# CHAPTER 2

# REVIEW OF LITERATURE

Mining industry is a necessary evil. It is necessary for meeting the material and energy needs of the society. It is evil as in the process of production of minerals, it generates huge quantity of environmental pollution for all the stake holders. Among various forms of environmental pollution, air pollution is very lethal for the community living nearby. Air pollution is the combined result of noxious gases and suspended particulate matter. The airborne particulate matter has been defined as $PM_{10}$, $PM_{2.5}$ etc. The size of the particulate matter is crucial as it determines their ability to affect the health of the lungs of human beings and causes diseases related to eye and skin. In addition to this, it also causes closure of stomata in plants and deposition of dust on the surface nearby. Hence we see that the flora and fauna of the areas surrounding the mines are influenced due to particulate matter.

Suspended particulate matter (SPM) poses world over, a serious concern for the mine environmentalists. Researchers () have significantly focussed on the factors affecting the genesis, type, size and impact of SPM on human health, flora and fauna and the quality of air. Many attempts () have also been made to quantify and develop analytical tools for estimating its various influencing parameters and the dispersal mechanisms. This chapter discusses the works of earlier researchers on these issues.

## 2.1 Surface Mining Activities

The unit operations of mining generate different types of air pollution. Table 2.1 shows the mining activities and the potential types of air pollution caused due to it.

**Table 2.1: Mining activities and the types of air pollution (CSE, 2009-2010)**

| **Mining Activities** | **Pollution** |
|---|---|
| Drilling | Dust pollution is the main concern |
| Blasting | Dust and gaseous pollutants like sulphur dioxide and oxides of nitrogen |
| Loading operation | Dust pollution is the main concern |
| Haul road | Dust pollution |
| Transportation | Dust and gaseous pollutants like sulphur dioxide and oxides of nitrogen |
| Crushing of ore | Dust pollution |
| Storage ore | Dust pollution |
| Solid waste handling and re-handling | Dust pollution |
| Tailing waste | Air pollution and water pollution |

Removal of the top soil and the reclamation of mined out area also generates large quantity of dust that is likely to be dispersed in the atmosphere. Generally the top soil consists of clay or silt material which is removed using scrapper, bulldozers, rippers and trucks. It is to be stored away from the mine site for use during reclamation. In current surface mining practice, reclamation is almost an ongoing process which runs simultaneously with the mining itself.

Central Mining Research Institute (CMRI, 1998), in a research for Indian surface mine, had concluded that 80.2% of the total dust emission in mining industry was coming from the transport road of the surface mines. Screening plant was the next source of the dust emission which contributed 8.1% of the dust while 2.8% was released from the overburden removal. Top soil handling released 2.6% of the total dust followed by the coal extraction (2.2%), drilling and blasting (1.3%), coal handling or stockpile (1.1%) of dust and 1.7% was from miscellaneous work. The percentage contribution of different mining activities in an Indian opencast coal mine, are summarised in Figure 2.1:

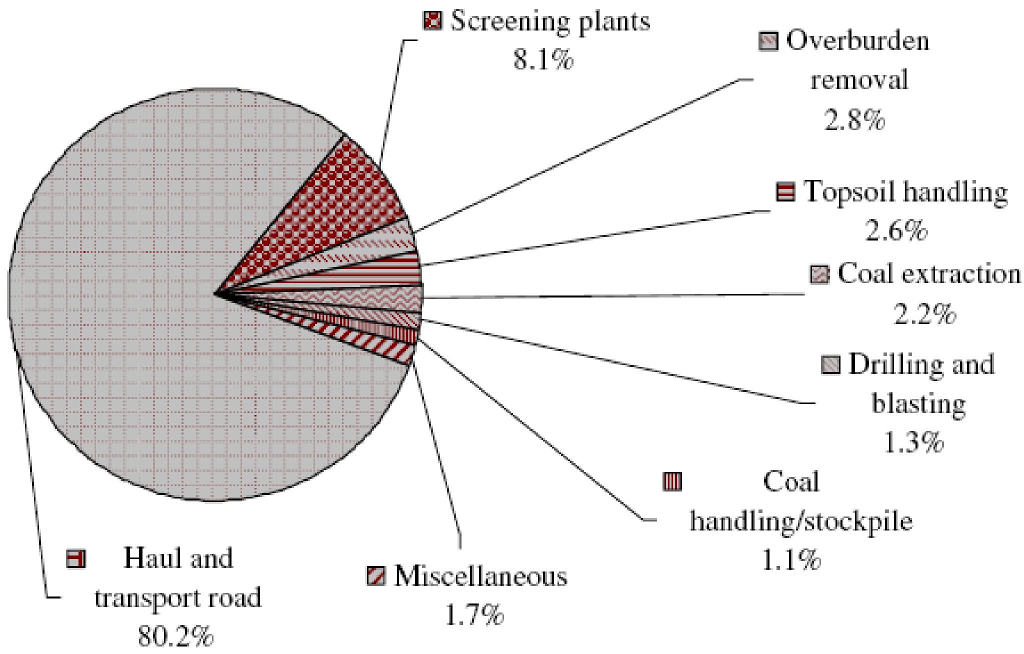


**Figure 2.1: Percentage contributions of different mining activities to total dust emissions from a typical Indian opencast coal mine (Mandal et al., 2012)**

A brief description of the above stated activities along with the contribution to the air pollution is given below.

## 2.1.1 Drilling

Large quantities of the fine dust were produced during drilling operations. These particles remain suspended in the air for long periods and get dispersed away from the source. The quantity of dust generated during the drilling operation is influenced by the factors such as material type, heat treatment condition for dill steel and rock material, temperature, and the associated chip formation mode (Balout et al., 2007). Ductile materials are likely to produce more dust. Hence, the plastic deformation during drilling generates a microscopic friction producing very fine dust particulates of the order of 2.5 microns (Balout et al., 2003). Stanton et al. on the basis of the South African Mining industry state that the dust generated from the drilling of overburden was the major source of dust (Stanton et al., 2006). Nair et al. (1999) reported that per meter drilling of a 250 mm. diameter drill hole of an iron ore opencast mine produced 1.46 kg of the respirable suspended particulate matter (RSPM) in the atmosphere. They also reported that though the drilling was the shortest duration mining operation, it produced the maximum quantity of RSPM. Table 2.2 presents the level of dust generation during drilling in coal, limestone and iron ore mining. It may be observed here that air borne dust generated during drilling increased with the drill diameter and rock hardness.

**Table 2.2: Quantity of total dust generated from drill hole for coal, limestone and iron ore for different drill hole diameters (Sinha et al., 1982)**

| Diameter of drill hole (mm) | Amount of dust (kg) generated per meter of drilling | | |
|---|---|---|---|
| | Coal | Limestone | Iron ore |
| 60 | 3.7 | 7.8 | 12.7 |
| 100 | 10.2 | 21.5 | 35.5 |
| 150 | 23 | 48.6 | 79.5 |
| 200 | 40.9 | 86.4 | 144.5 |
| 250 | 63.75 | 134.9 | 220.7 |
| 300 | 91.92 | 194.5 | 318.2 |

Dust generation from the drilling operations can be reduced in the surface mines using dust collectors that are the most suitable instrument with less cost and high performance efficiency (Pandey, 2012).

## 2.1.2 Blasting

Most of the minerals in the world require drilling and blasting operations for the extraction. The type and quantity of the dust generated from a blast in surface mine depends on the drill hole diameter, length of the hole, type of rock to be blasted, moisture content, delay pattern and other blast geometry parameters. It also depends on the explosive properties. The dispersion of the dust generated by the surface mine blast depended on the prevailing atmospheric and the meteorological conditions. Although the dust generated by the blasting is short-lived phenomenon but if there are many surface mines in the area then the blasting can create hazardous environmental conditions (Roy et al., 2011).

Not many studies had been carried out in the past for the dust generation from the blasting operation in the surface mine. But due to expansion of the projects and requirements of large chunk of the coal and the metal, extensive blasting is carried out. The knowledge of the dispersion factor at the different times of the day at the mine could be used as a tool for the control of the air pollution due to blasting. Roy et al. (2011) reported that the effective control of blasting dust could be done by the scheduling of the blast at the time, when the dispersion factor is maximum.

## 2.1.3 Loading and unloading of material

The operation of loading and unloading of any material acts as a point source for the dust generation. The emission of particulate matter takes place for very short duration during the loading of overburden and mineral from excavator to the dumper or truck. In the same manner, particulate matter is generated when the mineral or overburden is dumped to the stockyard, crushing zone or dumping area. The generation of dust or particulate matter can be reduced very efficiently by continuous water sprinkling in these areas. New South Wales report (2010) suggested that the dust generation can also be reduced from loading operation by decreasing the dumping height.

### 2.1.4 Haul Road and Transportation of material

Haul roads are structures that facilitate plying of empty and loaded HEMM for the movement of the material to the desired location. Every surface mine has these unpaved haul roads. Thompson and Visser (2001) reported that almost 48-54% of the total respirable particulate matter from a surface mine was generated by the haul roads. Although every mine usually has water sprinklers, which continuously sprays water on the unpaved haul road to minimise the dust generation but due to the heavy traffic on the haul roads, continuous generation and emission of the particulate matter, usually occurs.

The amount of dust that will be emitted from an unpaved haul road is a function of two basic factors (ARRB, 1996):

- the erodibility of the wearing course i.e. the way an upper layer of the unpaved haul road can be detached and transported by wind and
- the erosivity of the actions to which the wearing course is subjected i.e. erosion of the unpaved haul road due to rain and other actions.

As per Sinha & Banerjee (1997), the particle size distribution of the airborne dust emanating from the haul roads of an iron ore mine was found to be log-normally distributed and further a 60% of the total SPM was within 0-10μm size range ($PM_{10}$). United States Environmental Protection Agency (EPA) proposed method to estimate emissions factors for the unpaved haul roads. As per Reed et al., haul trucks generate the majority of the dust emissions from surface mining sites (accounting for approximately 78%-97% of the total dust emissions) (Reed et al., 2007).

### 2.1.5 Crushing and Storage of the material or overburden

Crushing of material is mostly done in an enclosed area but some time it is required to crush minerals prior to dispatching it to the processing plant, in open space. In such situations, crushing of mineral can act as the point source of dust. The mine stockyards behave like a volume source of the dust due to large deposits of minerals. Sometimes, small stockyard are also referred as area source. In both these conditions, stock yard can pollute the environment up to a very large extent. However, the process of the dust generation and the accompanying emissions is complex as it is an outcome of the stockyard, meteorology and vehicles. The calculation of emission rate from stockyard is complex due to all these reasons. Hence it is required to be evaluated carefully (Gregory et al., 2010).

Most of the surface mines are having external overburden dumps (large) in extent and can generate dust in a large quantity. One can restrict the emission of the dust by external dumps by growing vegetation on it. Due to the increase in the mining activity, it is essential to locate the dumps so that the impact of the emission from the overburden dump may be minimised on the local community (Konstantinos et al., 2011).

Opencast mine also operate internal overburden dumps. It is desirable to have internal overburden dumps, to fill it inside the mine after the completion of the mining activities over external dumps as far as possible. These internal dumps are exposed to the wind erosion. Hence, they are the continuous source of the dust emission in the mine and its surroundings. The volume of the overburden also increases with the excavation of the deeper part of the mine. This enhances the degradation of the environment due to emission of the dust from these sources.

### 2.1.6 Solid Waste Handling and Re-handling

The waste which is generated during the excavation of the overburden and the processing of mineral is generally considered as solid waste. Solid waste from overburden is generally kept as overburden dump. During the handling of this material, there is point emission of the dust at the loading and unloading site. This contributes a lot in the generation of particulate matter in the mining area as it was discussed earlier in the thesis. The solid waste which is generated as a solid material from the initial mineral processing operation are generally handled in the same manner as overburden, which also contributes significantly in the generation of particulate matter.

## 2.2 Particulate matter and its residence time

Environmental Protection Agency, USEPA has defined particulate matter as a complex mixture of extremely small particles and liquid droplets. Particle pollution is made up of a number of components, including acids (such as nitrates and sulphates), organic chemicals, metals, and soil or dust particles. Particulate matter (PM) is the term used for a mixture of solid particles and liquid droplets suspended in the air. These particles originate from a variety of sources, such as power plants, industrial processes and diesel trucks, and they are formed in the atmosphere by transformation of gaseous emissions (Fierro, 2000).

Coarse particulate matter or $PM_{10}$ is defined as particulate matter having aerodynamic diameter 10 μm or less. $PM_{10}$ is generally formed by industrial activities like crushing, grinding, abrasion of surfaces etc. $PM_{10}$ remains suspended from few minutes to hours and can travel up to ten kilometres. Whereas fine particulate matter or $PM_{2.5}$ is defined as particulate matter having aerodynamic diameter 2.5 μm or less. This kind of particulate matter is mainly generated by fossil fuel combustion, vegetation burning, and the smelting and processing of metals. They remain in the atmosphere from days to weeks and travels up to thousands of kilometres.

The geometric size of a particle does not fully explain its behaviour in its airborne state. The term "particle diameter" alone is not advisable while referring to particle size of airborne dust. Therefore, the most appropriate measure of particle size is particle aerodynamic diameter. It is defined as the diameter of a hypothetical sphere of density 1 g/cm$^3$ having the same terminal settling velocity in calm air as the particle in question, regardless of its geometric size, shape and true density (WHO, 1999).

## 2.2.1 Residence time of particulate matter

Figure 2.2 shows the typical residence times in the atmosphere for particle sizes within each size range, based on gravitational settling in mixed and stirred chamber models (Hinds, 1982). Fine particles of $PM_{2.5}$ size fraction have substantially longer residence time. Therefore, it has potential to affect particulate matter concentrations for larger distance from emissions sources in comparison to the particles with aerodynamic diameters exceeding 3 μm. In this regard, fine particles act more like gases than like coarser dust particles (Watson et al., 1997).

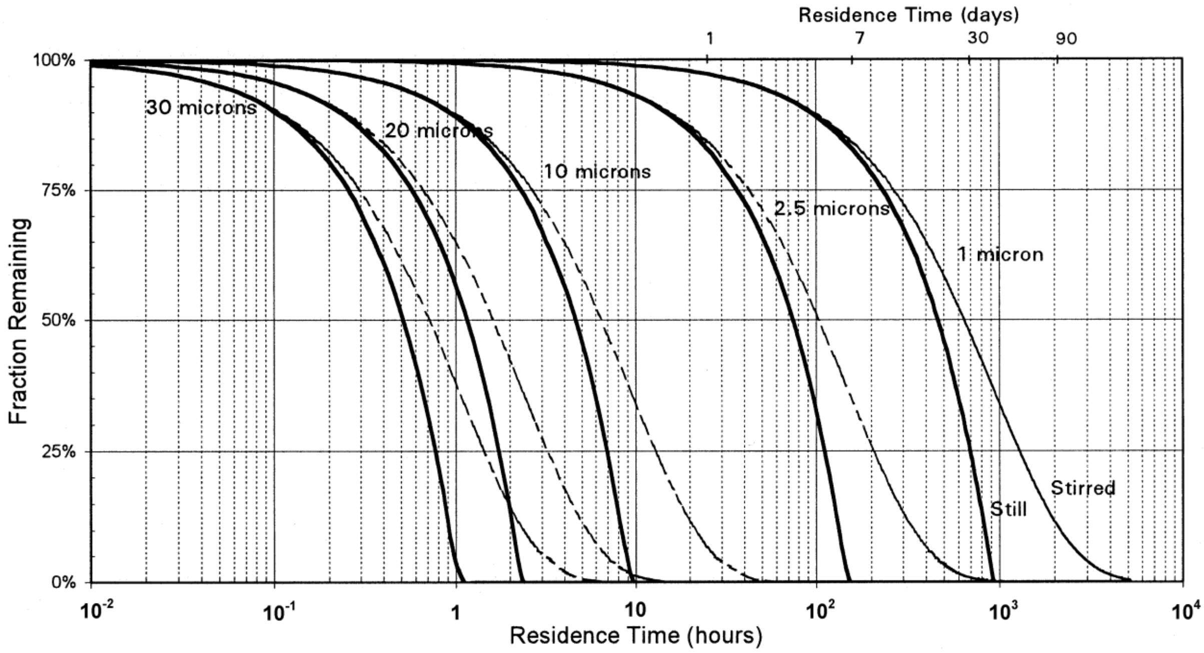


**Figure 2.2: Residence times for homogeneously distributed dust particles of different aerodynamic diameters (After Hinds, 1982)**

Particulate matter can be classified in two categories: Primary particulate matter and secondary particulate matter. Primary particulate matter consists of materials that is directly emitted into the air. Secondary particulate matter is created by the chemical reactions occurring in the atmosphere. Examples of primary particulate matter are clay, soil, and silica. Examples of secondary particulate matters are sulphate compounds and nitrate compounds. $PM_{10}$ consists mostly of primary particulate matter. Secondary particulate matter is more of a concern when dealing with particulate matters less than 2.5 μm ($PM_{2.5}$) (Seigneur, et.al., 1999).

## 2.2.2 Effects of particulate matter

Suspension of particulate matter in the atmosphere is harmful for the human as well as the surrounding environment which includes animal and plants too. The characteristics of particulate matter responsible for the health effects are physical nature, chemical nature, duration of suspension and the size.

Inhalation is the only route of exposure to the human being and the animals that is of concern in relation to the direct effects of suspended particulate matter on human health. A review of the respiratory system is required in order to understand the effects of dust, particularly silica and coal dust. Three regions of the respiratory system are important to understand the impact of dust on humans: the extrathoracic region (consists of the nose, mouth, pharynx, and larynx), the tracheobronchial region (extends from the trachea to the terminal bronchioles) and the alveolar region (contains the lungs) (Hinds, 1999). Figure 2.3 elaborates these regions:

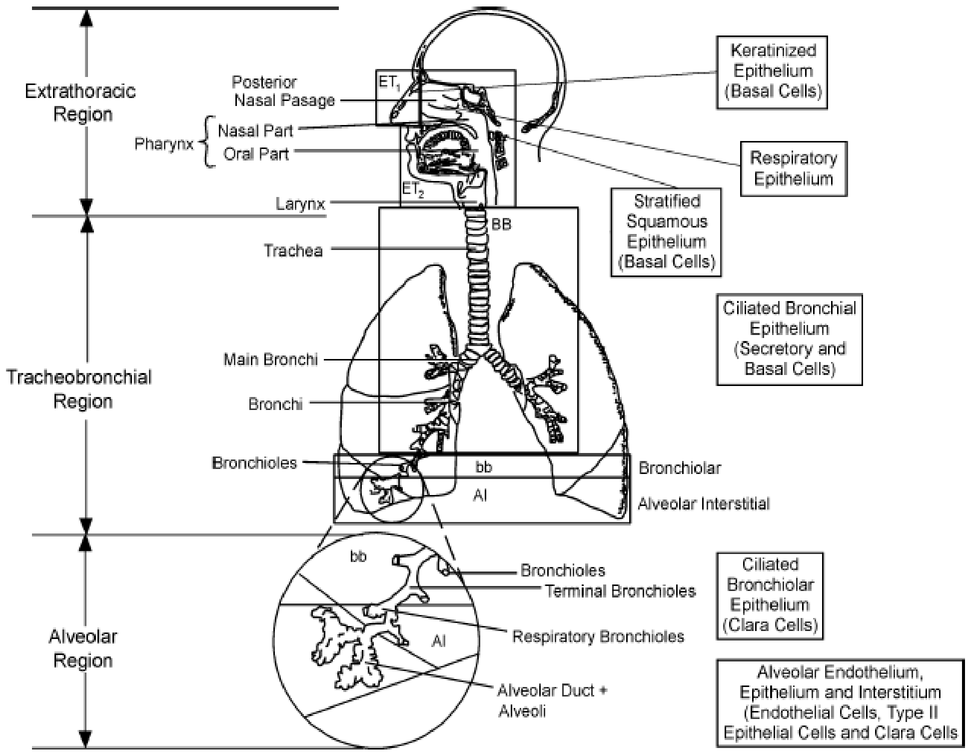


**Figure 2.3: Three regions of respiratory system and their parts (EPA, 1996)**

The alveolar region, is one where most of the impact from respiratory dust occurs. The extrathoracic and the tracheobronchial regions contain layers of mucus that help absorb and expel respiratory dust, but the alveolar region, where oxygen exchange takes place, does not have this mucus layer (Hinds, 1999). Instead, the alveolar region contains scavenging cells called microphages, which migrate to the respirable dust particles and surround and digest them, particularly if the particles are organic. However, mineral dusts are insoluble. Therefore, the microphages cannot digest these particles. Instead they attempt to move these particles to the tracheobronchial regions for the expulsion. This action may take months to years for the particles to be exhaled. Silica and the coal dusts interfere with the microphages removal attempts and are not exhaled. Instead they cause scarring of the lung tissue, also known as fibrosis which is also known as silicosis or pneumoconiosis (Wagner, 1980).

Generally coarser particles are filtered out by extrathoracic and nasopharyngeal region. In aerodynamic diameter terms, only about 1% of $PM_{10}$ particles gets as far as the alveolar region, so $PM_{10}$ is usually considered the practical upper size limit for penetration to this region. Maximum deposition in the alveolar region occurs for particles of approximately 2 µm aerodynamic diameter. Most particles larger than this are deposited further up the lung. All the deposition mechanisms are less efficient for smaller particles. Deposition is less for particles smaller than 2 µm until it is only about 10-15% at about 0.5µm. Most of these particles are exhaled again without being deposited. For still smaller particles, diffusion becomes an effective mechanism and deposition probability is higher. Deposition is therefore a minimum at about 0.5 µm (WHO, 1999).

A study done in India by Durvasula (1990) has reported the prevalence of silicosis among workers engaged in the quarrying of shale (sedimentary rock) and subsequent work in small poorly ventilated sheds. It has been reported that adults last about 14 years in this trade and are often replaced by their children who become severely ill within 5 years. An estimated 150 casualties have occurred every year and about 3500 have died in the last 25 years. The prevalence of silicosis is 54.65%, with 50% of male suffering from silicosis below 25 years of age. The same author reports, in small potteries, levels of respirable dust exceeds 25 to 90 times the occupational exposure limit of the American Conference of Governmental Industrial Hygienists (ACGIH) and the prevalence of silicosis is 31%.

Another study carried out by Saiyed and Chatterjee (1985) on slate pencil workers in Central India, for 16 months reported that 32% had progressed, and that mortality was high. The mean age of the workers who died was 35 and the mean duration of the exposure was 12 years.

In a surface coal mine, silica is mostly associated with the coal and the workers usually suffer from silicosis. The studies reported above also suggest that there are many health problems associated with these diseases.

There are other undesirable results from $PM_{10}$ exposure in addition to the health effects. $PM_{10}$ affects visibility in the air and also contributes to the climate change (Reed, 2003). Smaller air borne particles hinder visibility as they scatter and absorb light that travels to the observer from an object. This action results in extraneous light from the sources other than the observed object being detected by the observer, thus impairing visibility. Climate change may also occur, because the small particles in the atmosphere absorb and reflect the radiation from the sun, affecting the cloud physics in the atmosphere. $PM_{10}$ may also have an effect on materials such as paint, wood, metals, etc. The effects are dependent upon the amount of $PM_{10}$ in the atmosphere, the deposition of the $PM_{10}$ on the material, and the elemental composition of the $PM_{10}$ (USEPA, 1996).

## 2.2.3 Regulations pertaining to particulate matter

There are several regulations in India pertaining to the particulate matter and other air pollutants that are framed under the Environment (Protection) Act, 1986. Central Pollution Control Board in association with Indian Institute of Technology, Kanpur had revised the standards and named it as the National Ambient Air Quality Standards (NAAQS) in 2009. The Table 2.3 states important technical standards applicable in India.

**Table 2.3: National Ambient Air Quality Standards (NAAQS, 2009)**

| **S. No.** | **Pollutant** | **Time Weighted Average** | **Concentration in Ambient Air** | | |
|---|---|---|---|---|---|
| | | | **Industrial, Residential, Rural and other area** | **Ecologically Sensitive Area (notified by Central Government)** | **Methods of Measurement** |
| **1.** | $SO_2$, $\mu g/m^3$ | Annual* | 50 | 20 | ▪ Improved West and Gaeke<br>▪ Ultraviolet fluorescence |
| | | 24 hours** | 80 | 80 | |
| **2.** | $NO_2$, $\mu g/m^3$ | Annual* | 60 | 60 | ▪ Gravimetric<br>▪ TOEM<br>▪ Beta attenuation |
| | | | 24 hours** | 100 | |
| **3.** | $PM_{10}$, $\mu g/m^3$ | Annual* | 60 | 60 | ▪ Gravimetric<br>▪ TOEM<br>▪ Beta attenuation |
| | | 24 hours** | 100 | 100 | |
| **4.** | $PM_{2.5}$, $\mu g/m^3$ | Annual* | 40 | 40 | ▪ Gravimetric<br>▪ TOEM |

| | | | | | |
|---|---|---|---|---|---|
| | | 24 hours** | 60 | 60 | ▪ Beta attenuation |
| **5.** | $O_3$, µg/m$^3$ | 8 hours** | 100 | 100 | ▪ UV photometric<br>▪ Chemiluminescence<br>▪ Chemical Method |
| | | 1 hour** | 180 | 180 | |
| **6.** | Lead (Pb), µg/m$^3$ | Annual* | 0.50 | 0.50 | ▪ AAS/ICP method after sampling on EMP 2000 or equivalent filter paper<br>▪ ED-XRF using Teflon filter |
| | | 24 hours** | 1 | 1 | |
| **7.** | CO, mg/m$^3$ | 8 hours** | 2 | 2 | ▪ Non Dispersive Infra Red (NDIR) spectrosopy |
| | | 1 hour** | 4 | 4 | |
| **8.** | Ammonia ($NH_3$) µg/m$^3$ | Annual* | 100 | 100 | ▪ Chemiluminescence<br>▪ Indophenol blue method |
| | | 24 hours** | 400 | 400 | |
| **9.** | Benzene | Annual* | 5 | 5 | ▪ Gas chromatography based on continuous analyzer |

| | | | | | |
|---|---|---|---|---|---|
| | | | | | ▪ Adsorption and Desorption followed by GC analysis |
| **10.** | Benzopyrene (BaP) - particulate phase only, ng/$m^3$ | Annual* | 1 | 1 | ▪ Solvent extraction followed by HPLC/GC analysis |
| **11.** | Arsenic (As), ng/$m^3$ | Annual* | 6 | 6 | ▪ AAS/ICP method after sampling on EMP 2000 or equivalent filter paper |
| **12.** | Nickel (Ni), ng/$m^3$ | Annual* | 20 | 20 | ▪ AAS/ICP method after sampling on EMP 2000 or equivalent filter paper |

Note: The annual weighted average * is calculated by taking minimum 104 measurements in a year at a particular site at the rate of twice a week for 24 hour at uniform interval. Calculation of 24 hourly or 8 hourly or 1 hourly monitored values ** (as applicable), is compiled with 98% of the time in a year.

## 2.3 Field Studies of Dust Management at Surface Mine Operations

Several field studies have been carried out for the measurement of the dust concentrations as well as to validate the different air quality dispersion models for surface mining operations. These field studies are further categorised as follows:

### 2.3.1 Field Studies on the characterisation of dust from the mining activities

A study completed by Jamal et al. (1997) sampled dust from different mining operations at a surface coal mine in India and analysed it for textural and mineralogical characteristics. The operations sampled included drilling, blasting, hauling, dumping and loading of material. Dust deposition was measured at each of the operations, both upwind and downwind, using double-sided tape attached to a stub to collect the sample. Characteristics of dust analysed from each operation included particle shape, particle size, and particle composition using micron photosizer and petrological microscope. Particle shape was divided into several categories: angular, sub-angular, and sub-rounded. Particle shapes were determined from the samples taken, and placed into different categories, resulting in a particle size distribution. Particle size distribution was determined for each sample for different mining operations. The size categories were based on the following categories.

**Table 2.4: Classification of SPM in mine environment** (After Jamal et al., 1997)

| Group | Size range in µm | Effects |
|---|---|---|
| Superfine | Less than 0.5 | Least effect on respiratory system |
| Fine | 0.5 to 2.5 | Maximum effect on respiratory system |
| Medium | 2.5 to 5.0 | Moderate effect on respiratory system |
| Coarse | 5.0 to 15.0 | Mostly affects visibility of atmosphere |
| Very coarse | 15.0 and above | This fraction has fast settling rate usually causes soiling of machineries |

It was found that respirable dust (up to 5 µm) depends on the mining activity and respirable dust in mining operations was found to be in the range of 20% to 43% of total particulate matter, while in the residential areas they were higher - above 45%. Drilling operation was found to produce the most angular shaped particles. Composition of the dust was broken into categories of free silica, silicate, iron oxides, and coal particles. The percentage of coal particles was higher in coal handling locations, while the percentage of silica was higher in overburden drilling operations (Jamal et al., 1997).

Sinha & Banerjee (1997) carried out a study in Noamundi Iron ore mine of TISCO in January- February 1994. This study was mainly done to characterise the airborne aerosol contributed from the unpaved haul road, traffic exhaust and the re-entrained dust from other mining activities. SPM from different mining activities was collected by three high volume samplers and particle size distribution of SPM emanating from haul road of the mine was analysed by using a master particle sizer. Free silica content of SPM was found by orthophosphoric acid method after collecting it in cascade impactor. Eight trace elements in SPM was determined by atomic absorption spectrophotometer. It was concluded from the study that the particle size emanating from the haul road is log-normally distributed and about 60% of the total SPM of the range of 0-10 µm ($PM_{10}$). The free silica content for different size ranges of SPM was found between 1.1 to 1.3 % and slightly on higher side for finer size fraction. Concentrations of eight trace elements except lead was found within the permissible limit.

### 2.3.2 Field Studies on quantification of the dust and development of emission factor of the mining activities

Ghose et al. attempted a study on an opencast project (OCP) for coking coal to identify the main sources of air pollution from an OCP by (Ghose et al., 2000). The study observed that the opencast mining activities like the topsoil removal, overburden removal, coal extraction, size reduction etc. generate large quantities of dust. By utilising the emission factor data, they calculated that the topsoil removal generated 69.9 kg of dust per day. Overburden removal operations generated 666.0 kg of dust per day and the extraction of coal contributed about 256.9 kg of dust per day. Dust generation due to size reduction contributed much more amounting to 6812.5 kg of dust per day. Thus the mining activities in the area had generated 7798 kg of dust per day. Wind erosion had also contributed to dust generation depending on its velocity, direction and other influencing micrometeorological factors. Wind erosion had generated dust of about 1568 kg per day. Blasting had also caused dust generation but due to unavailability of emission factor for data, its actual quantity could not be estimated. It was concluded in the study that all the dust generated from the different mining operations were not dispersed in the surrounding atmosphere. It was also concluded that the main impact of the dust generation is was in the working area that is gradually diluted (Ghose et al., 2000).

Another study in the same mine by Ghose et al. (2002) was done for assessment of the concentration of SPM and RSPM from the different mining activities. At six different locations of the work zone, dust assessment was done by high volume samplers with impingers. Impingers were utilised to sample the sulphur dioxide ($SO_2$) and oxides of nitrogen ($NO_x$). Meterological data like wind speed and wind direction, mixing height and ventilation coefficient were collected from a monostatic and Doppler SODAR. The results of the study depicted that the maximum concentration was observed at the dragline section and the next higher concentrations was observed at the haul road. Both the sides of the haul roads were very dusty. For four season data, the status of work zone air quality was considered with all the activities occuring at the same time. Average concentration of the air pollutants were found to be 1473.66 $\mu g/m^3$ for SPM, 197.79 $\mu g/m^3$ for RPM, 74.90 $\mu g/m^3$ for $SO_2$ and 68.15 $\mu g/m^3$ for $NO_x$ as found in the annual average.

Xuan et al. estimated the dust annual mean emission rates of $PM_{50}$, $PM_{30}$ and $PM_{10}$, using the modified USEPA empirical formulae and performed comparative characterization of the dust sources in the Northern China by combining the geographical, pedological and 30-year (1951–1980) climatological data (Xuan et al., 2002). They characterised three broad types of dust sources in Northen China: Type 1. Deserts in dry-agricultural areas, Type 2. Gobi-deserts and deserts located on the plateaus, and Type 3. Deserts and gobi-deserts located in topographical lows. Type 1–3 sources contributed 1%, 35% and 64%, respectively, to the total annual mean emission of $PM_{10}$ dust. It was estimated in their study that the main dust sources in the Northern China were the Taklimakan Desert for which the annual mean $PM_{10}$ emission rate was 0.38 ton/ha yr, the Central Gobi-desert annual mean $PM_{10}$ emission rate was 0.28 ton/ha yr and the deserts located on the Alxa Plateau annual mean $PM_{10}$ emission rate was 0.05 ton/ha yr. It was concluded that the annual mean dust emission of $PM_{50}$, $PM_{30}$ and $PM_{10}$ was 42.6, 24.8 and 8.4 million tons, respectively and of these, more than half of the total annual dust was emitted in the spring season itself (Xuan et al., 2002).

Chaulya worked in Lakhanpur area of Ib Valley Coalfield of the Odisha state in India for the assessment and management of the air quality. The 24-hr average concentrations of the total suspended particulate (TSP) matter, respirable particulate matter ($PM_{10}$), sulphur dioxide ($SO_2$) and oxides of nitrogen ($NO_x$) were monitored during 1 year period. Samplings were done at a regular interval throughout the year, at 13 monitoring stations in the residential areas and four monitoring stations in the mining/industrial areas. The 24-hr average TSP and $PM_{10}$ concentrations ranged from 338.8 to 799.8 μg/m$^3$ and 102.5–425.6 μg/m$^3$ for industrial area, and 72.3–497.1 μg/m$^3$ and 40.8 2 171. μg/m$^3$ for residential area, respectively. During the study period, 24-hr and annual average TSP and $PM_{10}$ concentrations exceeded the respective standards set in the national ambient air quality standard (NAAQS) at most of the residential and the industrial areas. However, 24-hr and annual average concentrations of $SO_2$ and $NO_x$ were well within the prescribed limits of the NAAQS both in the residential and the industrial areas. The annual and the 24-hr average concentrations varied from 23.3 to 36.8 μg/m$^3$ and 16.0–55.2 μg/m$^3$ for $SO_2$ and 23.9 2 41.9 μg/m$^3$ and 19.0–58.1 μg/m$^3$ for $NO_x$, respectively. The temporal variations of TSP and $PM_{10}$ fitted polynomial trend with an average correlation coefficient ( $R^2$ ) of 0.77 (±0.17) for TSP and 0.85 (±0.10) for $PM_{10}$. On an average the $PM_{10}$ in the ambient air of the mining area constituted 31.94 (±1.76)% of the TSP. The linear regression correlation coefficient ( $R^2$ ) between TSP with $PM_{10}$ and $NO_x$ with $SO_2$ was 0.86 (±0.12) and 0.57 (±0.20) respectively. Chaulya concluded from the above study that the TSP and the $PM_{10}$ were the major sources of emission from various opencast mining activities and the emissions of $SO_2$ and $NO_x$ were negligible (Chaulya, 2004).

Ghosh (2007) examined the sources of dust from the coal mining activities. He focused on the quantification of dust emission with the development and the use of the emission factors. He utilized the prediction equations for the development of emission factors. For the applications of this concept, one large opencast coal project of Bharat Coking Coal Ltd. (BCCL) was investigated, and the total amount of dust emitted due to different mining activities was calculated by the utilization of emission factor data. He had also focussed on the significance of the environmental protection and likely impacts of this study (Ghose, 2007).

According to Ghose, the total amount of the dust generated, as calculated by the utilization of emission-factor data, was found to be 9368.2 kg/day. It was concluded that the total amount of the dust generated by the different mining operations, not all dispersed into the atmosphere. He concluded that the dust generated mainly impacted on the air quality of the working area (around the sources of pollutant generation), with gradual dilution (Ghose, 2007).

Petavratzi et al. investigated the propensity of a limestone to generate dust due to handling and comminution processes. Dust liberation mechanisms from the different quarry operations were also studied. The dustiness of a limestone was assessed using the Warren Spring Laboratory rotating drum (HSE-WSL). The effect of the operating parameters of the WSL rotating drum to the dustiness of the limestone was evaluated prior to testing. Preliminary testing on the effect of the operational parameters to the dustiness values reflected that the consistency of the end results were closely related to them. Thus, they need to be carefully controlled. The control testing was also done to identify the maximum dustiness value per operational parameter, so as to define an optimum set for the limestone sample. Petavratzi et al. found that the different operational parameters and the sample fractions, had not much influenced the particle size profile of the limestone dust. Around 25% of the particles had been defined with a particle size below2.5 µm and in average 80% of the limestone dust particles were described by a size below 10 µm. The percentages passing 10 and 2.5 µm were sufficient indicators of the hazardous potential of the limestone dust to human health and the environment. They concluded that the importance should be paid more on control over operational parameters in industrial processes such as the time scale, or the limestone mass in terms of the dust generation potential. Further lower operational time scales even with higher utilization of limestone masses could reduce dust levels (Petavratzi et al., 2007).

Sharma and Siddiqui focussed on the assessment and the management of the air quality at Jayant open cast coal mine situated at Jayant in Sidhi district of Madhya Pradesh, India from January 2007 to December 2008. The 24 hour average concentration of the total suspended particulate (TSP) matter, dust fall rate, sulphur dioxide ($SO_2$) and the oxide of nitrogen ($NO_x$) were monitored for 2 year period. Sampling were done at a regular interval throughout the year at five monitoring stations in the residential area, the mining area and the industrial areas. The sampling of particulate matter, $SO_2$ and $NO_x$ were done using High Volume Samplers (HVS). Samples were also collected for the two years using glass fibre filter paper on fort-nightly basis. The same filter paper was used for the determination of the air borne metals (Ca, K, Na, Ni, Cr, Zn, Pb, Mn and Cd) in TSP. The filter paper was cut into fine pieces and digested for 30 minutes in 100 ml of concentrated hydrochloride solution. The metals in the pooled filtrate were analysed on Atomic Absorption Spectro photometer. They observed that the gaseous pollutants had maximum concentration along the major road sides followed by the minor road side, over burden than in the residential area. High levels of $SO_2$ and $NO_x$ at road side might be attributed to the continuous movement of the heavy duty vehicles for the transporting of the coal from the mining place to the distribution or dumping. They concluded that higher annual average were recorded around coal mines in Singrauli field. The 24 hr. and annual average concentration of $SO_2$ for the residential area were 20.5-24.3 μ/m$^3$, for the industrial area were 15.3-30.8 μ/m$^3$ and for the non-residential were 19.7-25.3 μ/m$^3$. These figures were well within the prescribed limits of the NAAQS at all the monitoring stations on the diurnal scale. For settled dust, concentration of the metals were found in the following order Zn>Mn>Pb>Cd>Ni>Cr>. Unlike TSP in settled dust concentration Cd was more than those of Ni and Cr (Sharma & Siddiqui, 2010).

Chaulya (2006) studied three surface iron ore mines to determine the total suspended particulate emission rate and to develop the formulae for the emission rate of various surface iron ore mining activities. He covered various mining activities and locations including waste loading and unloading, iron ore loading and unloading, screening plant, exposed waste dump, stock yard, exposed pit surface, transport road and haul road. A set of 10 formulae were developed to calculate the total suspended particulate emission rate of various surface iron ore mining activities. Primary data generation was carried out for this project. This could be broadly categorised under three major sub-heads: micrometeorological data, emission inventory data, and dust or waste material quality data. The secondary data regarding mining details were also collected. The details for the geology, meteorological data etc. were collected from the different sources like mine plans, environmental management plan report, mine project reports and mine management of the respective mine. Hourly micro-meteorological parameters were measured by installing automatic weather monitoring stations at the respective study sites during the study period. Long-term meteorological data were also collected from the nearest meteorological stations of the Indian Meteorological Department at the respective areas. Air quality was measured by the means of the high volume samplers with an average flow rate greater than 1.1 m$^3$/min. The developed formulae were validated with the measured emission data at another surface iron ore mine. He concluded that the accuracy of the activity-wise emission rates calculated using the developed formulae was found to vary from 92 to 97% of the actual field measurement data.

## 2.3.3 Field Studies on the impact on environment and health from mining activities

Matejicek et al. (2008) carried out the spatio-temporal modelling of the living environment over surface mining areas as well as in the neighbouring residential zones. A wide range of the terrain measurements, existing spatial data, time series, results of spatial analysis and inputs/outputs from the numerical simulations were required. This was done using Geographical Information Systems (GIS). In order to demonstrate the integration of the spatial data, time series and methods in the framework of the GIS, this study focused on the modelling of the dust transport over a surface coal mining area, exploring spatial data from the 3D laser scanners, GPS measurements, aerial images, time series of meteorological observations, inputs/outputs of the numerical models and the existing geographic resources. The digital terrain models, layers including Global Positioning System (GPS) thematic mapping and scenes with simulation of wind flows were done to visualize and interpret the coal dust transport over the mine area and a neighbouring residential zone.

The study observed that a temporary coal storage and sorting site, located near the residential zone, was one of the dominant sources of the emissions. Using numerical simulations, the possible effects of wind flows were observed over the surface, modified by the natural objects and the man-made obstacles. It was identified in the study that the coal dust was drifted with the wind in the direction of the residential zone and was partially deposited in this area. A more accurate simulation of wind flows over the temporary storage and sorting site was obtained in the study by the 3D laser scanning and GPS thematic mapping which had created a more detailed digital terrain models. Thus the visualization of the wind flows over the area of interest combined with 3D map layers had enabled the exploration of the processes of coal dust deposition at a local scale. The study suggested an approach for dust-transport modelling coupled with spatial data focused on the construction of digital terrain models and thematic mapping. The numerical simulations based on Reynolds averaged Navier-Stokes equations that would provide realistic predictions (Matejicek et al., 2008).

Singh & Sharma studied the spatial distribution of the air Pollutants in the coal mining areas of the Raniganj Coalfields, India. The concentrations of SPM, sulphur dioxide and nitrogen oxides were measured using high volume samplers stationed at various locations surrounding the mining areas for one year duration. No separate measurements of the individual mining operations, such as drilling, loading, and hauling were made there. The results showed that the dust concentration levels differed between day and night. At areas surrounding underground operations the difference was less than 50 $\mu g/m^3$ and for surface operations the difference was more than 50 $\mu g/m^3$. The seasonal variations in the minimum background levels of dust concentrations had a range of 100 $\mu g/m^3$ for monsoon season, 150 $\mu g/m^3$ for summer season, and 200 $\mu g/m^3$ for the winter and spring seasons. From the data presented and work zone analysis, they concluded that very high concentration of SPM existed. Such levels were injurious to the workers' health.

### 2.3.4 Field Studies on the dust emission from the haul road

An intensive study was carried out by Sinha & Banerjee (1997) in Noamundi Iron ore mine of TISCO in January- February 1994. This study was mainly done to characterise the airborne aerosol mainly contributed from the unpaved haul road, traffic exhaust and re-entrained dust from other mining activities. SPM from the different mining activities were collected by three high volume samplers and the particle size distribution of SPM emanating from haul road of the mine was anaysed by using a master particle sizer. Free silica content of SPM was found by orthophosphoric acid method after collecting it in cascade impactor. Eight trace elements in SPM were determined by the atomic absorption spectrophotometer. The study concluded that particle sizes emanating from the haul road were log-normally distributed and about 60% of the total SPM were of the range of 0-10 µm ($PM_{10}$). The free silica content for different size ranges of SPM were found between 1.1 to 1.3% and were slightly on the higher side for the finer size fraction. Concentrations of eight trace elements, except lead was found within the permissible limits.

In another study Ho et al. , $PM_{10}$ and $PM_{2.5}$ chemical source profiles from Hong Kong were investigated for paved road dust and soil. These profiles were used for the urban-scale speculated emissions inventories and for source apportionment by receptor modelling. Five urban soil and five paved road dust samples were collected, dried and sieved, resuspended in a laboratory chamber and were air drawn through $PM_{10}$ and $PM_{2.5}$ inlet onto Teflon and quartz filters. The filter samples were submitted for both gravimetric and chemical analyses. The $PM_{2.5}$ was comprising 11–30% of the $PM_{10}$ in all geological samples. Al, Si and organic carbon (OC) were abundant constituents in all paved road dust and soil samples. Results were compared with the data obtained from Hong Kong Environmental Protection Department (HKEPD). It was observed in the study that high correlations ($r > 0.7$) were there in between mass concentration and traditional crustal elements (Si, Al, K, Ca, Ti, and Fe) in soil and dust samples. Also higher correlation was observed in $PM_{10}$ fraction than in $PM_{2.5}$ of both samples. However, when the calculated crustal matter concentrations were compared with the measured mass concentrations, only the measured mass concentrations of PM10 soil samples have good agreement with the calculated values (Ho et al., 2003).

Organiscak et al. conducted two field surveys to quantify fugitive dust generation and dispersion from the truck traffic on the unpaved and the untreated mine haulage roads. Results found at least 80% of the airborne dust generated by haul trucks was larger than 10 µm. Total thoracic and respirable gravimetric dust measurements, showed the highest concentrations near the road with a rapid decrease in concentrations at a distance of 30.5 m of the road. It was concluded in the research hat the thoracic and total concentrations were 3 to 4 times higher and 8 to 11 times higher, respectively, than the respirable dust concentrations (Organiscak et al., 2004).

The study also worked for the evaluation of dust exposure to the truck drivers following the lead haul truck (Reed et al., 2005). A time study was also conducted for the haul trucks using the haul road throughout the entire study time period. The haul truck time entering and exiting the test section of the road, type of the haul truck and speed and the direction of travel were recorded. At the coal preparation plant, the instantaneous respirable concentration levels ranged anywhere from 0.001 to 21.50 $mg/m^3$ within 5 to 15 seconds after the 45 to 54 t off-road haul trucks, travelling at 7.1 m/s had passed. However, the average peak instantaneous respirable concentration level were approximately 2.35 $mg/m^3$ within the same time period. It concluded from the percentile graphs that a truck driver might be exposed to respirable dust concentration levels above 3.55 $mg/m^3$ for 25% of the time that the driver is following 5 to 15 seconds behind other trucks. At stone quarry, the instantaneous respirable concentration levels ranged from 0.01 to 8.03 mg/m$^3$ within 0 to 20 seconds after the haul trucks had passed. Most of these were over-the-road tandem axle trucks, with some trailer trucks capable of 18 t payloads, traveling at 7.0 m/s. The high traffic volume at this site precluded the calculation of an average peak respirable concentration level. However, using the box and whisker graphs, they concluded that a truck driver might be exposed to respirable dust concentration levels within a range of 0.75 to 2.75 mg/m$^3$ for 15% of the time that the driver is following 10 to 20 seconds behind other trucks. Additionally, the respirable concentration levels can exceed 2.75 mg/m$^3$ for 10% of this time. Fianlly, from respirable and $PM_{10}$ dust sampling conducted during the study, it was found that airborne dust generated by the haul trucks consists of, on average, 14.5% <10 μm and 3.5% <3.5 μm. It had been assessed that the critical time period of following a truck is from 0 to 20 seconds, with maximum exposure occurring between 4 and 15 seconds. The study suggested by implementing a policy to ensure that trucks do not follow within 20 seconds of another truck can result in a 41% to 52% reduction in the airborne respirable dust exposure to the driver of the following truck.

### 2.3.5 Field Study on the dust emission from the overburden dump

Konstantinos et al. (2011) addressed the problem of atmospheric dispersion of particulate matter (PM) from the overburden dumps of a mine, using a steady–state Lagrangian numerical model and the integral model AERMOD. The analysis included most of the complex physical phenomena in atmospheric diffusion. This study was done for large mine that provided lignite for the thermal power stations of the Hellenic Public Power Corporation in the vicinity of the city of Amyndaion in Northern Greece. The excavated land was dumped in nearby open pits, which were planned to be extended towards South. These pits were continuous sources of air–suspended particulate matter that were affecting the nearby residential areas. The numerical model was applied for a number of specific meteorological scenarios, while the integral model for the five years period. The overall application of the complex numerical model in an environmental study was considered successful in the study. The comparison of the two models showed that both the predictions agreed fairly well, though the numerical model some time had tendency to underestimate the concentration levels. This could be due to use of a few meteorological scenarios and finite number of particles used in the Lagrangian model. However, caution must be shown on incorporating atmospheric boundary layer (ABL) parameters in the numerical model as it required experience on dealing both with numerical models and atmospheric phenomena. The results obtained from the study with both models had shown that the yearly average concentration induced solely by the overburden dump was approximately 2 μg/m$^3$. This concentration was quite lower than the European limit for $PM_{10}$ of 40 μg/m$^3$ (Konstantinos et al., 2011).

### 2.3.6 Field Studies for the development of a model and its validation on dust dispersion

Reed and Westman had undertaken a study related to the dispersion of the dust from a haul truck to develop a model which can replace Industrial Source Complex 3 (ISC3) model. The ISC3 model was created by USEPA to estimate the concentration of $PM_{10}$ from the different industrial operations. However, it had been proven that ISC 3 used to over-predict actual $PM_{10}$ dispersion concentrations by a factor of 2 – 5 (Cole & Zapert, 1995). In this study a new model called the Dynamic Component program was developed focusing on the estimation of $PM_{10}$ dispersion from haul trucks. Most of the researches had shown that the majority (80 – 96%) of $PM_{10}$ emissions from surface mining operations are from mobile sources.

The Dynamic Component model uses the same approach as the ISC3 model, where the dispersion of particulate matter from a point source is represented by the Gaussian equation 2.1 (USEPA 1995):

$$\chi = \frac{Q}{2\pi w_s \sigma_y \sigma_z}\left[exp\left\{-0.5\left(\frac{y}{\sigma_y}\right)^2\right\}\right] \qquad (2.1)$$

where:

$\chi$ = hourly concentration at downwind direction at distance x, ($\mu$g/m$^3$)
Q = the pollutant emission rate, (g/s)
K = conversion factor 1 x $10^6$ for $\chi$ in $\mu$g/m$^3$and Q in g/s
$w_s$ = wind speed, (m/s)
$\sigma_y$ = standard deviation of lateral concentration distribution, (m)
$\sigma_z$ = standard deviation of vertical concentration distribution, (m), and
y = distance to monitoring location, measured perpendicular to wind distance, (m).

The emission rate Q for $PM_{10}$ is calculated for the haul trucks using equation 2.2, the emissions factor published in AP-42 (USEPA 1998). This emission factor is represented as:

$$Q = \frac{2.6\left(\frac{s}{12}\right)^{0.8}\left(\frac{W}{3}\right)^{0.4}}{\left(\frac{M}{0.2}\right)^{0.3}} \qquad (2.2)$$

where:

Q = emissions from a haul truck in pounds/vehicle mile travelled, (lb/vmt)
s = surface material silt content, (%)
W = mean vehicle weight, (t), and

M = surface material moisture content, (%)

The Dynamic Component program was created using Microsoft Visual Basic 6.0 and the program was designed to calculate the concentration of $PM_{10}$ at a receptor from a haul truck travelling along a predetermined path using above equations. Only straight paths were allowed in the initial release of the program. The program emulates the calculations used in ISC3, but also introduces the Dynamic Component described above (Reed et al, 2005). Instead of dividing the emissions of the source over the area of the mobile source path, the entire emissions from the haul truck are applied at points along the path of the source in Dynamic Component program. This results in an array of $PM_{10}$ concentrations that are then averaged for each receptor. This methodology has produced promising results. The Dynamic Component results showed on an average 77% improvement over the ISC3 results in comparisons with actual field results (Reed, 2003).

Chaulya et al. attempted the determination of emission rate for the SPM to calculate emission rates of the various opencast mining activities and the validation of commonly used two air quality models Fugitive Dust Model (FDM) and Line Sources Model (PAL2) for Indian mining conditions. These objectives were achieved by selecting eight coal and three iron ore mines. Site specific emission data were generated by considering types of mining, methods of working, geographical locations, accessibility and resource availability. The study covered various mining activities and locations including drilling, overburden loading and unloading, coal/mineral loading and unloading, coal handling or screening plant, exposed overburden dump, stock yard, workshop, exposed pit surface, transport road and haul road. Validation of the models were done by running them separately for the same set of input data. Three receptors location were identified where actual measurements were done with the high volume samplers and the same locations of receptors were fed to both the models to predict the concentration of SPM. Statistical analysis was carried out to assess the performance of the models based on a set measured and predicted SPM concentration. The value of coefficient of correlation for PAL2 and FDM was calculated to be 0.990–0.994 and 0.966–0.997, respectively. They found a fairly good agreement between measured and predicted values of the SPM concentration. The average index of agreement values for PAL2 and FDM was found to be 0.665 and 0.752, respectively. This represented that the prediction by PAL2 and FDM models were accurate by 66.5 and 75.2%, respectively. They concluded that FDM was more suitable for the Indian mining conditions (Chaulya et al., 2003).

Gariazzo et al. (2004) developed, the MINERVE-SPRAY modelling system that was applied to a cement facility located near the city of Guidonia, about 30 km from Rome (Italy). The simulation was performed during a 4 days autumn period, characterized by a high pressure system and high insulation, for a square domain of 10 km width centred on the cement stack and covering the surrounding populated areas. Stack emission data of $NO_x$, $SO_2$, CO and plume temperature were provided to the model system. The Lagrangian particle dispersion model SPRAY, was then applied to reproduce the 3-D concentration fields. The comparison of simulation results with the monitoring data have shown good agreement for $NO_x$, and $SO_2$ in the study , though the latter could only be compared as order of magnitude due to the low concentration values observed (Gariazzo et al., 2004).

Kumar et al. statistically evaluated, the AEMOD, a main model for the dispersion modelling for an urban area. AERMOD was used to conduct multiple source evaluation on a 20 km x 20 km grid using emission data for Lucas County, Ohio for the year 1990. Subsequently evaluation shed light on some of the aspects of the performance of this regulatory model. As an input for emission inventory, there were about 123 stacks considered which were emitting sulphur dioxide in the Lucas County area. These stacks were divided into three groups. The first group was consisted of 16 stacks that cause about 96.0% of the pollutant emissions and the annual emission rate of each of the individual stacks is greater than 210 tons. The second group consisted of 28 stacks causing 3.7% of the pollutant emissions and each of the individual stack's annual emissions rate was greater than 5 tons but less than 210 tons. The third group consisted of 79 stacks causing 0.24% of the pollutant emissions and each of the individual stack's annual emission rate was less than 5 tons. The second and third groups of stacks were modelled as super stacks. Input meteorological parameters required by the pre-processor AERMET of AERMOD were obtained in raw form for the year 1990 from the WEBMET, a database of meteorological resource center. The upper air meteorological data for the AERMOD model were prepared for the year 1990 using the data from the Toledo Express Airport and Flint, Michigan. For evaluation, the data were collected from the two monitoring stations in Lucas County, located at 26 Main Street and 600 Collins Park. These monitoring data were consisted of sulphur dioxide hourly values of concentration observed at each station (Kumar et al., 2006).

All data sets of observed and predicted concentrations were divided into two categories, based on the Monin–Obukhov length (L) as used in the development of the AERMOD model. When L is greater than 0, the case is termed as stable case and convective case when L is less than 0. These categories were further divided into subcategories to evaluate the performance of the AERMOD model, using the above categories. It concluded that the performance of the model had not agreed as per the criteria stated by Kumar et al, 1993, with observed values for the 1-h and 3-h averaging periods. However, the model had shown a better performance for the 24-h concentrations as compared to the 1-h and 3-h averaging periods. In both the stable and convective cases, the model under predicted all the subcategories. The model predictions had become better as the averaging period increases. The 24-h predictions were better than the 3-h predictions and the 3-h predictions were better than the 1-h predictions (Kumar et al., 2006).

Kesarkar et al. tried to overcome the limitation of the AERMOD. The limitation of the AERMOD was that it required steady and horizontally homogeneous hourly surface and upper air meteorological observations. However, observations with such frequency were not easily available for most locations in India. To overcome this limitation, the planetary boundary layer and surface layer parameters required by AERMOD were computed using the Weather Research and Forecasting (WRF) Model (version 2.1.1) developed by the National Center for Atmospheric Research (NCAR). An offline pre-processor was developed to coupling with AERMOD. Using this system, the dispersion of the respirable particulate matter (RSPM/$PM_{10}$) over Pune, India had been simulated. Data from the emissions inventory development and field-monitoring campaign (13–17 April 2005) conducted under the Pune Air Quality Management Program of the Ministry of Environment and Forests (MoEF), India and USEPA, had been used to drive and validate AERMOD. The required coupler for WRF–AERMOD offline coupled system had been developed to derive the parametric boundary layer and surface parameters for a given location from WRF model output. It directly generated the AERMOD meteorological input files, by-passing the need for AERMET pre-processor and thus any observational data requirement. The comparison between the simulated and the observed temperature and wind fields showed that the WRF succeeded in the generation of meteorological inputs required for the AERMOD. The effect of the transient deviations between observed and simulated meteorological parameters can be better evaluated by comparing the hourly AERMOD outputs with hourly monitored values, which were not available for this study period. It was concluded that comparison of the observed and the simulated concentrations of $PM_{10}$ had shown that concentrations were generally underestimated in the simulations, except over residential area. Further, it was also shown that almost for all days during field campaign the concentrations of $PM_{10}$ pollutants were relatively higher in the central locations than as compared to the neighbouring regions of the Pune city. This may be attributed to the relatively heavy winds from the north-westerly/westerly directions than other wind directions in summer seasons and the accumulation of $PM_{10}$ (Kesarkar, 2006).

Tecer, proposed artificial neural networks to predict the concentrations of $SO_2$ and PM at two different stations in Zonguldak city. This is situated near Black Sea in a coal and steel industry region in Turkey. The established artificial neural network models had taken meteorological parameters and historical data on the observed $SO_2$, Particulate matter (PM) as input variables. The models were based on the three-layer neural network trained by a back-propagation algorithm. The neural network model was developed for each station (Bahçelievler, City Centre) and for each pollutant ($SO_2$, PM), individually. Thus, the four prediction models were trained and tested. In these models, meteorological data were used as input parameters in the prediction of $SO_2$ and PM concentrations for a day later. Furthermore, the $SO_2$ and PM concentrations pertaining to the day the meteorological data was taken and were also used as inputs. In this way, the potential effect of the perpetuity of pollution in the atmosphere was also included in the models. For validation of the models, regression method was used with the predicted values as independent variables and observed values as dependent variables. At the Bahçelievler station, the determination coefficient ($R^2$) between observed and predicted values of $SO_2$ concentrations for the training data was 0.829

and for testing data was found to be 0.668. For PM concentrations, $R^2$ was 0.820 for the training set, and 0.808 for the testing set, respectively. ANOVA analysis was performed to check statistical relationship between the percentage of the variability between observed values and the neural network model predictions. The P-value in the ANOVA analysis was found to be less than 0.01, which indicated a statistically significant relationship between the variables at the 99% confidence level. Correlations between observed and predicted values were highly significant at the 0.01 level. The results obtained through the proposed models had shown that artificial neural networks can efficiently be used in the analysis and prediction of air quality (Tecer, 2007).

Mishra and Jha worked on the dust dissemination using suitable mathematical model and the results were validated through actual field data in Mahanadi Coalfields Area. Activity-wise dust generation potential were assessed and the study on distance versus dust concentration from a coal transportation road was carried out to assess normal impact zone of a dust generating source. Their results had shown that the dust generation potential was directly proportional to the speed of the vehicle. Results had also shown that at least 80% of the airborne dust generated by the haul trucks were larger than 10 micrometres. The majority of the dumpers or tipper drivers' exposure was attributable to loading and dumping activities with the haul roads and ramps themselves accounting for 10% to 15% of the total exposure in open cab and 5% to 10% in closed cab. A 30% to 35% reduction in dust concentrations measured in a sealed cab as compared to an open cab were realized for the haul road related activities cycles. The accuracy between field measurement values and the derived values from the model was found to vary between 85.6 and 99.0% which indicated fairly good accuracy. It was also observed from the study that during the modelling exercise the output is very much sensitive to the meteorological data. Hence, very accurate data was required for feeding in the model and very slight variation would resulted in wide variation of the output (Mishra & Jha, 2010).

### 2.3.7 Field Studies on dispersion modelling from computational fluid dynamics

In a comparative study by Riddle et al., simulation of the dynamics of the basic atmospheric boundary layer using the FLUENT CFD was done and the coding was also done for prediction of gas dispersion from a single stack. The Computational Fluid Dynamics (CFD) results were compared with the predictions from the Atmospheric Dispersion Modelling System (ADMS), a well tested and validated quasi-Gaussian model. During the study FLUENT was set up to simulate the neutrally stable atmospheric boundary layer and the mean velocity profiles were well predicted and were maintained with downwind distance. The algebraic Reynolds stress turbulence model provided the best predictions for the turbulence kinetic energy (TKE) and dissipation as per the earlier works. The dissipation rate was maintained throughout the length of the model domain and, on an average, the TKE levels were within 80% of the expected values up to a height of 100 m, but at the ground reduced to 50% of the inlet values. Predictions of TKE using the simpler k-ε model turbulence was much poorer. Spread of the gas plume were predicted using an advection-diffusion (AD) method, a Lagrangian particle tracking (LP) method and a large eddy simulation (LES) method. The LP method gave the best results; the horizontal and vertical plume spreads were similar to those predicted by ADMS and ground level and plume centre line concentrations were close to ADMS values. However, some differences were observed with the ground level concentrations rising more rapidly with distance than for ADMS, but reaching similar peak values while the plume centreline concentrations dropped more rapidly than in ADMS (Riddle et al., 2004).

It was concluded that the CFD simulations with the LP method were satisfactory; however, they could not be considered as an appropriate alternative to a model such as ADMS for normal atmospheric dispersion studies because of the much larger run times and the greater complexity of setting up model runs.

A study by Silvester et al. (2009) was done to assess dispersion phenomenon of pollutant emissions within the atmosphere is principally determined by the background wind systems characterized by the atmospheric boundary layer (ABL). This study was done using computational fluid dynamics (CFD) model to replicate the development of the internal ventilation regime within a surface quarry excavation due to the presence of a neutral ABL above this excavation. This model was then used to study the dispersion and deposition of fugitive mineral dust particles generated during rock blasting operations. The paths of the mineral particles were modelled using Lagrangian particle tracking (Silvester et al., 2009).

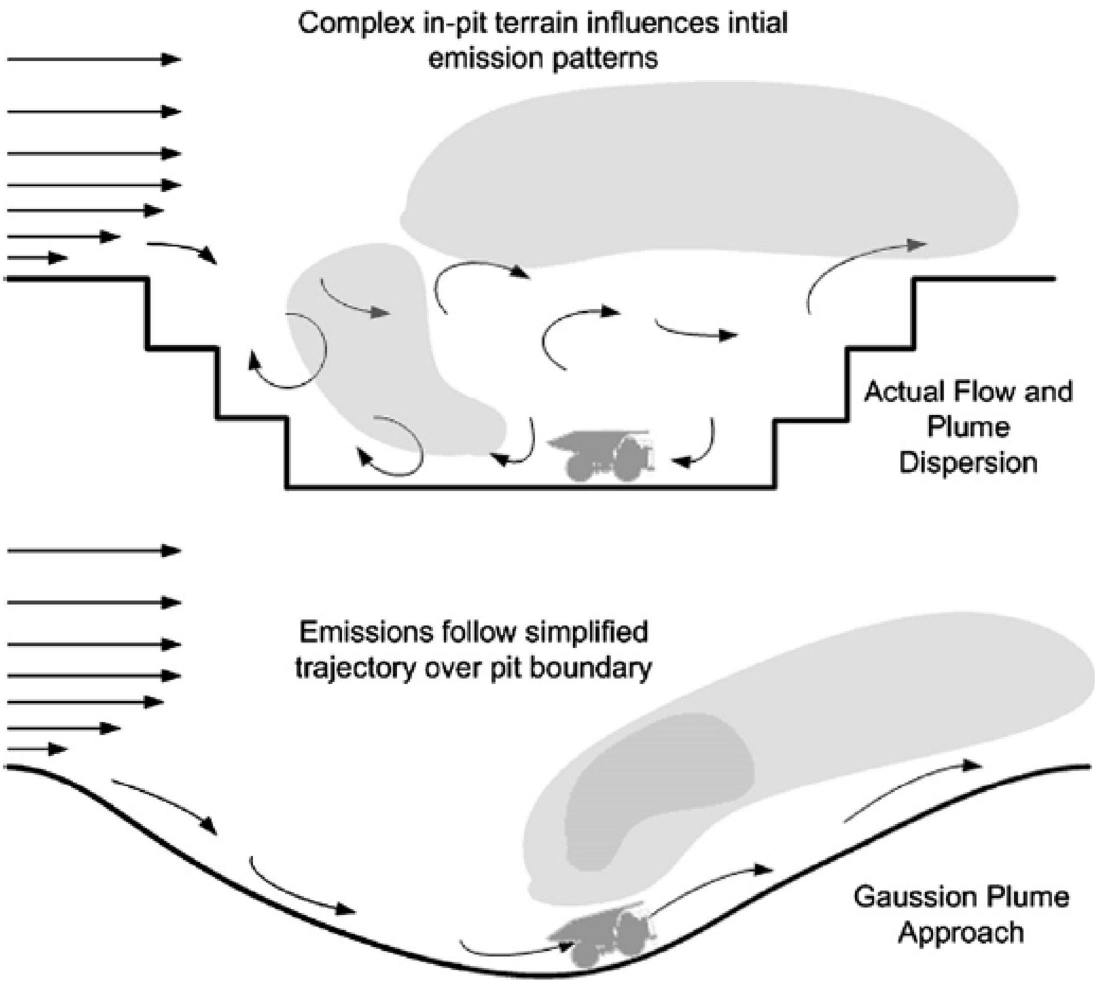


**Figure 2.4: A comparison of the actual dust dispersion within a mine with Gaussian plume and CFD models (After Lowndes et al., 2008)**

Gaussian plume models used for regulatory purposes were developed to predict downwind dispersion of dust from the sources over a flat or undulating terrain. These models could not account for the influence of the complex flow regimes that existed within quarries (Fig. 2.4). As fugitive dust emissions within a quarry are transported and dispersed by the local airflow within the quarry, there is a need to develop transport and deposition models that reproduce the local effects produced by these flows (Lowndes et al., 2008). This study had presented the modelling of the dispersion of a finite number of dust emission sources, created by the explosive blasting in an open-pit mine. Such a blast event generally produced a fragmented pile of rock and a cloud of fugitive dust that is dispersed by the interaction of the internal ventilation within the surface mine excavation and the external ABL. Dust clouds created on the bench blasting were generated as the result of material fracture, energy of release and the air volume displacement and translation due to the slumping of the fractured rock material to the ground. The formation and subsequent transport of the dust cloud was characterized by an initial forward motion that quickly attenuated to yield a large dust cloud. There were five representative bench blast sites selected across the excavation, to represent the range of potential extraction locations. The volume of each dust cloud was 100 m long by 60 m wide by 25 m high. The particle tracking simulations were conducted in a steady state flow field, where a volumetric emission source was generated from an array of 2000 near ground level injection points. The study had concluded that dust dispersion depended on the direction of the wind. A percentage of 30 to 60% of the emitted mineral particles were retained under neutral conditions within the quarry boundary. The study had demonstrated the important influence that emission location, the direction of the prevailing wind and the local in-pit ventilation flows can have on the level of near-field deposition that may be experienced from in-pit fugitive dust sources (Silvester et al., 2009).

## 2.4 Dust Dispersion Models used in Mining Industry

A large amount of research work has been done in the field of dust dispersion modelling in the past. Much of the research is focused on large-scale global or regional dispersion models. There were several mathematical models developed for prediction of dust dispersion as well as other air pollutants. But they are generalised for most of the industries. Some of the other models had been created for industry-specific purposes which have become a very useful tool for the management of dust and air pollutants in different industries.

There is a constant degradation of environment due to continuous generation of dust from all surface mining activities. The dust generated from an opencast mine is not only threat to the surrounding environment but there are several health implication to the human being and nearby flora and fauna. In this behalf, dust dispersion models have become a very useful tool to find out the concentration of different air pollutants for future work to be done in the related mining activities. This helps us in proper planning so that preventive measures can be taken to restrict the exposure to the human as well as to save the surrounding environment.

The use of unsuitable method for the modelling of dust or air pollutant over predicts the concentration, which can result in denial of air quality permits. Therefore, it is very important to opt a suitable modelling tool which can appropriately estimate the concentration of the dust (Reed, 2003).

### 2.4.1 Basic Mathematical Models

In most of the industries, modelling of pollutant dispersion is carried out using mathematical models. There are several basic mathematical models in use like the Box model, Gaussian model, Eulerian model, and Lagrangian model (Reed, 2005). Algorithms of these models are briefly discussed as follows:

#### 2.4.1.1 Box Model Algorithm

The box model is the simplest of the modelling algorithms. It has the assumption that the air shed is in the shape of a box. The air inside the box is assumed to have a homogeneous concentration. The box model is represented using the following equation 2.3:

$$\frac{dCV}{dt} = QA + uC_{in}WH - uCWH \qquad (2.3)$$

where

| | |
|---|---|
| $Q$ | = pollutant emission rate per unit area, |
| $C$ | = homogeneous species concentration within the air shed, |
| $V$ | = volume described by box, |
| $C_{in}$ | = species concentration entering the air shed, |
| $A$ | = horizontal area of the box (L*W), |
| $L$ | = length the box, |
| $W$ | = width of the bo,. |
| $u$ | = wind speed normal to the box, |
| $H$ | = mixing height, |

It assumes the pollutant is homogeneous across the air shed, and it is used to estimate average pollutant concentrations over a very large area. This mathematical model is very limited in its ability to predict dispersion of the pollutant over an air shed because of its inability to use spatial information (Collett & Oduyemi, 1997).

### 2.4.1.2 Gaussian Model Algorithm

The Gaussian models are the most common mathematical models used for air dispersion. They are based upon the assumption that the pollutant will disperse according to the normal statistical distribution. The Gaussian equation generally used for point-source emissions is as follows:

$$\chi = \frac{Q}{2\pi u_s \sigma_y \sigma_z}\left[exp\left\{-0.5\left(\frac{y}{\sigma_y}\right)^2\right\}\right]\left[exp\left\{-0.5\left(\frac{H}{\sigma_z}\right)^2\right\}\right] \qquad (2.4)$$

where

| | |
|---|---|
| $\chi$ | = hourly concentration at downwind distance x, |
| $Q$ | = pollutant emission rate, |
| $u_s$ | = mean wind speed at release height, |
| $\sigma_y$, $\sigma_z$ | = standard deviation of lateral and vertical concentration distribution, |
| $y$ | = crosswind distance from source to receptor, |
| $H$ | = stack height or emission source height. |

The terms $\sigma_y$ and $\sigma_z$ are the standard deviations of the horizontal and vertical Gaussian distributions that are used to represent the plume of the pollutant. These coefficients are based upon the atmospheric stability coefficients created by Pasquill and Gifford, and they generally increase with the increase in the downwind distance from the source (Pasquill, 1961 and Gifford & Skalarew, 1961).

### 2.4.1.3 Eulerian Model Algorithm

Eulerian models are developed by finding out a solution of a conservation of mass equation for a given pollutant. The equation generally is represented in the following form (Collett & Oduyemi, 1997):

$$\frac{\partial\langle c_i\rangle}{\partial t} = -\overline{U}.\nabla\langle c_i\rangle - \nabla\langle c_i' U'\rangle + D\nabla^2\langle c_i\rangle + \langle S_i\rangle \qquad (2.5)$$

where

$U$ $= \overline{U} + U'$,
$U$ = windfield vector U(x,y,z),
$\overline{U}$ = average wind field vector,
$U'$ = fluctuating wind field vector,
$c$ $= \langle c\rangle$ + c'
$c$ = pollutant concentration,
$\langle c\rangle$ = average pollutant concentration; $\langle\quad\rangle$denotes average,
$c'$ = fluctuating pollutant concentration,
$D'$ = molecular diffusivity,
$S_i$ = source term.

The wind field vector U, which is normally used, is considered turbulent and results in $\overline{U}$ and U', which are the components of turbulent wind field vector being used in equation 2.5. The turbulent wind shield vector also affects the pollutant concentration ***c*** in a similar manner with the terms $\langle c\rangle$ and c'. The term representing molecular diffusivity is neglected as the magnitude of this term is significantly small (Reed, 2005).

Equation 2.5 can be difficult to solve because the advection term $-\overline{U}.\nabla\langle c_i\rangle$ is hyperbolic, the turbulent diffusion term is parabolic, and the source term is generally defined by a set of differential equations. This type of equation can be computationally expensive to solve and requires some form of optimization in order to reduce the solution time required. Solutions have been achieved by reducing the problem to one and two dimensions rather than using three dimensions (Collett & Oduyemi, 1997).

### 2.4.1.4 Lagrangian Model Algorithm

Lagrangian models are used to predict pollutant dispersions based on a shifting reference grid. This shifting reference grid is generally based on the prevailing wind direction, or vector, or the general direction of the dust plume movement (Reed, 2005). The Lagrangian model has the following form as shown in equation 2.6:

$$\langle c(r,t)\rangle = \int_{-\infty}^{t}\int p(r,t|r',t')S(r',t')dr'dt' \qquad (2.6)$$

where

$\langle c(r,t)\rangle$ = 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'** (source) to location **r** at time **t**

The probability function must be derived as a function of the prevailing meteorology, which is appropriate for sources consisting of gases. If the source of emissions consisted of particles, then more information were to be incorporated in the function (such as the particle size distribution and the particle density) (Collett & Oduyemi, 1997).

This mathematical model has limitations when its results are compared with actual measurements. This is due to the dynamic nature of the model. Measurements are generally made at stationary points, while the model predicts pollutant concentrations based upon a moving reference grid. This makes it difficult to validate the model during initial use. The Lagrangian models are typically modified by adding an Eulerian reference grid to compensate this problem. This allows for the better comparison to actual measurements because it incorporates a static reference grid into the model (Collett & Oduyemi, 1997).

### 2.4.2 Surface Mine Models

There are several models available to estimate the dust concentration from different mining activities involved in surface mining and these models can also provide dust concentration emit from a surface mine as a single source. These models are developed by adapting several industrial air pollution model which are already there. They are modified to cater to suit the requisite parameters. The basis of all surface mine models are mathematical models which had already been discussed. Some of those surface mine models are as follows:

#### 2.4.2.1 Pit Retention Model

This model was developed by Cole & Fabrick in 1984. A study was mentioned in their report by Shearer stating that approximately one-third of the emissions from mining activities escape the open pit. This is a very simplistic model that is representative of the box model algorithm. This model calculates the mass fraction of the dust that escapes an open pit using the following mathematical model:

$$\varepsilon = \frac{1}{1 + \left(\frac{V_d}{K_z}\right) H} \tag{2.7}$$

where

$\varepsilon$ = mass fraction of dust that escapes an open pit,

$V_d$ = particle deposition velocity,

$K_z$ = vertical diffusivity, and

$H$ = pit depth.

Fabrick had also created an open-pit retention model which was based on the Gaussian algorithm and considered the wind velocity at the top of the pit. This model is given as (Cole and Fabrick, 1984):

$$\varepsilon = 1 - V_d \left[\frac{C}{u}\left\{\frac{1}{2} + ln\frac{w}{4}\right\}\right] \tag{2.8}$$

where

$\varepsilon$ = mass fraction of dust that escapes an open pit,

$V_d$ = particle deposition velocity,

$u$ = wind velocity at the top of the pit,

$C$ = empirical dimensionless constant equal to 7, and

$w$ = pit width.

The deposition velocity in both models was based on a gravitational settling velocity determined by Stoke's law.

### 2.4.2.2 Stockpile Emission Model

Gaussian dispersion models was modified by Pereira et al. in 1997 to find out the emission from stockpiles of an operating surface mine of Portugal. The equation is as follows for the estimation of emission from a stockpile (Pereira et al., 1997).

$$c = \frac{Q}{2\pi\sigma_y\sigma_z\bar{u}}\left[exp\left\{-0.5\left(\frac{y_r}{\sigma_y}\right)^2\right\}\right]\left[exp\left\{-0.5\left(\frac{h_e - z_r}{\sigma_z}\right)^2\right\}\right] \tag{2.9}$$

where

$c$ = pollutant concentration at location receptor ($x_r$, $y_r$, $z_r$) due to the emissions at source (0, 0, $h_e$),

$Q$ = emissions,

$\sigma_y$, $\sigma_z$ = horizontal and vertical standard deviations or dispersion coefficients, respectively,

$\bar{u}$ = average horizontal wind speed, and

$h_e$ = effective emissions height.

The modified version had taken into consideration the meteorological conditions, joint frequencies of occurrence of particular wind speed classes, wind direction sectors, and stability categories, the pollutant source characteristics, dimension of stockpile, density and size distribution of particles (Pereira et al., 1997).

The fugitive dust emission rates from stock piles is dependent on many factors like wind speed, turbulence, stock pile and shape, obstacles near the stockpile, median particle size and type, moisture content and surface character (Parrett, 1992). These factors are very difficult to calculate. Equation 3.7 was used to create risk maps of air quality for locations surrounding the mine site. No experimental validation was performed to determine the accuracy of these maps to actual conditions (Pereira et al., 1997).

### 2.4.2.3 ISC3 Model

This model was created by United States Environmental Protection Agency (USEPA/EPA) to predict pollutant dispersion from industrial facilities and is available as a computer program on their website. The pollutants for which it is designed include CO, $NO_x$, $SO_x$, Volatile organic compound (VOC), Pb and $PM_{10}$. Dust dispersion modelling for surface mining operations, as required for air quality protection, is generally carried out using an established model—the Industrial Source Complex model (ISC3) created by EPA. No other dust dispersion model has impacted the surface mining industry as much as the ISC3 model (Reed, 2005).

ISC3 model also has a module that can be used to model flat or complex terrain in an area. It has the ability to model dispersion from four types of emissions sources namely point (which are typically stacks), volume (which are typically buildings), area and open pit. In addition, the ISC3 model can calculate the deposition rates of $PM_{10}$ by using the deposition routine included in the model.

The ISC3 model is based on the Gaussian equation for point-source emissions, which is given as the following for the ISC3 model (EPA, 1995c):

$$\chi = \frac{QKVD}{2\pi u_s \sigma_y \sigma_z} exp\left\{-0.5\left(\frac{y}{\sigma_y}\right)^2\right\} \tag{2.10}$$

where

$Q$ = pollutant emission rate,

$K$ = scaling coefficient to convert calculated concentrations to desired,

$V$ = vertical term (dimensionless),

$D$ = decay term (dimensionless),

$u_s$ = mean wind speed at release height,

$\sigma_y$, $\sigma_z$ = standard deviation of lateral and vertical concentration distribution,

$\chi$ = hourly concentration at downwind distance '$x$', and

$y$ crosswind distance from source to receptor.

Equations for vertical term are as follows:

$$V = exp\left\{-0.5\left(\frac{z_r - h_e}{\sigma_z}\right)^2\right\} + exp\left\{-0.5\left(\frac{z_r + h_e}{\sigma_z}\right)^2\right\} + \sum_{i=1}^{\infty}\left[exp\left\{-0.5\left(\frac{H_1}{\sigma_z}\right)^2\right\} + exp\left\{-0.5\left(\frac{H_2}{\sigma_z}\right)^2\right\} + exp\left\{-0.5\left(\frac{H_3}{\sigma_z}\right)^2\right\} + exp\left\{-0.5\left(\frac{H_4}{\sigma_z}\right)^2\right\}\right] \quad (2.11)$$

where

$h_e$ $=h_s+\Delta h$,
$h_e$ = plume height,
$h_s$ = stack height,
$\Delta h$ = plume rise ,
$H_1$ $=z_r - (2iz_i - h_e)$,
$H_2$ $=z_r + (2iz_i - h_e)$,
$H_3$ $=z_r - (2iz_i + h_e)$,
$H_4$ $=z_r + (2iz_i - h_e)$,
$z_r$ = receptor height above ground, and
$z_i$ = mixing height.

Equation 2.11 was further modified to save the computational time without sacrificing accuracy which is as follows:

$$V = \frac{\sqrt{2\pi}\sigma_z}{z_i} \quad (2.12)$$

Equations to calculate decay term are as follows:

$$D = exp\left(-\varphi\frac{x}{u_s}\right) \qquad for\ \varphi > 0 \quad (2.13)$$

$$D = 1 \qquad for\ \varphi > 0 \quad (2.14)$$

where $\varphi$ is decay coefficient, which ca also be calculated as follows:

$$\varphi = \frac{0.693}{T_{1/2}} \quad (2.15)$$

where

$T_{1/2}$ = pollutant half-life,
$x$ = downwind distance, and
$u_s$ = mean wind speed at release height.

Generally, the default value of decay coefficient were taken as zero unless specified.

The ISC3 model is used to calculate the $PM_{10}$ concentration for receptor locations based on the Cartesian coordinate system where each source and receptor had X and Y coordinate. These coordinates were input into the downwind and crosswind distance equations. These equations calculated the downwind distance *x* and crosswind distance *y*, which were input to the equation 2.10.

ISC3 required the determination of the area surrounding any facility into rural or urban, thus establishing the set of horizontal and vertical dispersion curves (Pasquill-Gifford, 1961 for rural or McElroy-Pooler, 1968 for urban). There were no intermediate or other dispersion rates used. ISC3 used routine meteorological data to calculate the height of the well-mixed layer. For plumes rising less than the mixing-height, the plume was "trapped" and continued to mix within the layer by the use of the reflection concepts. For plumes rising above the mixing-height, the plums could no longer diffuse to the ground (Hanna et al., 2001).

The EPA stated in their study that there was a significant over prediction of $PM_{10}$ emissions from the surface coal mining operation by the ISC3 model (EPA 1995a). Another study from the EPA had figured out significant over prediction, by a factor of 2, at a single site where predicted and measured results were compared.

### 2.4.2.4 Dynamic Component Model

A study was completed on the ISC3 model using a theoretical rock quarry by Reed et al. in 2001 for the validation of the ISC3 model. This study has also concluded that the hauling operations had contributed the majority of $PM_{10}$ concentrations and that the haul truck emission factors could be part of the cause of the over prediction of $PM_{10}$ concentrations by the ISC3 model. However, further analysis of the data provided by the Cole and Zapert study had presented another hypothesis explaining the cause of the ISC3's over prediction. This hypothesis had stated that since the majority of the sources producing $PM_{10}$ at surface mining operations were moving or mobile sources. The ISC3 model could not accurately predict dust concentrations from the observed mining operations, because it was a model designed for predicting the dust dispersion from the stationary sources (Cole & Zapert, 1995). This led to further investigations on dust dispersion modelling at surface mining operations, focusing on modelling the dispersion of dust generated from the haul trucks. Dynamic Component program was specifically developed to allow modelling of dust dispersion from haul trucks. The Dynamic Component model had used the same approach as the ISC3 model, where the dispersion of particulate matter from a point source was represented by the Gaussian equation (Reed, 2003 & Reed et al., 2005):

$$\chi = \frac{QK}{2\pi u w_s \sigma_y \sigma_z} exp\left\{-0.5\left(\frac{y}{\sigma_y}\right)^2\right\} \tag{2.16}$$

where

$\chi$ = hourly concentration at downwind distance '$x$',

$Q$ = pollutant emission rate,

$K$ = conversion factor for $\chi$ and $Q$,

$w_s$ = wind speed,

$\sigma_y$, $\sigma_z$ = standard deviation of lateral and vertical concentration distribution, and
$y$ = the distance to monitoring location, measured perpendicular to wind distance.

The emission rate $Q$ for $PM_{10}$ can be calculated for haul trucks using the emissions factor which is represented as:

$$Q = \frac{2.6\left(\frac{s}{12}\right)^{0.8}\left(\frac{W}{3}\right)^{0.4}}{\left(\frac{M}{0.2}\right)^{0.3}} \tag{2.17}$$

where
$Q$ = the emissions from a haul truck,
$s$ = the surface material silt content,
$W$ = the mean vehicle weight, and
$M$ = the surface material moisture content.

There were two field studies carried out at different surface mine locations, one at stone quarry and another one at coal mine to validate the results of the Dynamic Component Program with actual measurements. The results of the field studies had shown that the Dynamic Component Program was an 85% improvement over the ISC3 model in predicting dust dispersion from haul trucks. However, the Dynamic Component Program had a limitation that it could only predict dust dispersion of haul trucks travelling in a straight line. More work was required to perform modelling on haul trucks travelling on the haul roads that contained curves and to continue validating the model to ensure its accuracy (Reed, 2003).

## 2.4.2.5 ADMS Model

Atmospheric Dispersion Modelling System (ADMS) was developed by Cambridge Environmental Research Consultants (CERC) to make use of the most up to date understanding of the behaviour of the lower levels of the atmosphere in easy to use computer modelling systems for atmospheric emissions. There are several models of ADMS available for modelling air pollutants from the different kind of industries as well as the different kind of activities.

ADMS is 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 which is using two parameters, namely boundary layer and Monin - Obhukhov length to describe the atmospheric boundary layer and also utilised a skewed Gaussian concentration distribution to calculate the dispersion under convective conditions (CERC, 2007). The equation is as follows for the determination of concentration of air pollutant (CERC, 2012):

$$C = \frac{Q}{2\pi\sigma_y\sigma_z U} e^{-y^2/2\sigma_y^2} \times \left\{ e^{-(z-z_s)^2/2\sigma_z^2} + e^{-(z+z_s)^2/2\sigma_z^2} + e^{-(z+2h-z_s)^2/2\sigma_z^2} + e^{-(z-2h+z_s)^2/2\sigma_z^2} + e^{-(z-2h-z_s)^2/2\sigma_z^2} \right\} \quad (2.18)$$

where,

$C$ = hourly concentration,
$Q$ = pollutant emission rate,
$\sigma_y$, $\sigma_z$ = lateral and vertical dispersion parameter,
$U$ = wind speed at release height,
$y$ = crosswind distance from source to receptor,
$z$ = height of the receptor,
$z_s$ = height of the source, and
$h$ = atmospheric boundary layer height.

The vertical dispersion parameter ($\sigma_z$) at the mean height of plume ($z_m$) is linked directly to the vertical component of the turbulence ($\sigma_m$) and the travel time from the source ($t$) by the relationship:

$$\sigma_z = \sigma_w t \left( \frac{1}{b^2} + \frac{N^2 t^2}{1 + 2Nt} \right)^{-1/2} \quad (2.19)$$

where

$N$ = buoyancy frequency
$\sigma_w$ = root-mean-square vertical turbulent velocity at $z_m$, and
$b$ $= z_s/h$.

The factor $b$ has ensured a smooth transition between the solution from the surface releases to elevated releases in the equation 2.19.

The lateral or transverse dispersion parameter ($\sigma_y$)is given by the following equation:

$$\sigma_y^2 = \sigma_{yt}^2 + \sigma_{yw}^2 \quad (2.20)$$

The spread due to turbulence ($\sigma_{yt}$) is represented as follows:

$$\sigma_{yt} = \sigma_v t \left( 1 + \sqrt[3]{15.6} \frac{u_* t}{h} \right)^{-1/2} \quad (2.21)$$

where,

$\sigma_v$ = root-mean-square vertical turbulent velocity, and
$u_*$ = friction velocity at the Earth's surface.

The spread due to variations in the mean wind direction ($\sigma_{yt}$) is equal to $\sigma_\Theta x$. Parameter $\sigma_\Theta$ is the standard deviation of the mean wind direction. This may either be specified as a meteorological input parameter in degrees, or estimated by the meteorological data pre-processor using the following expression (CERC, 2012):

$$\sigma_\theta = 0.065\sqrt{\frac{7T}{U_{10}}} \qquad (2.22)$$

where,

$T$ = averaging time in hours, and

$U_{10}$ = mean wind speed at height of 10m.

A study was done by Neshuku in Rossing open pit Uranium mine. A comparison between AERMOD and ADMS was done in this study using various statistical measures. The study concluded that ADMS was superior to AERMOD for calm conditions (when the wind speed is less than 1 m/s). Performance of the AERMOD had improved but not up to the acceptable limit even after excluding the calm conditions from the input. ADMS was considered best modelling tool for prolonged calm conditions (Neshuku, 2012).

Another study had been done in the New Delhi for the assessment of the total suspended particulate matter (TSPM) from ADMS-Urban as well as AERMOD. Comparative study was also made and suggested than ADMS-Urban had shown greater tendency towards under prediction of TSPM concentration than those of AERMOD. It was also revealed in the study due to use of more number of meteorological data in AERMOD, it performs slightly better than ADMS-Urban (Mohan et al., 2011).

#### 2.4.2.6 AERMOD Model

American Meteorological Society (AMS) and U.S. Environmental Protection Agency (EPA) Regulatory Model (AERMOD) was one of the latest modelling tool available for the dispersion modelling in the mining industry. This model was the latest software after ISC3 from USEPA. In this model boundary layer is characterised with computation of the Monin–Obukhov length, surface friction velocity, surface roughness length, sensible heat flux, convective scaling velocity and both the shear and convection driven mixing heights. These parameters are used in conjunction with the meteorological measurements to characterize the vertical structure of the wind, temperature, and turbulence (Cimorelli et al., 2004). AERMOD's has a meteorological pre-processor AERMET in which surface characteristics, cloud cover, a morning upper-air temperature sounding and one near-surface measurement of wind speed, wind direction, and temperature are taken as an input. With these input, AERMET is used to calculate the boundary layer parameters like friction velocity, Monin–Obukhov length, convective velocity scale, temperature scale, mixing height and surface heat flux.

AERMOD simulates a plume, in elevated terrain, as a weighted sum of concentrations from two limiting states: a horizontal plume (terrain impacting) and a

terrain following plume. Each plume state is weighted using the concepts of the critical dividing streamline and
a receptor specific terrain height scale (Venkatram et al. 2001; Cimorelli et al. 2004).

The general equation form of the AERMOD dispersion model with terrain is as follows:

$$C_T\{x_r, y_r, z_r\} = fC_{c,s}\{x_r, y_r, z_r\} + (1-f)C_{c,s}\{x_r, y_r, z_p\} \quad (2.23)$$

where,

$C_T\{x_r, y_r, z_r\}$ = the total concentration,

$C_{c,s}\{x_r, y_r, z_r\}$ = the contribution from the horizontal plume (subscripts $c$ and $s$ refer to convective and stable conditions, respectively),

$C_{c,s}\{x_r, y_r, z_p\}$ = the contribution from the terrain-following plume,

$f$ = the weighting factor,

$\{x_r, y_r, z_r\}$ = the receptor coordinate,

$z_p$ $= z_r - z_t$, which is the receptor height above local ground, and

$z_t$ = the local terrain height.

There are different equations used to calculate the concentration of the air pollutant in the atmosphere based on the atmospheric conditions. This can be either stable or convective in nature. The decision is made based on the values of friction velocity ($u_*$) and Monin- Obukhov length ($L$) with main boundary layer parameters. These can be calculated as follows:

$$u_* = \frac{ku_{ref}}{\ln(z_{ref}/z_0) - \psi_m(z_{ref}/L) + \psi_m(z_0/L)} \quad (2.24)$$

$$L = -\frac{\rho c_p T_{ref} u_*^3}{kgH} \quad (2.25)$$

where,

$k$ = the von Kármán constant,

$u_{ref}$ = the wind speed at reference height,

$z_{ref}$ = the lowest surface layer measurement height for wind,

$z_0$ = the roughness length,

$\psi_m$ = defined for the CBL by Panofsky and Dutton (1984) and for the SBL and by Van Ulden and Holtslag (1985)and braces are used throughout to denote the functional form of variables,

$\rho$ = the density,

$c_p$ = the specific heat at constant pressure,

$T_{ref}$ = the ambient temperature (K) that is representative of the surface layer,

$g$ = the acceleration of gravity, and

$H$ = sensible heat flux.

Assuming neutral conditions, $u_*$ and $L$ are initialised using equations 3.22 and 3.23 and final values of these parameters are found by iterating the above equations until convergence (Wyngaard, 1988).

During stable conditions (i.e. stable and neutral stratifications when $L > 0$), AERMOD has utilised the following equations for the estimation of the concentration of air pollutant (Cimorelli et al., 2004):

$$C_s(x_r, y_r, z_r) = \frac{Q}{\sqrt{2\pi}\tilde{u}\sigma_{zs}} F_y \times \sum_{m=-\infty}^{\infty} \langle exp\left[-\frac{\left(z - h_{es} - 2mz_{ieff}\right)^2}{2\sigma_{zs}^2}\right] + exp\left[-\frac{\left(z + h_{es} + 2mz_{ieff}\right)^2}{2\sigma_{zs}^2}\right]\rangle \tag{2.26}$$

where,

$F_y$ = the lateral distribution function,

$\tilde{u}$ = effective wind speed,

$z_s$ = the total vertical dispersion,

$h_{es}$ = the plume height, and

$z_{ieff}$ = the effective mechanical mixing height.

AERMOD is used to predict the total concentration ($C_c$) in the CBL (i.e., convective and neutral stratifications when $L < 0$) by summing the contribution from the three sources. For the horizontal plume state, the equation is as follows (Cimorelli et al., 2004):

$$C_c(x_r, y_r, z_r) = C_d(x_r, y_r, z_r) + C_r(x_r, y_r, z_r) + C_p(x_r, y_r, z_r) \tag{2.27}$$

Where,

$C_d(x_r, y_r, z_r)$ = the direct source contribution to concentrations in the CBL,

$C_r(x_r, y_r, z_r)$ = the indirect source contribution to concentrations in the CBL, and

$C_p(x_r, y_r, z_r)$ = the penetrated source contribution to concentrations in the CBL.

**Concluding remark:** Based on the literature survey, AERMOD was found to be a suitable modelling tool for the predication of the dispersion of air pollutant. It has been used on a very large scale in mining industry. Hence, AERMOD would be used as a modelling tool for this work. Before the use of AERMOD for main study, it was also validated for the Indian geo - mining conditions using a mine 'A'.

AERMOD is acronym for the American Meteorological Society/Environmental Protection Agency Regulatory Model. It has been deployed universally for the prediction of the generated particulate matter from different industrial activities, especially for the prediction of the dust generation from the mining activities (Cimorelli et. al., 2004).