the elevation of the rim of the pit did not yield materially improved results, although the index of agreement improved slightly. In general, the performance of AERMOD was very poor and simulated extremely high concentration values. This led to the conclusion that AERMOD is not a suitable model to use when prolonged calm conditions occur frequently.

ADMS performance was superior over AERMOD as was evident from the values of various performance statistical measures and a conclusion reached was that ADMS is likely to be a better model to use in cases where prolonged calm conditions are experienced.

5.2 Recommendations

Further studies must set up an upper air weather station close to the study area in order to take measurements of the actual weather parameters instead of using simulated meteorological data as was done in this study due to the absence of an upper air weather station close to the study area.

An improvement must be made to the ADMS algorithms to enable the use of .aai files when the hill option is selected since skipping the meteorological hours with calm conditions may affect the overall performance of the model. An .aai file is a file that is used when you want to use model options not available in the interface, e.g. when modeling calm conditions.

The AERMOD model algorithm should be reviewed to improve the model performance during prolonged stagnant conditions like calm conditions, as the results from the study showed that AERMOD performs very poorly during these conditions.

Further studies should take the background concentration into account since, due to lack of equipments, this was not feasible in this study.

There is limited knowledge on the wind patterns and how the plume behaves during calm conditions at present. Therefore, further studies on plume behavior and wind flow patterns during calm conditions are recommended.

Chapter 6: Reference

Aipanda, TK (2010) *Reconciliation of cycle times at Rössing Uranium mine*, BSc dissertation, Department of Mining Engineering, University of Pretoria, Pretoria, South Africa.

AMS, (2010) "Glossary of Meteorology" http://www.amsglossary.allenpress.com/glossary [2010 August 08].

Basson, IJ and Greenway, G (2004) "The Rossing Uranium Deposit: a product of late-kinematic localization of uraniferous granites in the Central Zone of the Damara Orogen, Namibia" *African Earth Sciences*, 38, 413-435, http://www.portergeo.com.au/database/mineinfo [2009 October 8].

Brode, RW (2006) "AERMOD Technical Forum", presented at *The EPA R\S\L Modellers workshop,* 16 May, 2006, San Diego, California.

Brooke, D, Stiff, S and Webb, A (2007) "A comparison of results from ADMS and AERMOD with measured data", paper presented at *The eleventh conference on Harmonization within Atmospheric Dispersion Modelling for Regulatory Purposes*, 2-5 July, 2007, Cambridge, UK.

Carruthers, DJ, Holroyd, RJ, Hunt, JCR, Weng, WS, Robins, AG, Thomson, DJ and Smith, FB (1994) "UK ADMS, A New Approach to Modelling Dispersion in The Earth's Atmospheric Boundary Layer" *Wind Engineering and Industrial Aerodynamics,* 52, 139–153.

CERC, (2001) "ADMS 3.1 User Guide" http://www.cerc.com [2009 October 03].

CERC, (2005) "ADMS 3 Validation summary" http://www.cerc.com/software [2009 October 7].

CERC, (2007) “Atmospheric Dispersion Modelling System (ADMS 4) User Guide version 4.0” http://www.cerc.com/softwares [2009 October 3].

Chakraborty, MK, Ahmad, M, Singh, MRS, Pal, D, Bandopadhyay, C and Chaulya, SK (2002) “Determination of the emission rate from various opencast mining operations” *Environmental Modelling and Software,* 17 (2002), 467–480.

Chaulya, SK, Ahmad, M, Singh, MRS, Pal, D, Bandopadhyay LK, Bandopadhyay, C and Mondal, GC (2002) “Validation of two air quality models for Indian mining condition” *Environmental Monitoring and Assessment,* 82(1), 23 – 43.

Cimorelli, AJ, Perry SG, Venkatran A, Weil JC, Paine RJ, Wilson RB, Lee RF, Peters RW, Brode, RW and Pavimer,JO (2004) “AERMOD: Description of model formulation version 02222” U.S. Environmental Protection Agency, Research Triangle Park, North Carolina.

Cooper, CD and Alley, FC (2002) *Air pollution control: the design approach,* 3rd edition. Waveland Press, Inc, United State of America.

Colls, J (2002) *Air Pollution,* 2nd edition, Taylor & Francis, New York.

Cora, MG and Hung, YT (2003) “Air Dispersion Modelling: A Tool for Environmental Evaluation and Improvement” *Environmental Quality Management*, www.interscience.wiley.com [2010 February 2].

Cox, W and Tikvart, J (1990) “A statistical procedure for determining the best performing air quality simulation model” *Atmospheric Environment* 24 A 2387-2395.

CSIR (1991) "An Environmental Impact Statement for the Rössing Uranium Mine, Namibia", Technical Report to Rössing Uranium Limited, Air Quality Information Systems, Division of Water Technology, CSIR, Pretoria.

D'Abreton, P (2009) "Air Quality Modelling Best Practice Guidance for the Australian Alumina Industry" Queensland Environment consultants report prepared for Australian Aluminium Council, Australia, http:// www.paeholmes.com [2009 July 6].

Danish, F (2006) *Examination of the performance of AERMOD model under different wind conditions*, MSc Thesis, Department of Civil engineering, University of Toledo, USA.

Demael, E and Carissimo, B (2007) "Local atmospheric dispersion modelling of pollutants issued from a nuclear power plant: a comparison using a CFD code and ADMS with wind tunnel data", paper presented at ", Paper presented at *The eleventh conference on Harmonization within Atmospheric Dispersion Modelling for Regulatory Purposes",* 2-5 July, 2007, Cambridge, UK.

Dunkerley, F, Spanton, AM, Hall, DJ, Bennett, M and Griffiths, RF (2001) "An Intercomparison of the AERMOD, ADMS and ISC Dispersion Models for Regulatory Applications: Dispersion over Terrain", Paper presented at *The 7th International Conference on Harmonization within Atmospheric Dispersion Modelling for Regulatory Purposes,* 28-31 May, 2001, Belgirate, Italy.

El-Harbawi, M, Mustapha, S and Rashid, ZA (2009) "Air Pollution Modelling, Simulation and Computational Methods: A Review" Paper presented at the *ICERT 2008: International Conference on Environmental Research and Technology"* Penang, Malaysia.

Farmer, AM (1993) "The effects of dust on vegetation: A review", *Environmental Pollution,* 79 (1993), 63-75.

Fernandes, HM, Veiga, LHS, Franklin, MR, Prado, VCS and Taddei, JF (1995) "Environmental impact assessment of uranium mining and milling facilities: A study case at the Pocos de Caldas uranium mining and milling site, Brazil", *Geochemical Exploration* 52 (1995) 161-173.

Fung, D (2005) "Particulate Matter (PM)" http://www.ph.ucla.edu/ehs [21 June 2010]

Grantz, DA, Garner, JHB and Johnson, DW (2003) "Ecological effects of particulate matter", *Environment International*, 29 (2003) 213– 239.

Hall, DJ, Spanton, AM, Dunkerley, F, Bennett, M and Griffiths, RF (2000a) "A Review of Dispersion Model Intercomparison Studies Using ISC, R91, AERMOD and ADMS" UK Environment Agency, R&D Technical Report No. P353.

Hall, DJ, Spanton, AM, Bennett, M, Dunkerley, FN, Griffiths, RF, Fisher, BEA and Timmis, RJ (2000) "An inter-comparison of the AERMOD, ADMS and ISC dispersion models for regulatory applications" UK Environment Agency, R&D Technical Report P362.

Hanna, SR, Egan, BA, Purdum, J and Wagler, J (1999) "Evaluation of the ADMS, AERMOD, and ISC3 dispersion models with the Optex, Duke Forest, Kincaid, Indianapolis and Lovett field data sets", Paper presented at *The Sixth International Conference on Harmonisation Within Atmospheric Dispersion Modelling for Regulatory Purposes*, October 11–14 1999, INSA de Rouen, France.

Harsham, KD and Bennett, M (2008), "A Sensitivity Study of the Validation of Three Regulatory Dispersion Models", *Environmental Sciences,* 4 (1), 63-76.

Hill, RA, Lowles, I, Teasdale, I, Chambers, N, Puxley, C and Parker, T (2001) "Comparison between field measurements of 85 – Kr around the BNFL Sellafield reprocessing plant and the predictions of the NRPBR – 91 and UK –ADMS", *Environment and pollution,* 16 (1-6), 315 – 327.

Hirtl, M, Baumann-Stanzer, K, Kaiser, A, Petz, E, and Rau, G (2007) "Evaluation of three dispersion models for the Trabovlje power plant, Slovenia", Paper presented at *The 11th International Conference on Harmonisation,* 2–5 July 2007, Cambridge, UK.

Holmes, NS and Morawska, L (2006) "A review of dispersion modelling and its application to the dispersion of particles: An overview of different dispersion models", *Atmospheric Environment,* 40, 5902 – 5928.

Hurley, PJ (2006) "An evaluation and inter-comparison of AUSPLUME, AERMOD and TAPM for seven field datasets of point source dispersion", *Clean air and Environmental Quality,* 40, 45-50.

Ihuhua, RT (2009) "5 Year water consumption forecast", Rössing Uranium Limited, Arandis, Namibia.

IMA-Europe (2009) "Occupational Exposure Limits in mg/m^3 – Respirable dust In EU 271 plus Norway & Switzerland", http://www.imaeurope.eu/ otherPublications.html [21 June 2010].

Inyang, HI and Bae, S (2006) "Impacts of dust on environmental systems and human health" *Hazardous Materials,* 132 (2006), v–vi.

Jacobson, MZ (2002), *Atmospheric pollution history, science and regulation,* Press Syndicate of the University of Cambridge, Cambridge, UK.

Kadhila-Amoomo, A *(*2008) “Ambient air quality management plan”, Rössing Uranium Limited, Arandis, Namibia.

Kanevce, G and Kanevce, L (2006) “Dispersion modelling for regulatory applications” *Thermal Science*, 10 (2006), 141-154.

Kesarkar, AP, Dalvi, M, Kaginalkar, A and Ojha, A (2007) “Coupling of the Weather Research and Forecasting Model with AERMOD for pollutant dispersion modelling, a case study for PM10 dispersion over Pune, India”, *Atmospheric Environment* 41 (2007) 1976–1988.

Kotze, W (1999) *Haul Roads Dust Suppression*, BSc dissertation, Department of Mining Engineering, University of Pretoria, Pretoria, South Africa.

Kumar, A, Dixit, S, Varadarajan, C, Vijayan, A and Masuraha, A (2006) “Evaluation of the AERMOD Dispersion Model as a Function of Atmospheric Stability for an Urban Area” *Environmental Progress*, 25 (2) no.2, 141-151.

Leggatt, H (2009) “Rössing Uranium light in gloom”, http://www.rossing.com/files [2009, October 23].

Li, Y (2009) *Evaluation of AERMOD and CALPUFF air dispersion models for livestock odour dispersion simulation,* MSc Thesis, University of Saskatchewan, Saskatoon, Saskatchewan.

McHugh, CA, Higson, DJ and Dyster, SJ (1999) “Comparison of model evaluation methodologies with application to ADMS 3 and US model” Cambridge Environmental Research Consultants, 3 Kings Parade, Cambridge, UK.

Moeller WK, (2001) *Dust Palliation Analysis Rossing Uranium,* BSc dissertation, Department of Mining Engineering, University of Pretoria, Pretoria.

Ninham, Shand (2008) “Social and Environmental Impact Assessment: Proposed Expansion Project for Rössing Uranium Mine in Namibia: Phase 2: Extension of current SJ open pit mining activity, new mining activity in SK area, increased waste rock disposal capacity, increased tailings disposal capacity, establishment of acid heap leaching facility and sulphur handling in the Port of Walvis Bay”, Final Scoping Report, *Report No. 4626/402239* for Rossing Uranium Limited.

NPI (2001) “Emission Estimation Technique Manual: Mining”, Report for Environment Australia, http://www.npi.gov.au, [2009 August 20].

NSW Health, (2006) “Mine dust and you”, www.health.nsw.gov.au [2010 March 5].

OSHA (2008) “Dust Control Systems”, http://www.osha.gov/SLTC, [2009 July 7]

Perry, SG, Cimorelli, AJ, Paine, RJ, Brode, RW, Weil, JC, Venkatram, A, Wilson, RB, Lee, RF and Peters, WD (2004) “AERMOD: A Dispersion Model for Industrial Source Applications. Part II: Model Performance against 17 Field Study Databases” *Applied Meteorology*, 44, 695-707.

Petavratzi, E, Kingman, S and Lowndes, I (2005) “Particulates from mining operations: A review of sources effects and regulations”, *Mineral engineering* 18 (2005) 1183 – 1199.

Pitts, O (2006) “Improvement of NPI fugitive particulate matter emission estimation Techniques” Sinclair Knight Merz report to the WA Department of Environment, Perth, Australia.

Reed, WR (2005), “Significant Dust Dispersion Models for Mining Operations”, Report for National Institute for Occupational Safety and Health, Research Laboratory, Pittsburgh, IC 9478 information circular/2005.

Riddle, A, Carruthers, D, Sharpe, A, McHugh, C and Stocker, J (2004), “Comparisons between FLUENT and ADMS for atmospheric dispersion modelling” *Atmospheric Environment* 38 (2004) (7), 1029–1038.

Rio Tinto (2003) "Occupational health standards" http://www.chamberofmines.org.na [2011 April 27].

Roose, E (1996) "Land husbandry - Components and strategy", FAO, Montpellier, France, http://www.fao.org/docrep [2011 March 20].

RUL (2006) "Rössing Uranium in 2006", http://www.rossing-com.info, [2009 March 2].

RUL (2008) "Report to Stakeholders Rössing Uranium Limited: Remaining on a path of growth", http://www.rossing-com.info/reports/stake_report, [2009 March 3].

RUL (2009) "Report to Stakeholders Rössing Uranium Limited: Investing in our future", http://www.rossing-com.info/reports/stake_report [2010 April 20].

Sandu, I, Ionescu, C and Ursache, M (2005) "Statistical performance of two dispersion models (OML and ADMS) for measurements obtained in a life pilot study- assessment system for urban environment(assure)", Paper presented at The *9th Int. Conf. on Harmonization within Atmospheric Dispersion Modelling for Regulatory Purposes*, 1 – 4 June 2004, Garmisch-Partenkirchen, Germany.

Shikongo, INN (2005) *Optimizing the hauling operation at Rössing*, BSc dissertation, Department of Mining Engineering, University of Pretoria, Pretoria, South Africa.

Sidle, C, Tuckett-Jones, B, Ng, B and Shi, JP (2002) "Model Intercomparison between ADMS 3.1 and AERMOD and AERMOD PRIME", paper presented at the *9th Int. Conf. on Harmonization within Atmospheric Dispersion Modelling for Regulatory Purposes*, 1 – 4 June 2004, Garmisch-Partenkirchen, Germany.

Silvester SA, Lowndes, IS and Hargreaves, DM (2009) “A computational study of particulate emissions from an open pit quarry under neutral atmospheric conditions”, *Atmospheric Environment* (2009) 1–10.

Singh, G, Prabha, J and Giri, S (2006) “Comparison and performance evaluation of dispersion models FDM and ISCST3 for a gold mine at Goa”, *Industrial pollution control,* 22(2), 297 – 303.

Sivacoumar, R, Raj, SM, Chinnadurai, SJ and Jayabalou, R (2009) “Modelling of fugitive dust emission and control measures in stone crushing industry”, *Environmental Monitoring,* 11, 987–997.

SKM, (2005) “Improvement of NPI Fugitive Particulate Matter Emission Estimation Techniques” SKM consultants report to WA Department of Environment, Australia.

Thomas, RG (2008) *An air quality baseline assessment for the Vaal Airshed in South Africa,* MSc Thesis, University of Pretoria, Pretoria, South Africa.

Trivedi, R, Chakraborty , MK and Kumar, S (2008) “Dust generation and its dispersion due to mining activities in Durgapur open cast coal project of W.C.L. – A Case Study”, *The Indian Mining and Engineering Journal,* 68, 24 – 31.

Tshukudu, T (2003) *Development of an air quality model for BCL Limited*, MSc Thesis, University of Pretoria, Pretoria, South Africa.

Turner, DB (1994) *Workbook of Atmospheric Dispersion Estimates: An Introduction to Dispersion Modeling* (2nd Edition), CRC Press, US.

US EPA (1998) “Compilation of Air Pollutant Emission Factors, AP-42”, Fourth Edition, Office of Air Quality Planning and Standards, Research Triangle Park, North Carolina, 27711.

US EPA (2006) “Compilation of Air Pollutant Emission Factors, AP-42”, http://www.epa.gov/ttnchie1/ap42/, [2009 July 8].

US EPA (2004) “User’s guide for the AMS/EPA regulatory model – AERMOD”, Office of Air Quality Planning and Standards, Emissions Monitoring and Analysis Division Research Triangle Park, North Carolina, 27711.

Vannucci, M, Colla, V and Haines, N (2008) “Air Dispersion Modelling for the Assessment of ULCOS Technologies” *Paper presented at the 4th* ULCOS *seminar,* 1-2 October 2008, France.

Venkatram, A (2008) “Introduction to AERMOD”, AERMOD fundamentals: Micrometeorology and dispersion notes, Department of Mechanical engineering, University of California www.engr.ucr.edu/~venky [2010 August 9].

Vora, J (2010) *Dust dispersion modelling for opencast mines*, BSc dissertation, National Institute of Technology Rourkela, India.

Walker, JI, Scaplen, M, and George, F (2002) “ISCST3, AERMOD and CALPUFF: A Comparative Analysis in the Environmental Assessment of a Sour Gas Plant” Jacques Whitford Environment Limited report paper. Paper No: 25.

Wikipedia (2010) “Atmospheric dispersion modelling”, http://en.wikipedia.org/wiki/Atmospheric_dispersion_modeling , [2010 February 10].

Wikipedia (2010) “Erosion”, http://en.wikipedia.org/wiki/Erosion , [2010 November 01].

Wikipedia (2010) “Particulates”, http://en.wikipedia.org/wiki/Particulates , [2010 March 20].

Zawar-Reza, P, Kingham, S and Pearce, J (2005) “Evaluation of a year-long dispersion modelling of PM using the mesoscale model TAPM for Christchurch, New Zealand” *Science of the Total Environment,* 349 (2005) 249– 259.

Zoras, S, Triantafyllou, AG and Deligiorgi, D (2006) “Atmospheric stability and PM10 concentrations at far distance from elevated point sources in complex terrain: Worst-case episode study" *Environmental Management* 80 (2006), 295–302.

Zou, B, Zhan, FB, Wilson, JG and Zeng, Y (2010) "Performance of AERMOD at different time scales" *Simulation Modelling Practice and Theory* 18 (2010) 612–623.

www.epa.gov.

www.cerc.com