ABSTRACT

Title of Document: ENVIRONMENTAL COSTS FOR HIGHWAY ALIGNMENT EVALUATION

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Directed By: Professor Paul Schonfeld, Department of Civil and Environmental Engineering

The regulatory measures have set standard to be met for evaluating environmental cost of a proposed highway. However, these measures do not consider the health effects of increased concentrations of pollutants. This thesis seeks to develop a methodology for estimating the environmental cost of a new highway with a specified alignment. The proposed methodology for estimating the environmental health costs of a highway quantifies the social cost of the emission impacts. An example of a proposed highway parallel to an existing highway is considered in a rural area. The environmental costs consider emissions at the source, dispersion of particles, and population exposure. The total emissions of nitrogen dioxide are estimated using the vehicle-specific approach and the transport of these emissions is estimated with a Gaussian model for pollutant dispersion. The chronic respiratory diseases, asthma,

and cardiovascular cases resulting from the dispersion of pollutant are estimated using the concentration exposure relationship. The results are analyzed for factors that influence the effects of emissions, i.e. vehicle volume, vehicle mix, wind direction, wind speed, meteorological conditions, gradient and population density.

ENVIRONMENTAL COSTS FOR HIGHWAY ALIGNMENT EVALUATION

By

Ammara Asif

Thesis submitted to the Faculty of the Graduate School of the
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Advisory Committee:
Professor Paul Schonfeld, Chair
Professor Elise Miller-Hooks
Associate Professor Cinzia Cirillo

Dedication

My work is dedicated to the people suffering from respiratory problems and to my parents.

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I would like to thank Professor Schonfeld for his guidance, instructions and support. Without his generous support and flexibility, this work would not have been finished on time. He reviewed my work as quickly as he could so I can get more time to work. He polished my amateur work in first draft. I am very grateful to you for helping me from very beginning to the end.

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Chapter 1: Introduction

1.1 Background

The approval for construction of a proposed highway project requires various evaluations through benefit cost analysis in which several costs and benefits are calculated based on certain assumptions. The feasibility report consists of the costs and benefits that are needed to assess highway selection for a region. The costs and benefits are calculated in monetary terms that enable the evaluation of aspects such as congestion, travel time, public health, wildlife and surrounding ecology. The construction cost, user cost or travel time cost, change in travel behavior of the commuters, travel demand forecasting, environmental impacts assessment are the costs that are usually considered in evaluating feasibility of a new highway in an area. Design and construction costs are estimated based on the efficiency and work hours dedicated by the workforce to carry out the task. User benefit or cost reduction is the travel time saving of commuters that results from the proposed highway. Change in travel behavior is an estimate of how commuters are diverted from existing highways to the new highway and how that will affect the existing transportation system for the proposed highway in the given system. Environmental impacts are estimated for a proposed highway to assess the harmful effects of pollutants concentrations on the surrounding life. The EPA guidelines for environmental impact assessment use certain threshold values of criteria pollutants to regulate the concentration of air pollutants in an area. However, this impact assessment method relies on the set standards for individual air pollutants and does not consider the dispersion of air pollutants in the surrounding area and their direct impact on people's health. Long exposure to the air pollutants pose negative impacts on human health in the form of various respiratory problems, cardio-vascular and neurological disorders As of 2004, in US approximately 67000 and 4000 deaths of adults and children under age of 5 were attributable to outdoor pollution. For 2008, approximately 1.34 million premature deaths were attributable to outdoor air pollution in cities around the world. (WHO 2011). Given the health effects of air pollutants the dispersion across the area it is equally important to consider these effects in the evaluation process of a highway project.

1.2 Problem

The current practice of environmental impact assessment checks whether the standards for concentration of each criteria pollutant are met. Criteria pollutants are the pollutants that are regulated by the US EPA. Criteria pollutants are: ground level ozone (which contributes to smog formation), oxides of nitrogen, carbon monoxide, sulfur dioxide, lead, and particulate matter. Particulate matter is regulated in two size categories: (1) particles with an aerodynamic diameter not exceeding 2.5 micrometers (PM2.5); and (2) particles with an aerodynamic diameter not exceeding 10 micrometers (PM10, which includes PM2.5) (EPA 2006). Although the source concentrations are essential for determining the effects of emissions, the pollutants' dispersion and behavior are equally important in predicting the exposure risks. The proposed methodology includes these components, i.e. dispersion of air pollutants and exposure assessment, in the evaluation stage. The problem consists of four parts: 1) calculating the emissions (factors to consider); 2) area consideration (dispersion in area or isolation); 3) risk assessment; and 3) determining the monetary values for the environmental costs.

Although models such as AERMOD (EPA 2012) are available for dispersion modeling, they are used for point sources i.e. industries rather than for traffic emissions. Dispersion of pollutants depends on a number of factors, including chemical and physical characteristics of the pollutants, meteorological conditions such as atmospheric stability, wind speed and direction, and ambient temperature, topography, distance from the source (Hallmark 2004). Presence of industries and the existing road network in the surroundings also contributes to the air pollutant concentration in an area. This is one of the reasons why calculating dispersed concentrations for one candidate highway becomes difficult. After being emitted, carbon dioxide, carbon monoxide, sulfur dioxide, and nitrogen dioxide gases disperse into the air. On reaction with air these gases form secondary pollutants, namely nitric acid, sulfuric acid, and ozone. Formation of secondary pollutants is the result of a photo-chemical reaction which affects the environment in the form of acid rain and the humans in the form of frequent respiratory and heart problems. The dose-exposure assessment based on risk assessment is made for point source emissions. The risk assessment approach can also be adopted for mobile sources i.e. traffic emissions. The assessment of the risks of the air pollutants on the surrounding life is important because of the number of related deaths and health problems reported every year. These health risks are assessed in this study using the dose-exposure assessment and then converted to the probability of having health problems. A crucial part of determining the environmental cost is the monetary value attributed to human life or human health.

The objective of the research is to formulate a methodology that not only estimates the impacts of air pollutant concentration but also integrates the immediate effects of the dispersion of air pollutants on human life during the evaluation process of a candidate highway. The environmental costs of vehicle emission concentration, their dispersion and the direct risks involved due to these pollutants on human life, across the area, will also be calculated for a candidate highway.

1.3 Scope

This study focuses on the development and demonstration of a methodology for estimating the costs related to the environment. The impacts of chronic exposure to nitrogen dioxide (NO_2) are estimated in this study. Other factors such as additional pollutants from the neighboring highway and deposition of pollutants are also incorporated in estimating emission rate and pollutant dispersed concentration. These impacts are estimated for a given traffic mix and atmospheric condition.

1.4 Thesis Organization

This thesis is organized as follows:

Chapter 1 provides an introduction of the problem and scope of the research work. Chapter 2 reviews the literature relevant to this research. Chapter 3 describes the assumptions and methodology proposed for evaluating highway projects. Chapter 4 discusses the data and results of analysis. Chapter 5 draws conclusions and recommends extensions.

Chapter 2: Literature Review

In order to study the evaluation of highways from an environmental perspective the literature is reviewed that include regulatory requirements for highways that are related to emissions, air pollutants' dispersion, impacts of dispersion, and their social costs. The information on these topics is collected from books, journals, and the internet.

2.1 Transportation Conformity

The transportation-related air emissions are regulated by Office of Transportation and Air Quality, OTAQ (US EPA). Transportation Conformity is a requirement under Clean Air Act (EPA 2012). The Clean Air Act (CAA) requires that, "in areas experiencing air quality problems, transportation planning must be consistent with air quality goals". In the places where air quality goals are not being met, the state and local transportation officials face the challenges of finding alternatives for reducing emissions from transportation sources. Failure in meeting the standards requires implementing Air Quality Management that uses a State Implementation Plan, SIP. This implementation plan guides transportation agencies on how to mitigate the situation through Transportation Control Measures that define Reasonably Available Control Measures (RACM). These control measures are developed to achieve the air quality and mobility goals simultaneously. The RACM suggests taking steps such as, changing travel patterns, reducing the number of single-occupant vehicles, and encouraging useful alternative modes of transportation (such as transit and bicycles) as an increasingly important part of the transportation network (Transportation Conformity 2006) (FHWA 2010).

An important factor in this regulation is that the Air Quality Management is required in the non-attainment areas and Environmental Impact Analysis is required for the attainment areas. Attainment areas are those areas where national air quality standards have been met and non- attainment areas are those that have not met the standards. Regardless of the area designation, the EPA-approved model MOVES 2010 is used to estimate the emission concentrations from highway vehicles. It is used for project-level impact assessment. The highway emissions estimated with this model are those at the source.

This model does not estimate the effects of individual highway emission on the population. Instead, it uses threshold values based on estimated effects. However, most of the air pollutants disperse into the air and after a period of time disappear; that depends on the chemistry of the pollutants being emitted. Fine particles of sulfates, nitrates, organic compounds, and metal can travel hundreds and thousands of kilometers and can remain present for several days or weeks (Wilson 1995). This indicates that the highway emissions might be related with illnesses. There are many studies that found relation between the distance from the highway and frequency of illnesses in the nearby area. For instance, a study conducted in California (Kim, et al. 2008) assessed exposure with several measures of residential proximity to traffic that were calculated using geographic information systems, including traffic within a given radius and distance to major roads. The study measured Oxides of nitrogen and nitrogen dioxide correlations with household and traffic levels. The findings of the study showed that the asthma cases in children were double for the households living in the high-exposure area. The study also found that the households living within 75 meters from a highway or freeway are more prone to asthma episodes. In a similar study, the percentage change in pollutant concentration is found linked with health indicators such as night time asthma, wheeze-cough, slow playing etc (O’Connor, et al. 2008).

In another study (Zhou and Levy 2007), the researchers found the spatial extent of the pollutants. This study found the spatial extent of pollutants emitted from mobile sources may be 100 to 400 meters for elemental carbon, 200 to 500 meters for nitrogen dioxide, and 100 to 300 meters for ultrafine particles. The study also suggested using the population exposure to find the zone of influence of mobile sources for benefit cost analysis. These studies suggest that the dispersion of the pollutants and population exposure should be considered in the evaluation process of the highway. However, these two components are not present in the current evaluation process for highways. Therefore, it is necessary to develop a methodology for traffic-related emission assessment that combines the pollutants’ dispersion and population exposure. The dispersion models are available for non-attainment areas for point and non-point sources; and the exposure assessment is done for point sources and in epidemiological studies for public health. Integrating dispersion of emissions with exposure assessment enables analyzing the impacts that are being overlooked in the impact assessment. Therefore, the air dispersion and exposure modeling approaches used in different areas of studies are reviewed to exemplify the methodology.

2.2 Air Dispersion Models

Many models are approved in an EPA list for modeling dispersion of point sources (US EPA 2012). AERMOD, CAL3QHC, and CALINE-4 are the most preferred models for dispersion of point and non-point sources. Modeling along a road is affected by the area and source assumption. Various models are available for describing air pollutant diffusion with variation in these assumptions, such as area source models, elevated point source models, a street canyon sub-models, and highway sub-models (Aalst, et al. 1998). A crucial part of the modeling process is the evaluation of a given mathematical model that describes a system accurately. There are three approaches for modeling the dispersion of air pollutants (Moreira and Vilhena 2010): a) the Eulerian Approach, b) the Lagrangian Approach, and c) the Large Eddy Simulation using Computational Fluid Dynamics.

The Eulerian approach is based on the conservation of mass and can incorporate the numerous second and higher order chemical kinetic equations necessary to describe photochemical smog generation. The two types of Eulerian models are the Box and Gaussian models. The Lagrangian stochastic dispersion models numerically simulate particles to account for flow and turbulence space-time variations. Lagrangian models can successfully describe the turbulent dispersion of passive contaminants because they address the important aspects of turbulence, but they are limited to a simplified set of reacting species. Large Eddy Simulation (LES) is a mathematical model for turbulence used in Computational Fluid Dynamics, CFD. The CFD simulates the energy-containing eddies and deterministically models the rest of the particles.

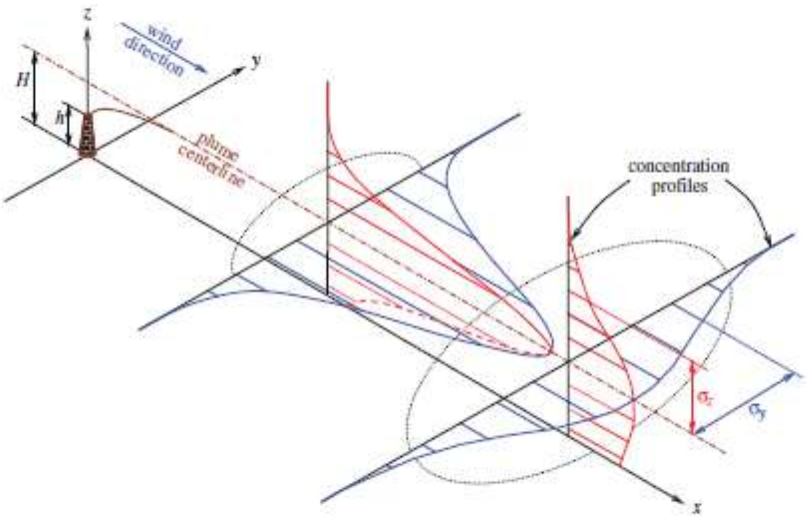

Figure 1 Gaussian Model Parameters

Source: Stockie, John M. "The Mathematics of Atmospheric Dispersion Modeling." *Society of Industrial and Applied Mathematics, SIAM* 53, no. 2 (2011): 349-372.

The computational fluid dynamics CFD may be useful for modeling dispersion where the transport of pollutants is complex and difficult to model using an Eulerian or stochastic approach. The Lagrangian modeling approach simulates particles in time-space. It models transport of particles without regard to others and is useful for less reactive or non-reactive pollutants. The box model does not consider the fluctuations that may occur due to wind direction or speed and it determines the concentration as an average for an air shed considered. Therefore, if the intention is to find population exposure, this may not be the best model to use. The Gaussian models use a normal probability distribution which is converted into other forms of distributions depending on the case scenario and conditions. That is why there are different modified forms of this model available for different cases.

The assumptions for the Gaussian model are as follows: Steady state conditions are considered (i.e. $\partial c/\partial t= 0$), vertical velocity component is neglected as it is very small (i.e. v = 0), removal of pollutants is neglected i.e. R=0), wind speed and eddy diffusivity coefficients are assumed constant.

"The Gaussian solution in a system of coordinates where x is along the wind direction, y is transversal to wind, z is the height and the source of intensity is located at (0,0,H), shown in Figure 1, is given by the following equation

$$C(x,y,z) = \frac{Q}{2\pi u\sigma_y\sigma_z} e^{(-\frac{y^2}{2\sigma^2{}_y})}[e^{(-\frac{(z-H)^2}{2\sigma_z{}^2})} + e^{(-\frac{(z+H)^2}{2\sigma_z{}^2})}] \qquad \text{Eq. 1}$$

with boundary conditions C = 0, $|y|$, $z\rightarrow \infty$, $-Kz\ \partial C/\partial z = 0$, z = 0, C (0,y,z) = 0 (deleted neighborhood)" (Moreira and Vilhena 2010). Here, C is the concentration, Q is the source pollution emission rate in g/s, u is the horizontal wind velocity along the plume center line, meters/second, H is height of emission plume centerline above ground level, meters, σ_y, σ_z are horizontal and vertical standard deviation of the emission distribution, meters, z is the height of surface layer, and Kz is the vertical eddy diffusion coefficients. However, this model is intended for point sources and in using this model to predict the concentration from a ground level source for line sources, the height of emissions is assumed to be zero. (Stockie 2011, Moreira and Vilhena 2010). If a straight long road that runs perpendicular to the wind direction is approximated by a linear source of infinite length along y-axis, the boundary condition changes to $c(0,y,z)=Q_L/u\ \delta(z)$. Q_L is a constant emission rate per unit length of road, and is distinct from the emission rate parameter Q. The solution for this boundary condition is

$$C(x,y,z) = \frac{2 * Q}{\sqrt{2\pi} * \sigma_z * u} * \exp\{-\frac{1}{2} * \left(\frac{z}{\sigma_z}\right)^2\} \qquad \text{Eq. 2}$$

This model has been modified for an infinite line source by omitting the exponent component hence forming the formula that is used in CALINE, the EPA regulatory dispersion model.

The calculation of sigmas is the step that distinguishes many modifications of Gaussian Model. The Pasquill-Gifford approach for calculating sigma (Gifford 1957) uses atmospheric stability classes that are defined by wind velocity at the ground, insulation and at night cloud cover. There are many other methods that modify the sigma calculation based on Pasquill-Gifford approach such as Brookhaven sigmas and Brigg's sigmas. Brigg's sigmas feature diffusion in open country and urban environments.

Although, the Gaussian model discussed above is very useful, modified and more accurate forms of this model have been developed recently. The formulation of the basic Gaussian model is modified for wind direction by many researchers such as (Hanna 1990), (Sienfeld 1986), (Yamartino 2008), (Luhar and Patil 1989), (Esplin 1995), and (Venkatram and Horst 2006). (Hanna 1990) uses the polar coordinate system to estimate the emission from a point source. In this study, the lateral dispersion parameter sigma y is analyzed under light-wind stable atmospheric conditions. If this condition is changed from stable to unstable condition the results might be erroneous. The use of polar coordinates makes it a less preferable model; however the approach results are useful. (Venkatram and Horst 2006) discuss the variation of Gaussian Model for the receptors of air pollutant concentration in an area. In this research the Gaussian model is modified, the error in the emission is predicted for wind direction, and this error is compared with errors in other previous models developed by (Luhar and Patil 1989), and (Esplin 1995).

This approximation is based on the receptor based approach. The results vary with the direction of wind and the considered affected entity. The approximation is found to have sufficient error reduction. This model is later improved by Briant et al (2011) and others, to reduce the measurement error for more accurate concentrations.

(Briant, Korsakissok and Seigneur 2011) develop an improved model that takes into account various wind directions which were previously approximated. The HV approximation of the Gaussian model is improved. This approximation quantifies the error in dispersion caused by wind direction and then finds the corrected approximation of the analytical formulae. The model considers different angles of wind that can affect the dispersion of the pollutants for different stability classes, emission heights, near-road measurements of wet and dry deposition flux, and computational times. This model offers accuracy and "is claimed" to be applicable to road networks, but with more computational effort. The model has not been demonstrated for its applications. The accuracy is not a big issue, considering the objective. This model provides more accuracy than the HV approximation. However, the HV approximation is also adequate in approximating the dispersion as perceived by the receptor (resident, monitoring station, etc) located at any angle other than 90°, shown in Figure 2 . The HV approximation requires less computational effort with appropriate accuracy. The effective downwind distances are obtained from the source highway, expressed in Eq. 3. The model uses the sigma y and z calculated by the distances shown in Eq. 4. These distances are measured from the start and end points of the highway.

$$X_{eff} = X_r / \cos\theta \qquad \text{Eq. 3}$$

$$d_i = (x - x_i)\cos\theta + (y - y_i)\sin\theta \qquad \text{Eq. 4}$$

where X_{eff} is the effective downwind distance, X_r is the x-coordinate of receptor, x and y are the coordinates of the receptor, x_i and y_i the coordinates of the source extremities i (with i= 1 or 2), d_i is the downwind distance from the extremities, and θ is the angle between the normal to the line source and the wind direction. The positions of highway and receptors are explained in Figure 2.

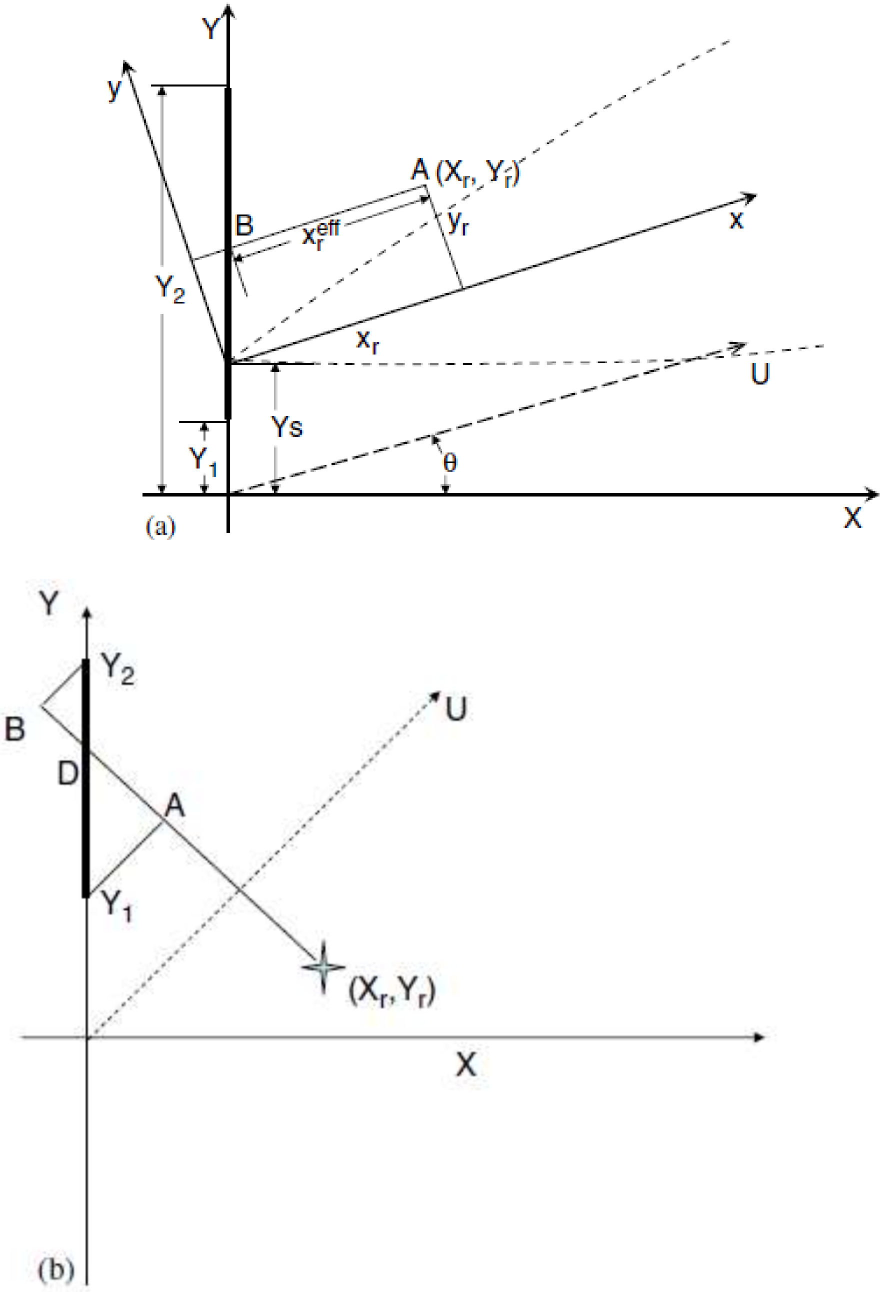

Figure 2 a) Co-ordinate Systems Used to Calculate Contribution of Point Source at Y_s to Concentration at ($X_{r,}$ Y_r). The System X–Y as the X-Axis along the Mean Wind Direction, which is at an Angle Y to the Fixed X-Axis. (B) The Line Source is Y_1Y_2. The Segment DY_2 is Downwind of the Receptor at (X_r, Y_r).

Source: Venkatram, A. and Horst, T.W. Approximating Dispersion from a Finite Line Source, Atmospheric Environment, Vol. 40, pp. 2401–2408, 2006.

This model is given by

$$C(x,y,z) = \frac{Q}{2\sqrt{2\pi}\, ucos\,\theta\sigma_{z\ (deff)}} exp\left(-\frac{z^2}{{\sigma_z}^2{}_{(deff)}}\right) * \left[\left(erf\,\frac{(y-y_1)cos\theta - x\,sin\,\theta\}}{\sqrt{2}\,\sigma_{y(d_1)}}\right) - \left(erf\,\frac{(y-y_2)cos\theta - x\,sin\,\theta\}}{\sqrt{2}\,\sigma_{y(d_2)}}\right)\right] \quad \text{Eq. 5}$$

This model is considered more accurate than the basic Gaussian model for predicting the diffusion of air pollutant particles in the surrounding area. The dispersed concentrations obtained from this model will be used in calculating the associated health risks. The dispersed concentrations are estimated using the emission rate that is obtained from ground level emissions. The ground level emissions' are not estimated using deposition. Deposition is another important parameter that directly affects the life of humans, plants and animal. The deposition refers to removal of air pollutant from the environment.

Deposition (US EPA 2012) includes two types: wet deposition and dry deposition. Wet deposition is a process in which the transport of pollutants or chemicals in the air occurs in a wet environment i.e. during natural processes of rain, snow, fog, or mist.

These pollutants mix with water and form acidic rain which then hits the ground and affects the plants and animals it comes in contact with and penetrates through. Dry deposition is a process in which transport of pollutants occurs when the pollutants mingled with the dust particles in the air settle on the ground, buildings, cars and trees. The particles settled on the surface of tree leaves participate in photosynthesis. During photosynthesis, the air that is inhaled in the presence of sunlight and water is used in making essential nutrients for their survival. The trees can only absorb some particles while other particles are deposited on the surface of plants. These particles are later washed away by rain, returned to atmosphere or drop off to the ground during fall. The deposition of an air pollutant for a given period of time is the product of pollutant flux, total green area and time period. The pollutant flux F is the function of dry deposition velocity and concentration of pollutant. The dry deposition velocity depends on aerodynamics, boundary layer and canopy characteristics.

The pollution removal for local use is standardized using national data (Nowak 2002). The total pollution removal is the product of average pollutant concentrations, standardized removal rates and total tree cover. In this study, the annual pollution removal rate per square meter of area and total air pollutant removal per year are estimated to be 2.7 g/m^2 and 107 metric tons per year respectively. These values are calculated for the pollutant NO_2 in Baltimore MD. Most studies consider the removal of pollutants from the atmosphere with the objective of increasing tree planting, increasing parks area, or improving temperature in an area. That is why this removal rate is useful in estimating the pollutant’s dispersion in other studies.

Another important aspect to consider in estimating the air pollutant dispersion is the pollutant's chemical properties. Nitrogen dioxide is a reactive gas. In the presence of ultraviolet radiation, oxygen (O_2) and nitrogen dioxide (NO_2) react in the atmosphere to form Ozone and Nitric Oxide (NO) through the reactions given in Eq. 6 and Eq. 7.

$$NO_2 + hn \rightarrow NO + O \qquad \text{Eq. 6}$$

$$O + O_2 \rightarrow O_3 \qquad \text{Eq. 7}$$

This resulting ozone reacts quickly to form nitrogen dioxide by the titration process specified in Eq. 8. In isolation the reaction continues till equilibrium or steady state is achieved. In the absence of anthropogenic emissions, these reactions normally result in natural ozone concentration (Altshuller and Lefohn 1996).

$$O_3 + NO \rightarrow NO_2 + O_2 \qquad \text{Eq. 8}$$

This photochemical reaction shows that nitrogen dioxide is present in the atmosphere after the chain reaction of transformation into ozone and again converting to nitrogen dioxide. Here, the objective is to find the chronic population exposure due to nitrogen dioxide, and the pollutant photochemistry suggests that the pollutant is present in the air after the chain reaction. Hence the pollutant may be responsible for increased illnesses. The problem is now how to estimate the population exposure of the air pollutant. To answer this question the health risk assessment is reviewed.

2.3 Health Risk Assessment

The impacts of the diffused concentrations in the atmosphere can be determined by risk management strategy. Health impacts can be estimated by risk assessment. Risk

Assessment (Theodore and Theodore 2010) involves integration of the information and analysis of health risk evaluation to measure magnitude of risk and degree of confidence associated with its characterization. The health risk evaluation involves four steps described as: a) Hazard Identification, b) Dose-Response Assessment, c) Exposure Assessment, and d) Risk Characterization. The risk can be evaluated qualitatively and quantitatively.

The hazard identification may be helpful in identifying the adverse effects of new toxic compound that has been found linked to certain new conditions or increase in some illness incidences. The hazardous effects of nitrogen dioxide have been long known and have been proven in many studies conducted in the past. The prominent illnesses are asthma, upper and lower respiratory problems, cardiovascular problems and cerebral-vascular problems. The dose-response assessment is useful in assessing the exposure on a population. As the proposed methodology examines the dispersion and exposure of the pollutant, using the dose response relation will help in predicting the population exposure. Therefore the dose-response assessment, exposure assessment, and risk assessment, the components of health risks assessment that need to be investigated. These components are related to each other.

Exposure, as defined in the guidelines for exposure assessment (US EPA 1992), “is the contact of a chemical with the human outer boundary. The process of a chemical entering the body can be described in two steps: contact (exposure), followed by actual entry (crossing the boundary)”.

Exposure over a period of time can be represented by a time-dependent profile of the exposure concentration. The area under the curve of this profile is the magnitude of the exposure, in concentration-time units (National Research Council, NRC 1990) (Lioy 1990)):

$$E = \int_{t_2}^{t_1} C(t)dt \qquad \text{Eq. 9}$$

where E is the magnitude of exposure, C(t) is the exposure concentration as a function of time, and t is time, t_2 - t_1 being the exposure duration (ED). If ED is a continuous period of time (e.g., day, week, year, etc.), then C(t) may be zero during part of this time. There are different types of doses. Regardless of their type, doses are presented in terms of dose rates, or the amount of a chemical dose (applied or internal) per unit time (e.g., mg/day), or as dose rates on a per-unit-body-weight basis (e.g., mg/kg/day). In this study, the chronic exposure population will be estimated for medium to high traffic emissions.

Exposure is assessed based on the dose rates, as intermittent or weighted average exposure. Time-weighted average dose rate is the total dose divided by the time period of exposure, usually expressed in units of mass per unit time, or mass/time normalized to body weight (e.g., mg/kg/day). Time-weighted average dose rates, such as the lifetime average daily dose (LADD) are used in dose-response equations to estimate effects or risk. Intermittent air exposures are estimated for the duration of exposure, e.g., for 8 hours exposed/day times a cubic meter of air inhaled/hour. The intermittent air exposure does not seem sufficient for this study. That is why the time-weighted exposure which is usually used to assess the impacts of carcinogens is considered.

The Average Daily Doses (ADDs) are used for assessing the time-weighted exposure. This parameter does not differentiate between acute effects and long term effects. The ADD is given by:

$$\mathrm{ADD_{pot}} = \frac{[\mathrm{C^{-}.\ IR^{-}.\ ED}]}{[\mathrm{BW.AT}]} \qquad \text{Eq. 10}$$

where $\mathrm{ADD_{pot}}$ is the average daily potential dose, ED is the sum of the exposure durations for all events, and $\mathrm{C^{-}}$ and $\mathrm{IR^{-}}$ are the average values for these parameters, BW is body weight, and AT is the time period over which the dose is averaged (converted to days). Sometimes the dose–exposure function is also termed concentration-response function.

Risk (US EPA 1992) (WHO 2000) is calculated based on exposure scenarios for evaluating proposed options for action. Exposure scenarios are calculation tools intended to help develop estimates of exposure, dose, and risk. An exposure scenario generally includes facts, data, assumptions, inferences, and sometimes professional judgment about the location, pollutant characteristics, and exposed population. Defining a scenario or a case is very important in almost any study or research.

The risk descriptors are estimated by either the probabilistic descriptor or the percentage of population. The probabilistic descriptor of health effect cases estimated in the population of interest over a specified time period. This descriptor is obtained either by summing the individual risks over all the individuals in the population, or by multiplying the slope factor obtained from a carcinogen dose-response relationship, the arithmetic mean of the dose, and the size of the population.

The second type of population risk descriptor is an estimate of the percentage of the population, or the number of persons, above a specified level of risk. This descriptor is obtained by measuring or simulating the population distribution. This analysis is done through Monte Carlo simulation method. This approach is helpful when no data are available but requires some input. The first approach requires a variety of data that are difficult to obtain. However, if there are other studies that have developed a probabilistic function through extensive research, then that is useful for finding the population risk.

The guidelines (WHO 2000) for using the epidemiological evidences for environmental health risk assessment demonstrate how relative risk can be modeled for a given population. In quantification guidelines for health impacts of exposure to air pollution (WHO 2000), various estimators of the health impact of air pollution have been employed in recent health impact assessments are discussed. These guidelines describe a somewhat similar risk assessment methodology to that of the EPA. However, these estimators are estimated for a population with different characteristics and with different health consequences.

(Spadaro and Rabl 2002) conducted a study in which the cost of damages due to air pollutants from power plant, the damages are calculated as a function of slope of dose-response function, receptor density, depletion velocity and emission from a source. A uniform world model was used for population and the health function was determined in terms of cases per person-yr-μgm/m^3. This approach was similar to the aforementioned guidelines.

The risk assessment tool, BENMAP has also been developed by EPA (US EPA 2011) to assess the exposure of pollutants emitted from point sources. The health effects in this tool are estimated from the model:

$$\text{Health Effect} = \text{Air Quality Change} * \text{Health Effect Estimate} * \text{Exposed Population} * \text{Health Baseline Incidence} \quad \text{Eq. 11}$$

where Air Quality Change is the difference between the baseline and changed air pollution level; Health Effect Estimate is the percentage change in an adverse health effect due to a unit change in ambient air pollution; Exposed Population is the number of people affected by the air pollution reduction; and Health Baseline Incidence is an estimate of the average number of people that die in a given population over a given period of time. This model is useful if the health effect estimate is known. The health effect from a similar study can also be used in this health effect model. It will complete the risk assessment for non-carcinogenic nitrogen dioxide.

From the literature review it has been found that the total environmental cost can be estimated in detail by using the approach used for point source of air pollution. This is possible by using an appropriate model for dispersion, concentration- response function and the associated risks. The proposed methodology developed after the literature review is described in the next section.

Chapter 3: Methodology

The evaluation of a highway for construction is important in a highway network. Mostly, in evaluating a highway, the environmental and social costs are not estimated in detail. The environmental costs are estimated in detail for the point sources. Despite the difference in the type of source, the environmental assessment methodology is also applicable to the non-point or mobile sources. The methodology for point sources includes the emission at source, air pollutant dispersion, and health risk assessment. From the literature reviewed in previous section, the different approaches of dispersion modeling and health risk assessment were evaluated. By using the existing components available from other studies and areas the environmental costs can be estimated to address the negative chronic impacts that may arise due to air pollutants. The following steps can be used for estimating the environmental costs and a flow chart of these steps is shown in Figure 4.

- Estimate the dispersion of the air pollutants for expected induced traffic
- Find the exposure from the pollution
- Find the social cost associated with air pollutants

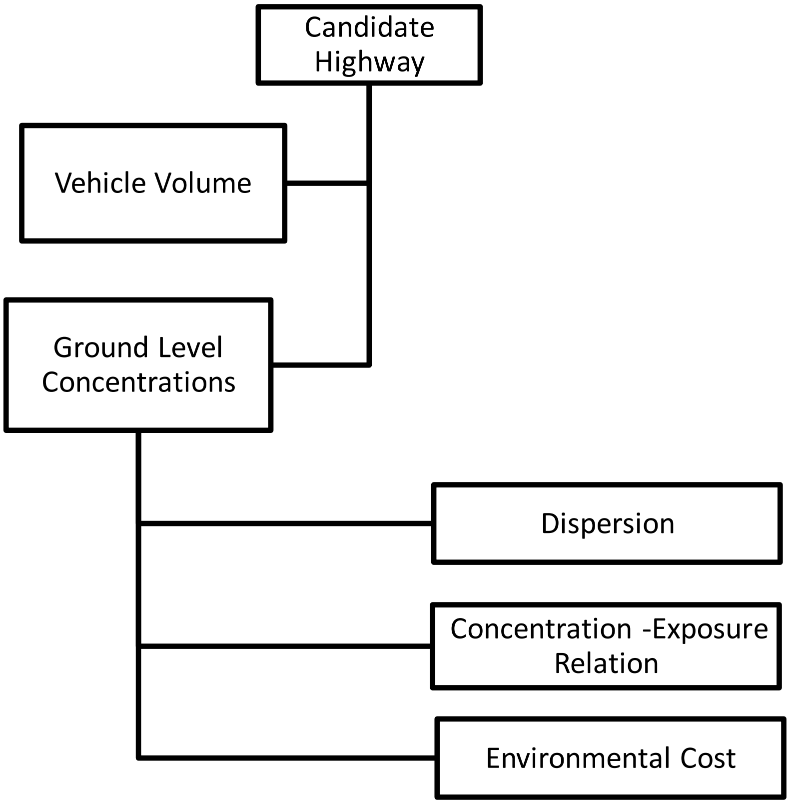

Figure 3 Proposed Evaluation Methodology

In Figure 3, the methodology steps are shown in their order of occurrence. These steps show how the emissions' costs can be estimated for a candidate highway by using the population exposure. The initial case is defined in the Figure 4. It shows that the proposed highway is located east of the existing highway, the tree cover is adjacent to the new highway and the residential area is east of the new highway. The objective is to find the exposure and risk, to the residential area that is protected by tree cover, of the pollutants that are generated from the new highway and a nearby existing highway. The estimate is made for 1 km stretch of highway. The emissions at the source are calculated from vehicle-specific emission approach. From the emissions at the source, the deposition of nitrogen dioxide is deducted due to the presence of trees spread over half a kilometer. These deducted emissions are used as input into the dispersion model. After modeling dispersion from the new highway, the background concentrations from a nearby highway are added to obtain the total dispersion. Therefore the total concentration (C) at any point (x,y) is given by

$$\Sigma C(x,y) = C_{background} + C_{dispersed} \qquad \text{Eq. 12}$$

This nearby highway is located 1 kilometer from the new highway. The concentration of nitrogen dioxide is estimated in a rural area at downwind distances x = 0.1, 0.25, 0.5, 0.75, 1, 2.5, 5, 7.5, 10, 15, 30 kilometers.

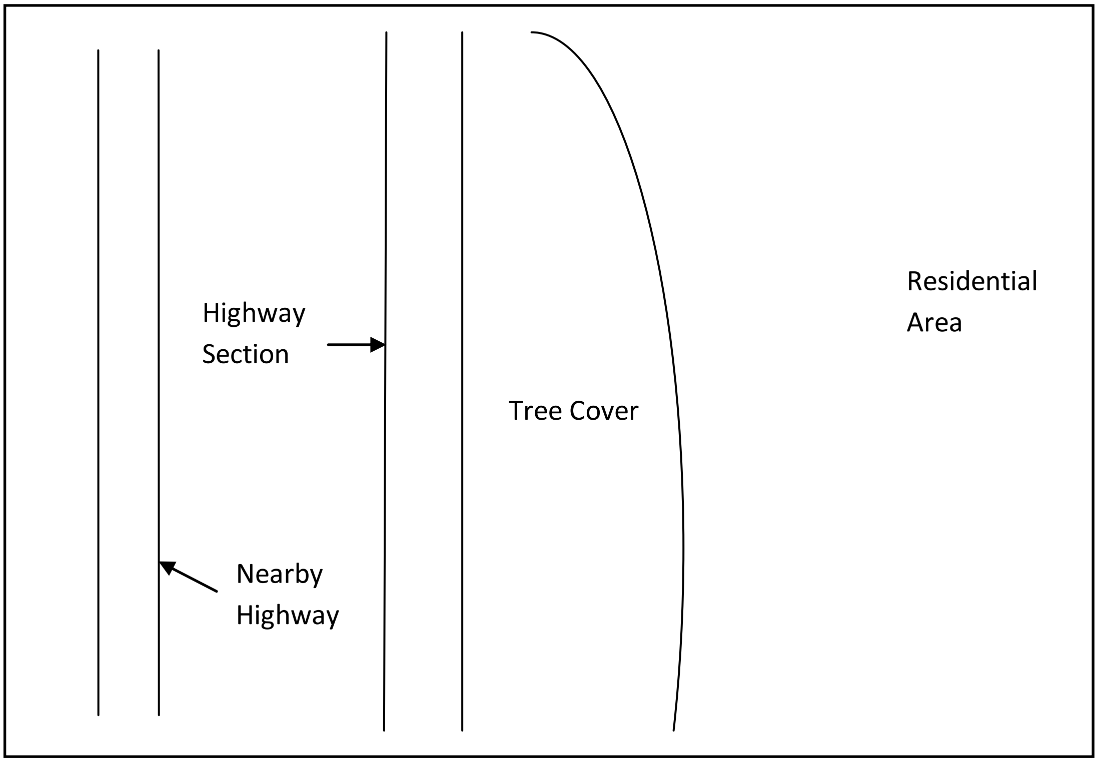

Figure 4 Case Illustration

<u>3.1 Emissions Estimation</u>

3.1.1 Emissions at Source

Various methods based on different approaches can be used to estimate the traffic related emissions for a highway. For macroscopic emission estimates, the vehicle miles of travel are used to estimate the individual pollutant as well as the total emissions. At a microscopic level, the emissions at the source are estimated using two approaches: vehicle-specific power, and physical parameterized fuel rate (US EPA 2005, US EPA 2003). The emission rates used in MOVES are generated from hybrid approach, Physical Emission Rate Estimator (PERE) that uses the Vehicle Specific Power, VSP and Comprehensive Modal Emissions Model, CMEM models. The VSP approach the road-load based approach that incorporates the resistive forces, tire resistance, gradient and aerodynamics in estimating the power generated by a car to move its weight. The CMEM approach also considers these resistive forces in estimating power. The method of estimating the values of coefficient for resistive forces differs from that of VSP approach. The tractive (or brake) power (in kW/tonne) is:

$$\mathrm{Pb}(\mathrm{m},\mathrm{v},\mathrm{a}) = \frac{\mathrm{A} * \mathrm{v} + \mathrm{B} * \mathrm{v}^2 + \mathrm{C} * \mathrm{v}^3 + \mathrm{m} * \mathrm{v}\,(\mathrm{a} + \mathrm{g}\sin\theta)}{\mathrm{m}} \qquad \text{Eq. 13}$$

where A and B are rolling resistance coefficients and C is an aerodynamic resistance coefficient. These coefficients are determined from dynamometer coastdown tests. v is vehicle speed (assuming no headwind) in m/s; a is vehicle acceleration in m/s^2, g is acceleration due to gravity and m is vehicle mass in metric tonnes. It uses the vehicle specific power times mass to physically represent the emissions at microscopic level. The coefficients in this model are determined by using I/M (Inspection/Monitoring) dyno testing for the Track Road Load Horse Power (TRLHP) which is given by

$$TRLHP = \frac{0.5 * \frac{ETW}{32.2} * (v_1^2 - v_2^2)}{(550 * ET)} \qquad \text{Eq. 14}$$

where ET is Elapsed Time for the vehicle on the road to coast down from 55 to 45 mph, ETW is Equivalent Test Weight in pounds, V_1 is Initial velocity in feet/second, V_2 is Final velocity in feet/second. To calculate this TRLHP the look up tables in IM240 technical guidance (US EPA 2000) are used. Because the guideline suggests using the reference data when empirical data are unavailable, the value of elapsed time based on IM240 lookup tables is used. The coast down (the decelerating part of a trip) values of speed in this table from 55mph (219 seconds) to 45 mph (226 seconds) corresponds to an elapsed time of 22 seconds. The weight for LDV is taken as an average 3300 pounds (NHTSA 2008) that gives a TRLHP value of 9. The values of coefficients A, B and C at 50mph are determined by the following expressions provided in the look up tables:

$$A = 0.35*0.746*TRLHP/(50) \qquad \text{Eq. 15}$$

$$B = 0.1* 0.746* TRLHP/(50)^2 \qquad \text{Eq. 16}$$

$$C = 0.55* 0.746* TRLHP/(50)^3 \qquad \text{Eq. 17}$$

The values of these coefficients suggest the VSP of 29 KW/tonne for a speed of 60 mph (88 m/s), acceleration of 3 ft/sec^2 (0.915m/sec^2) in a level grade. After calculating the VSP, the corresponding emission rate for NO_2 is obtained from Table 1. It gives the average emission rates in mg/ sec for VSP in KW/tonne are for the Tier 1 light duty gas vehicles from NCHRP data set with engine displacement < 3.5 L and odometer <50,000 miles (Frey, et al. 2002).

Table 1 Air Pollutant Emission Rate for Corresponding Vehicle Specific Power

VSP Mode	**Definition (kw/t)**	**NO_2 (mg/sec)**	**HC (mg/sec)**	**CO (mg/sec)**	**CO_2 (g/sec)**

1	VSP < -2	0.9	0.5	7.8	1.7
2	- 2 ≤ VSP < 0	0.6	0.3	3.9	1.5
3	0 ≤ VSP < 1	0.3	0.4	3.3	1.1
4	1 ≤ VSP < 4	1.2	0.4	8.3	2.2
5	4 ≤ VSP < 7	1.7	0.5	11	2.9
6	7 ≤ VSP < 10	2.4	0.7	17	3.5
7	10 ≤ VSP < 13	3.1	0.8	20	4.1
8	13 ≤ VSP < 16	4.2	1.0	29	4.6
9	16 ≤ VSP < 19	5.1	1.1	36	5.2
10	19 ≤ VSP < 23	5.9	1.4	55	5.6
11	23 ≤ VSP < 28	7.6	2.1	114	6.5
12	28 ≤ VSP < 33	12.1	3.4	208	7.7
13	33 ≤ VSP < 39	15.5	4.9	442	9.0
14	39 ≤ VSP	17.9	10.9	882	10.9

Notes: The average confidence intervals for these VSP modes in % are ±7, ±10, ±13, and ±2, respectively for NO_2, HC, CO, and CO_2;

For running exhaust pollutant emission, the model uses FTP and SFTP (Federal Test Procedure and Standard Federal Test Procedure) with power <18KW/tones and >18KW/tones respectively; and adjusts these values for age and future forecast. The FTP standard values for different vehicle classes suggest different rates with respect to vehicle size. The emissions rates at the source for light duty and heavy duty diesel trucks are taken from EPA (US EPA 2009). These emission rates are based on the extensive data results obtained using the Portable Emission Measurement System (PEMS) and are also used in the regulatory models to estimate the emissions generated. The emission rates of nitrogen dioxide for different vehicle types are given in Table 2. These emission rates are used in estimating the emissions generated from the assumed vehicle mix.

Table 2 Emission Rates for NO_2

Vehicle Type	**NO_2 Emission**
LDV (Tier 1) (From Table 1)	12.1 mg/sec (running)

LDV (diesel, 2005-2008) (US EPA 2009)	5.5 gm/hr (running)
LDT (diesel, 2005-2008) (US EPA 2009)	6.47 gm/hr (running)
HDT (diesel, 2007+) (US EPA 2009)	4.0 gm/bhp-hr

The vehicle mix assumed for initial calculations is based on the National Highway Statistics (US FHWA 2010) and National Highway Traffic Safety Administration (NHTSA 2008) data that defines the car composition of 82.7% Light Duty Vehicles and Trucks running on gasoline, 0.8% Light Duty Vehicles, 2.49% of Light Duty Trucks run on diesel, and 12.2% Heavy Duty Trucks in a rural setting. The volume is assumed to be 16000 veh/day and 667 vehicles per hour for a two lane road. Table 3 shows the emissions generated from this vehicle mix. Now, for calculating deducted emissions, the annual pollution removal rate per square meter of area calculated by (Nowak 2002), discussed in Section 2, is used. The total NO_2 emission after removal by trees is given in Table 4. The deducted emissions show slight reduction in emission estimate.

Table 3 Emissions Generated from Vehicles

Vehicle Type	**Nox Emission Rate (gm/ hr)**	**% Vehicle Type**	**Vehicles**	**Emission/ hr**	**Emission/ day**	**Emission/ yr**
LDV	43.56	82.7	551	24,016	384,257	140,253,907
LDV (diesel)	5.50	0.8	5	29	469	171,307
LDT (diesel)	6.47	2.5	17	107	1,718	627,228
HDT (diesel, 400 bhp)	1,600.0	12.2	81	130,133	2,082,133	759,978,667
	Total Emissions in gm			154,286	2,468,578	901,031,108
	In kg				2,469	901,031

	In tons				2	887

Table 4 Emission Estimate after Removal by Trees

	Emissions/hr	**Emissions/day**	**Emissions/yr**
Total Emissions, in gm	142,072	2,456,364	901,018,894
In kg		2,456	888,816
In tons		2	875

3.1.2 Dispersion of Air Pollutants

In the literature review, it is found that a Gaussian model modification formulated by Venkatram and Horst (HV) is preferable. This model approximates the effect of wind angle in the concentration of pollutant emitted from the vehicles on road. The HV model is given in Eq. 5 and the related inputs are also explained in Section 2. From preliminary results of the simple and HV models, it is found that the effect of wind direction on accuracy may not be important in estimating the overall emissions in an area. The results of the HV model give a composite distribution of air pollutants. That is why it is not appropriate for use in this methodology. The dispersion models are usually described for one parameter and neglect the other. Therefore the dispersion of pollutants is approximated for any given condition using a simple line dispersion model for highway, expressed in Eq. 2. This simple model uses the sigmas calculated by Brigg's approach (Table 5) discussed in Section 2.

Table 5 Briggs Formulation for Sigmas

Land Category	**Stability Class**	**σ_y**	**σ_z**
Rural	A	0.22x(1+0.0001x)-½	0.20x
	B	0.16x(1+0.0001x)-½	0.12x
	C	0.11x(1+0.0001x)-½	0.08x(1+0.0002x)-½
	D	0.08x(1+0.0001x)-½	0.06x(1+0.0015x)-½
	E	0.06x(1+0.0001x)-½	0.03x(1+0.0003x)-½
	F	0.04x(1+0.0001x)-½	0.016x(1+0.0003x)-½
Urban	A-B	0.32x(1+0.0004x) -½	0.24x(1+0.001x) -½
	C	0.22x(1+0.0004x)-½	0.20x
	D	0.16x(1+0.0004x)-½	0.14x(1+0.0003x)-½
	E-F	0.11x(1+0.0004x)-½	$0.08x(1+0.0015x)^{-1}$

The emission rate (gm/hr) estimated in Table 4 is used in the simple line source model. The new highway is assumed to be perpendicular to the wind direction. The atmospheric stability condition is neutral, wind speed is 10 kph and the stack height z is 1 meter above ground. The emission of NO$_2$ at source is 39.4*10 mg/sec. The concentrations along the highway are calculated at 10 points, presenting 1 km of road. The results are shown in Table 6 and Figure 5. Next, the background concentration is estimated for nitrogen dioxide.

Table 6 Results of the Air Dispersion Model for the New Highway

Length (km)	**Estimated Concentration of Ground-Level Pollution (mg/m^3) on Plume Centerline at Selected Distances (km) from Source**										
	0.1	0.25	0.5	0.75	1	2.5	5	7.5	10	15	30
0.1	553.87	245.40	138.71	101.95	82.95	45.75	30.60	24.49	20.99	16.96	11.86
0.2	553.87	245.40	138.71	101.95	82.95	45.75	30.60	24.49	20.99	16.96	11.86
0.3	553.87	245.40	138.71	101.95	82.95	45.75	30.60	24.49	20.99	16.96	11.86
0.4	553.87	245.40	138.71	101.95	82.95	45.75	30.60	24.49	20.99	16.96	11.86
0.5	553.87	245.40	138.71	101.95	82.95	45.75	30.60	24.49	20.99	16.96	11.86

0.6	553.87	245.40	138.71	101.95	82.95	45.75	30.60	24.49	20.99	16.96	11.86
0.7	553.87	245.40	138.71	101.95	82.95	45.75	30.60	24.49	20.99	16.96	11.86
0.8	553.87	245.40	138.71	101.95	82.95	45.75	30.60	24.49	20.99	16.96	11.86
0.9	553.87	245.40	138.71	101.95	82.95	45.75	30.60	24.49	20.99	16.96	11.86
1	553.87	245.40	138.71	101.95	82.95	45.75	30.60	24.49	20.99	16.96	11.86

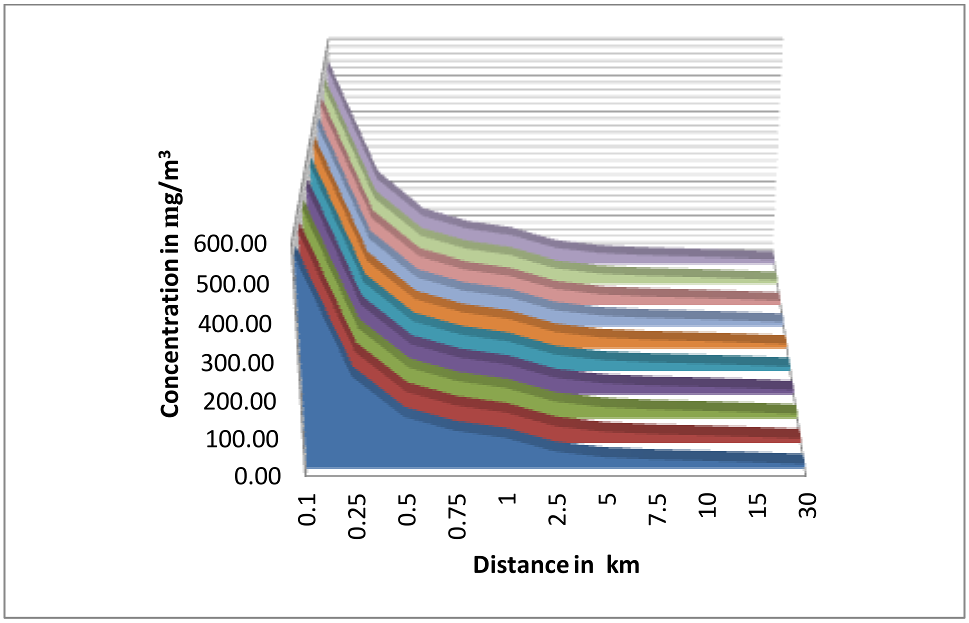

Figure 5 Concentration of NO_2 from the Highway

3.1.3 Background Concentration

The background concentration is considered to come from a nearby existing highway. The total emission concentration is defined in Eq. 12. Let’s assume that an existing highway is inclined at 60° from the new highway and carries a volume that is equal to that of Baltimore-Washington Parkway. The Average Annual Daily Traffic, AADT on this existing highway is 77,632 veh/day (MD SHA 2011). The vehicle composition used for new highway is based on the rural highway statistics in Maryland therefore the same vehicle mix is used for the existing highway. The resulting hourly highway emissions for the existing highway are shown in Table 7.

Table 7 Hourly Emissions for Existing Highway

	Nox Emission (gm/ hr)	**% Vehicle Type**	**Vehicles**	**Emission gm/hr**
LDV	43.56	82.7	2,675	116,526
LDV (diesel)	5.50	0.8	26	142
LDT (diesel,2011+)	6.47	2.5	81	521
HDT (diesel, 400 bhp,till 2004)	1,600.0	12.2	395	631,407
				748,596

An emission rate of $207.943*10^3$ mg/sec is used for the existing highway. The prevalent wind direction in the area is south west and north east, contingent upon the seasonal patterns. The highways that are perpendicular to this direction will use the simple line model as it is. However, if the highway is not perpendicular to the wind direction then the model is adjusted for this angle by dividing the dispersion equation with the sine of angle i.e. 60°. Here, this adjustment is being made and the pollutant's dispersion for the existing highway is shown in Table 8. The other conditions are the same as for the new highway.

Table 8 Dispersion of Pollutants from Existing Highway

Estimated Concentration of Ground-Level Pollution (mg/m^3) on Plume Centerline at Selected Distances (km) from Source										
0.1	0.25	0.5	0.75	1	2.5	5	7.5	10	15	30
3479.28	1502.25	845.62	620.94	505.04	278.39	186.19	149.01	127.72	103.19	72.19

3.1.4 Total Dispersion

Depending on the distance and direction from the existing highway, the concentration added to the background concentration varies. The farther the highways are, the smaller is the impact on concentration. Table 9 shows the total dispersion values for different distances from new highway with background concentration emitted 1 km away. The background concentration at 1 km combined with the dispersion concentration at 0.1 km from the highway is

$$\Sigma C(x,y) = C_{\mathrm{background}} + C_{\mathrm{dispersed}} = 505.04 + 553.87 = 1058.91\ \mathrm{mg/m^3}$$

After the total dispersion estimate the next step in the methodology is estimating the population exposure and risk.

Table 9 Total Concentrations in mg/m^3 at Downwind Distances

	0.1	**0.25**	**0.5**	**0.75**	**1**	**2.5**	**5**	**7.5**	**10**	**15**	**30**
New Highway	553.87	245.40	138.71	101.95	82.95	45.75	30.60	24.49	20.99	16.96	11.86
Existing Highway	3479.28	1502.25	845.62	620.94	505.04	278.39	186.19	149.01	127.72	103.19	72.19
Total	1058.91	750.44	643.75	606.99	587.99	550.78	535.64	529.53	526.03	522.00	516.90

3.2 Exposure and Risk Assessment

The health risk assessment is carried out by using the exposure assessment through the dose-expose procedure based on the second approach (discussed in Section 2) is shown in Figure 6. First, the exposure doses of the pollutants are calculated from the concentration. Second, the Monte Carlo method is used to simulate the expected number of deaths. The mortality data are used to obtain pseudorandom numbers. Finally, the expected number of deaths is estimated from the doses and random numbers for a population.

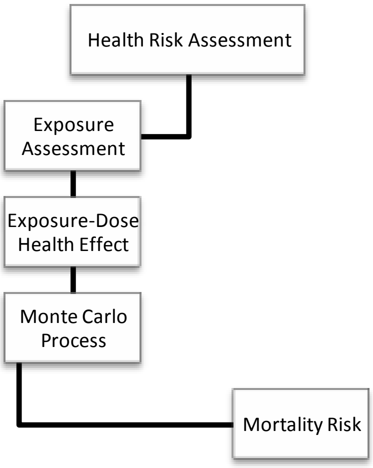

Figure 6 Risk Assessment

The Average Daily Dose Potential (ADD_{pot}) is estimated by the relation given in Eq. 10. The inputs for this model are as follows: Concentration is the dispersed concentrations of a pollutant corresponding to the downwind distances; Intake rate = 1 per sec; Body weight = 70 kg; Averaging Time in hours is 1 hr which is converted to days i.e. 0.0416 days; and Exposure Duration is 3 hrs each weekday. The ADD_{pot} for nitrogen dioxide at the selected distance is shown in Table 10.

Table 10 Average Daily Doses for NO_2

Distance from Road (km)	**'C' Concentration at Distances (µgm/m^3)**	**ADD_{pot} (mg/kg/day)**
0.1	946.62	0.0406
0.25	691.27	0.0296
0.5	602.95	0.0258
0.75	572.52	0.0245

1	556.79	0.0239
2.5	525.99	0.0225
5	513.46	0.0220
7.5	508.40	0.0218
10	505.50	0.0217
15	502.16	0.0215
30	497.95	0.0213

Exposure is estimated for the population which is affected by the emission pollutants at selected distances from a proposed highway. The exposure assessment is carried out by using the population risk distribution in the absence of health data using the Monte Carlo Simulation. To estimate the risk theoretically the data for mortality causes are obtained from the Center of Disease Control (NVSR 2009). From these data the mortalities due to outdoor air pollution are obtained for Maryland. The categories of cerebrovascular diseases, influenza and pneumonia, and chronic lower respiratory diseases are selected to use in simulation. For a population of 100,000, the expected number of deaths from relative probability (shown in Table 11) suggests that 22 out of 100 influenza and pneumonia cases, 52 out of 100 cerebrovascular cases, and 47 out of 100 chronic lower respiratory diseases cases are result of outdoor pollution. These data are used to generate random numbers. The relative probability of each of the causes is calculated from the number of deaths. The pseudorandom numbers are generated from a normal distribution (mean of 0.0405 and standard deviation of 0.01600) of relative probability. The mean and standard deviation are obtained from the descriptive statistics of the death and relative probability of risk.

The hazard potential is calculated for the pollutant nitrogen dioxide by using the threshold value of 0.053 ppm (0.0997 mg/m^3) (US EPA) and the expected deaths (E(p)) from the observed data. A population of 100 people is used for expected number of deaths. Hazard potential = threshold value/E(p); the expected number of deaths caused from the concentration at the downwind distances of the highway is calculated by using Eq. 18.

Expected number of deaths = hazard potential * ADDpot * population Eq. 18

The expected risk estimate for a population of 100,000 is 1163.4 for the nearest resident, as shown in Table 12. From these results it is found that an average of 830 people per 100000 dies theoretically from the air pollutant nitrogen dioxide. However, these results are not accurate because they do not consider the age factor and nitrogen dioxide is not as toxic. The results are also not reliable because the attributable data used are for total outdoor pollution and not for individual pollutants. That is why the health function approach is employed, to estimate population exposure.

Table 11 Relative Probability of Illness

Cause	**Number**	**Rate**		**Relative p**
Cerebrovascular diseases	2,288	40.1	40.4	0.052186
Influenza and pneumonia	976	17.1	17.3	0.022261
Chronic lower respiratory diseases	2,064	36.2	36.8	0.047077

Table 12 Theoretical Expected Mortality Risk

Random Number	**E(p)**	**Hazard Potential**	**Expected No. of Cases from Additional Concentration per 100,000 people**
0.028604	2.860370016	0.286765	1163.391205
0.029312	2.931185093	0.293865	870.6003633
0.02647	2.647033482	0.265377	685.7578762
0.04186	4.185964001	0.419662	1029.705848
0.033566	3.356626705	0.336517	803.0134744

0.043241	4.324098056	0.433511	977.245887
0.03444	3.443960024	0.345273	759.7808433
0.036009	3.600909706	0.361008	786.581134
0.036009	3.600909706	0.425178	921.120876
0.036009	3.600909706	0.255352	549.5514931
0.036009	3.600909706	0.271023	578.3785521

The health function approach uses Eq. 11 to estimate the population exposure. To calculate the health effect, the change in air quality is obtained by comparing the base-line emission concentration. The average value of concentration is used at different distances. A log-linear relationship is used to find the base line incidence rate

Eq. 19

$$y_0 = B.e^{\beta.PM}$$

where the parameter B is the incidence rate of y when the concentration of PM is zero, the parameter β is the coefficient of PM, and the change in incidence rate is given by

Eq. 20

$$\Delta y = B.e^{\beta\, PM_0}\left(1 - e^{-\beta(PM_0 - PM_c)}\right) = y_o.(1 - (1/\exp(\beta.\Delta PM)))$$

In Eq. 20, y_o is the baseline incidence rate of the health effect (i.e., the incidence rate before the change in PM). The change in the incidence of adverse health effects is then calculated by multiplying the change in the incidence rate, Δy, by the relevant population. The coefficient of PM is determined by the relation (for 97.5% Confidence Interval)

Eq. 21

$$\beta = \ln(RR)\,/\Delta PM$$

(Peel, et al. 2005) examines the associations between air pollution and respiratory emergency department visits i.e., asthma, COPD, URI, pneumonia, and an all respiratory-disease group. The study is carried out with seven years data in Atlanta, GA. The occurrence of visits is related to air pollution as a result of Poisson regression analysis.

The relative risk of 1.047 is used for a single-pollutant model, with increase of 20 ppb in NO_2 concentration. Using this value of RR= 1.047 and an increase in concentration of nitrogen dioxide as a function of dispersion concentration to find the coefficient β. The average dispersion concentration for a distance from 0.1 to 5 km is 171 mg/m^3. Therefore the coefficient β is

$\beta = \ln(RR)/\text{change in concentration} = \ln(1.047)/ 171) = 3*10^{-4}$ Eq. 22

Here, the health effects considered are asthma, respiratory and cardio–vascular. The baseline incidence rate (y_o) is calculated for ages between 0 to 99 and considers health effects. These rates are obtained from the data in BENMAP appendices (US EPA 2011). The baseline incidences (y_o) for emergency department visits and hospitalization rate obtained from this data are shown in Table 13 and Table 14. Table 13 shows that the average emergency department incidence rates are 0.5224, 3.8828, and 4.0976 for the illnesses asthma, respiratory problems (excluding asthma), and cardiovascular problems, respectively. The baseline incidence rates for hospitalization of asthma, respiratory problems (chronic lung diseases), and cardiovascular problems (non-fatal) are 0.1721, 0.5602, 0.4319, respectively. These incidence rates are expressed per 100 persons per year. The baseline incidence rates are used to find the health impact function, which is assumed to be lognormal function.

Table 13 Health Effects for Emergency Department Visit Rates (per 100 persons per year)

Illness	**Age Group**					**Average y_o**
	0-17	18-44	45-64	65-84	85+	
Asthma	0.865	0.557	0.441	0.381	0.368	0.5224
Respiratory	4.907	2.332	1.832	3.894	6.449	3.8828

Cardiovascular	0.029	0.302	1.695	5.95	12.512	4.0976

Table 14 Health Effects for Hospitalization Rate (per 100 persons per year)

Illness	**Age Group**										**Average** y_o
	0-1	2--17	18-24	25-34	35-44	45-54	55-64	65-74	75-84	85+	
Asthma	0.33	0.155	0.051	0.055	0.094	0.136	0.16	0.198	0.24	0.302	0.1721
Respiratory	0.35	0.158	0.057	0.064	0.135	0.299	0.503	1.028	1.502	1.506	0.5602
Cardiovascular	0	0	0.01	0.017	0.071	0.215	0.379	0.673	1.096	1.858	0.4319

The health impact function expressed in Eq. 20 is used to estimate the health effect of nitrogen dioxide on persons living within 5 km of the highway. This function uses the coefficient β calculated in Eq. 22, baseline incidence rates calculated in Table 13 and Table 14, and concentrations based on the downwind distances. The health impact functions with respect to distance are shown in Table 15 and Table 16 for emergency department visits and hospitalization rate. The health effect estimate is obtained by multiplying the health impact function by population. The population density is 100 persons per 5 square kilometers. The health effect estimate is presented in the last column of this table. The health effect estimate is in terms of number of cases per year.

Table 15 Cases of Emergency Department Visits per Year

Illness	**Health Function** **ΔY**				**Health Effect Estimate for 5 sq. km of New Highway**
	Concentration in mg/m^3 at Distances in km				
	(0.1-0.5)km	***(0.5-1)km***	***(1-2.5)km***	***(2.5-5)km***	
	312.661mg/m^3	*107.871mg/m^3*	*64.348 mg/m^3*	*38.173 mg/m^3*	
Asthma	0.042	0.015	0.009	0.005	7.115
Respiratory	0.312	0.111	0.066	0.040	52.882
Cardiovascular	0.329	0.117	0.070	0.042	55.807

Table 16 Cases of Hospitalization per Year

Illness	**Health Function ΔY**				**Health Estimate for 5 sq. km of New Highway**
	Concentration in mg/m^3 at Distances in km				
	(0.1-0.5)km	***(0.5-1)km***	***(1-2.5)km***	***(2.5-5)km***	
	312.661mg/m^3	*107.871mg/m^3*	*64.348 mg/m^3*	*38.173 mg/m^3*	
Asthma	0.014	0.005	0.003	0.002	2.344
Respiratory	0.045	0.016	0.010	0.006	7.630
Cardiovascular	0.035	0.012	0.007	0.004	5.882

3.3 Monetary Cost of Environmental Impacts

The monetary values given to human mortality and morbidity are evaluated for estimating the monetary cost of health effects as a result of a new highway. The value is determined from the life cycle cost of estimated damages caused by emissions costs. (Small and Kazimi 1995) suggested the damages cost to be ¢0.24 to ¢1.48 per vehicle-mile of gasoline cars for NOx. The newer values estimated (Litman and Doherty 2012) for tailpipe emissions suggest a value of $0.04 per VMT in a rural setting. Here the environmental cost estimates used are dependent on the cost of illness on a household. The illnesses are caused by the combined effect of emissions rather than the individual impact of a single pollutant. The illnesses associated with pollutants have been discovered by numerous researches in epidemiology and public health. Therefore the dollar value of illness is used to find the health effect due to emissions. Although this method is not very accurate, it helps in estimating the cost of illnesses due to population exposure. The unit cost of an emergency visit for asthma as suggested by (Smith, et al. 1997) is $312. The unit cost of hospitalization depends on the length of stay in hospital and hospital charge for specific illness. For all hospital admissions, available in BENMAP Appendices (US EPA 2011), estimates of hospital charges and lengths of hospital stays are based on discharge data provided by the Agency for Healthcare Research and Quality's Healthcare Utilization Project National Inpatient Sample (NIS) database (2007). The monetary costs of emergency cases and hospitalization cases are presented in Table 17 and Table 18. Since nitrogen dioxide is not directly linked to deaths, the total environmental cost for NO_2 is the sum emergency cases and hospitalization.

$$C(\$/yr) = \Sigma C_{hospitalization} + C_{Emergency\ visit} \qquad \text{Eq. 23}$$

$$C(\$/yr) = 36{,}130.72 + 329{,}834.22 = 365{,}964.94$$

The total cost of emergency cases and hospitalization is approximately 0.36 million dollars per year. The total cost of the illnesses is for the pollutant nitrogen dioxide; however the resulting cost is not for this single pollutant but rather for the considered illnesses. The results of the proposed methodology are analyzed in the next section.

Table 17 Monetary Cost of Emergency Cases

Cost of Illness	**Unit Value**	**Cost of Cases in Dollars**
Asthma	312	$ 2,219.82
Respiratory	312	$ 16,499.08
Cardiovascular	312	$ 17,411.82
Total		$ 36,130.72

Table 18 Monetary Cost of Hospitalization Cases

Cost of Illness	**Mean Hospital Charge (yr 2000$)**	**Mean Length of Stay (days)**	**Total Cost of Illness (unit value in 2000$)**	**Cost of Cases**

Asthma	10,852	3.37	11,323	$ 26,540.09
Respiratory	19,009	5.35	19,612	$ 149,632.19
Cardiovascular	25,605	4.59	26,123	$ 153,661.94
Total				$ 329,834.22

Chapter 4: Analysis of Results

From the proposed methodology, the results are obtained by modeling the emission, dispersion, and health risks from a new highway. The case was defined in the previous chapter that considered a one kilometer section of proposed highway that is 60° from an existing highway, and whose emissions are blocked by tree cover before reaching the residents living at a certain distance from the new highway. The resulting risk from the concentration- exposure relation is contingent upon the traffic emissions and their dispersion, and population density. The dispersion tends to change with direction of wind, atmospheric stability and wind speed, vehicle mix, volume and its' composition, gradient, speed, temperature, and length of road.

4.1 Wind Direction

If the direction of the existing highway and the new highway are changed the background concentrations will also change. The variation in direction and the concentration at 1 km from the highway is shown in the Figure 7. By changing the angle of wind direction for the existing highway the background concentration shows larger change for lesser angle and if it is parallel to the new highway i.e. perpendicular to the wind direction, the concentration is minimized. Thus, if the wind direction for the new highway also differs from perpendicular to some angle the dispersion values will also change. The highest concentrations from highway are observed for the angles of 30° and 45°.

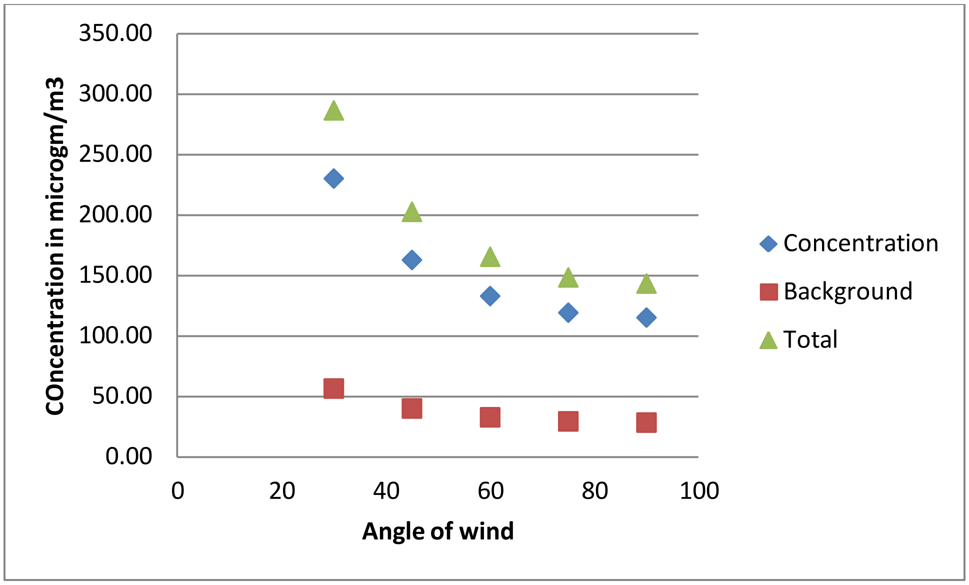

Figure 7 Effect of Wind Direction on Concentrations

4.2 Vehicle Mix

The vehicle composition of traffic heavily influence the emission concentration and so the NO_2 concentrations. The vehicle mix in the initial modeling process is analyzed for 5% and 15% increase in LDV and HDT (diesel); and 25% increase in LDV (diesel) and LDT (diesel). The changes are presented in Figure 8. The emissions are more sensitive to increase in LDV than other vehicle types. An increase of 15% in HDT significantly increases the amount of hourly emissions.

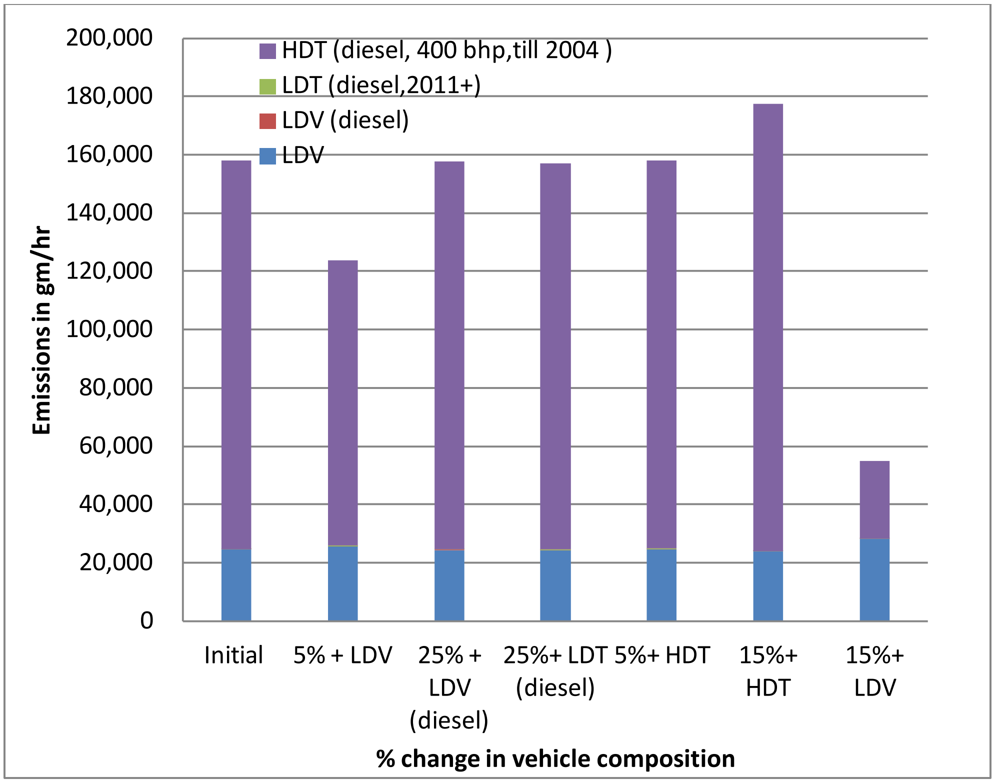

Figure 8 Effect of Vehicle Mix on Hourly Emissions

4.3 Meteorological Conditions

The dispersion of nitrogen dioxide is critical to meteorological conditions. The atmospheric stability is a parameter that defines meteorological conditions. The atmospheric stability is classified into six categories from very unstable (A) to stable condition (F). If the atmospheric stability class is changed from neutral to very unstable and stable, this will change the wind speed, shown in Table 19.

The atmospheric-stability class used in the previous section is neutral i.e. D. This class is applied since it can deal with any wind speed. To determine the effect of stability class on dispersion, the corresponding values of wind speed are plotted against the concentration. Figure 9 shows the concentrations of nitrogen dioxide (estimated in previous section) for unstable to neutral atmospheric-stability classes with their corresponding wind speeds. It is clear from this figure that the concentrations are higher for class A and lowest for class D. The results also show that in neutral atmospheric-stability condition the pollutant can travel farther than in unstable conditions.

Table 19 Pasquill-Gifford Method to Determine Stability Classes

Surface Wind (Measured at 10 m) (m / sec)	**Surface Wind (Measured at 10 m) (mph)**	**Day Incoming Solar Radiation (Insolation)**	**Day Incoming Solar Radiation (Insolation)**	**Day Incoming Solar Radiation (Insolation)**	**Night (Thin Over last or >= 4/8 Cloudiness)**	**Night (<= 3/8 Cloudiness)**
		(Strong)	**(Moderate)**	**(Slight)**		
< 2	< 4.5	A	A-B	B	F	F
2--3	4.5 - 6.7	A-B	B	C	E	F
3--5	6.7 - 11.2	B	B-C	C	D	E
5--6	11.2 - 13.4	C	C-D	D	D	D
6	13.4	C	D	D	D	D

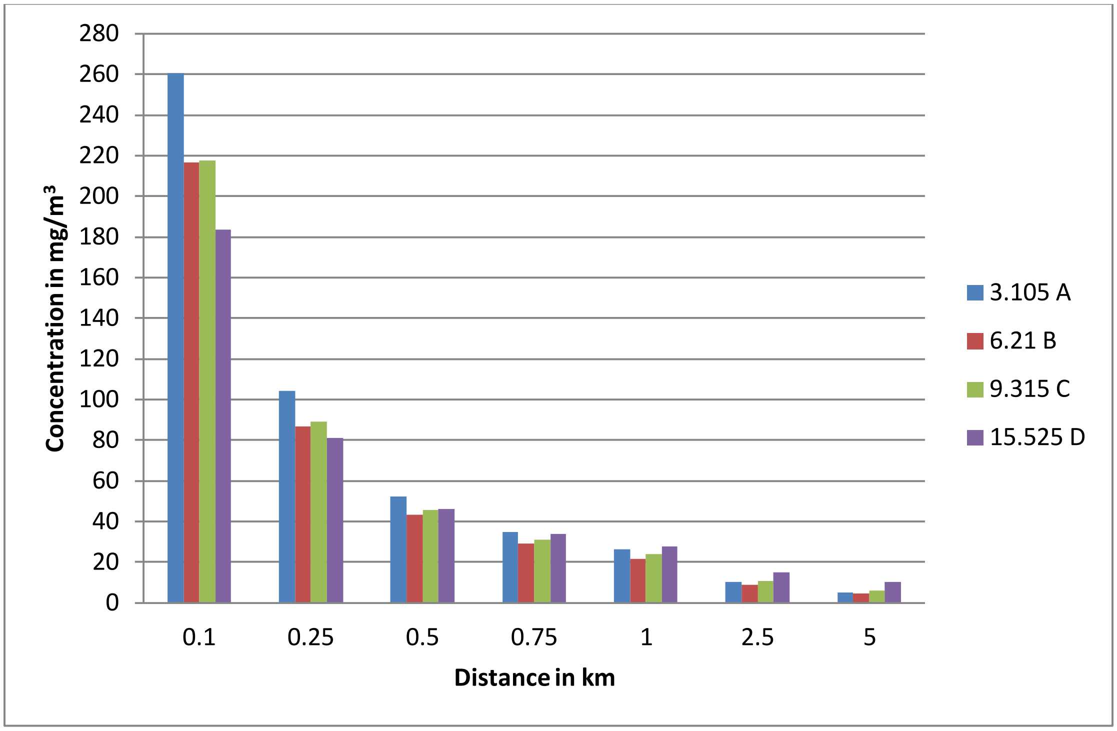

Figure 9 Effect of Atmospheric Stability on Dispersion Concentration

4.4 Vehicle Volume

The traffic volume on the highway is another factor that affects the emissions. The volume fluctuates throughout the day with peak hours during morning and evening. At other times the off-peak volume is observed. The effect of volume on emissions is analyzed for each hour throughout the day. This vehicle volume is adjusted by gradually reducing the off-peak flow to 117 veh/hr, and the peak hour flow does not exceed 667 veh/ hr. The hourly volume fluctuations throughout the day are shown in Figure 10. This figure shows the hourly volume has morning and evening peak flows.

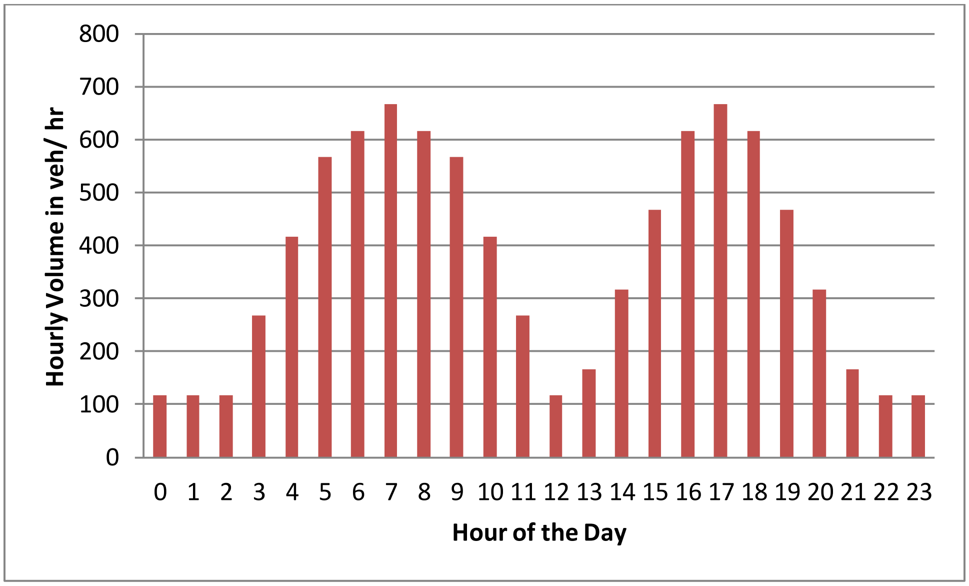

Figure 10 Volume Variations throughout the Day

Total NO_2 emissions are estimated for the initial vehicle mix. The resulting NO_2 emissions from this hourly variation are plotted (Figure 11) on a logarithmic scale to clearly observe the mode specific emissions. These emissions comprise 302.665 kg/hr from Light Duty Vehicles; 0.979 kg/ hr from Light Duty Vehicles running on diesel; and 5.73kg/hr and 1425.28 kg/hr from Light and Heavy Duty Trucks, respectively. The total nitrogen dioxide emissions after hourly adjustment are 1734.68 kg/hr.

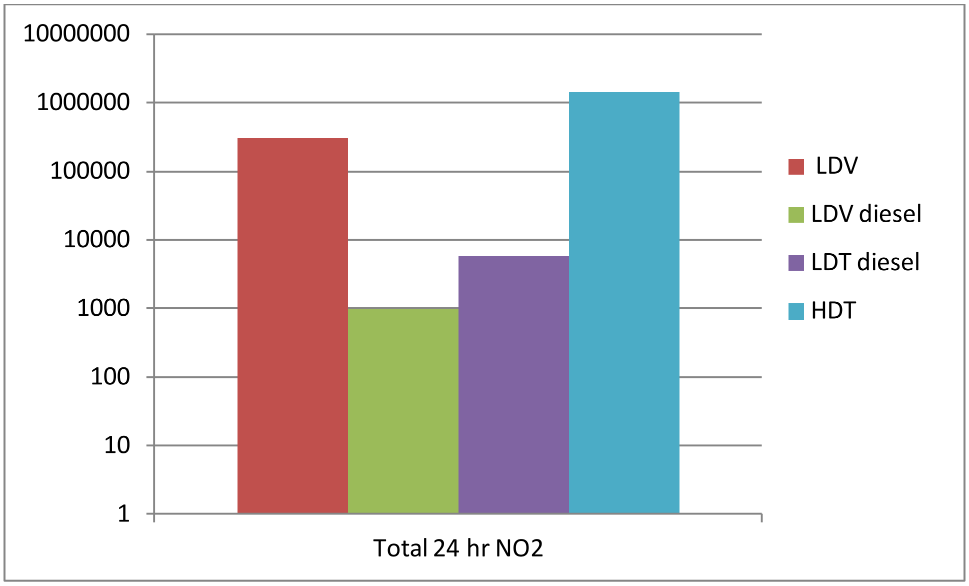

Figure 11 Nitrogen Dioxide Emissions over 24 Hours on Logarithmic Scale

4.5 Gradient

The gradient is directly related to vehicle emissions. That is, if the gradient increases the emissions also increase and vice versa. Similarly, the dispersion of air pollutants also changes with gradient. For analyzing the effect of gradient on emission rates, the emission rates are estimated for light duty vehicles on gradients of 0% to 5%.

The emission rates on gradients are estimated with a unit volume of vehicles. The emission rates, presented in Table 20, show that the major change in emission rate occur for 1% and 2% gradients. This is because the emission rates used here depend on Vehicle Specific Power mode (VSP mode) rather than Vehicle Specific Power (VSP in KW/tonne) itself. The VSP mode (in Table 1) uses ranges of vehicle specific power to look up the emission rates of Light duty vehicles. The Table 20 also shows that the VSP mode exceeds the maximum range of VSP i.e. 39 KW/tonne after 2% grade and stays unchanged thereafter. However, the vehicle specific power continues to increase. The resulting emission rates are used in estimating the total vehicle emissions that will be utilized in modeling the pollutant dispersion.

Table 20 Emission Rates for Different Gradients

Grade (%)	**VSP (KW/tonne)**	**Emission Rate (gm/sec)**	**VSP Mode**
0	29	12.1	12
0.5	32	12.1	12
1	34	15.5	13
1.5	36	15.5	13
2	41	17.9	14
2.5	43	17.9	14
3	45	17.9	14
3.5	48	17.9	14
4	50	17.9	14
4.5	52	17.9	14
5	52	17.9	14

After determining the emission rates, the change in dispersion concentration due to gradient is analyzed. If the gradient is changed, the case of line source changes to a case of continuous series of elevated point sources. The basic formula for dispersion in air, which is described in Chapter 2 (Eq. 1), is used for this type of source. By changing the case scenario, the dispersion pattern of the air pollutant is changed from Poisson to Normal distribution. This occurs when a highway section is grade-separated or is situated in rolling or mountainous terrain.

The unit grade-specific emission rates for light duty vehicles are used to find the total emission rate input. The stability conditions, wind speed and wind direction are the same as in the previous case. The additional inputs used in this model are stack height, stack diameter, gas exit temperature, and ambient temperature. The stack height is calculated from vertical curve design:

$$Y = AL/200 \qquad \text{Eq. 24}$$

where Y is the offset at the end of the vertical curve in ft (m), A is the absolute difference between slopes $|G_1\text{-}G_2|$, and L is the length of the curve. The stack height for 1% grade and 1 kilometer length of roadway is 5 meters. The stack diameter is 7.3 meters (24 ft) which is the width of a two lane road. The gas exit temperature is 241^{o}C (Fournier and Kennerly 2008). The ambient temperature is 20^{o}C, i.e. the average temperature in and around Baltimore. The resulting dispersion concentrations from these inputs are shown in the Figure 12. This figure shows the effect of gradient on air pollutant dispersion through space. The dispersed pollution concentrations show slight decrease with increase in gradient. The concentration of pollutant observed for 0% grade is 298 mg/m^3 at 7.5 meters from the source, which exceeds the concentration for 5% grade at 7.5 meters, i.e. 167 mg/m^3.

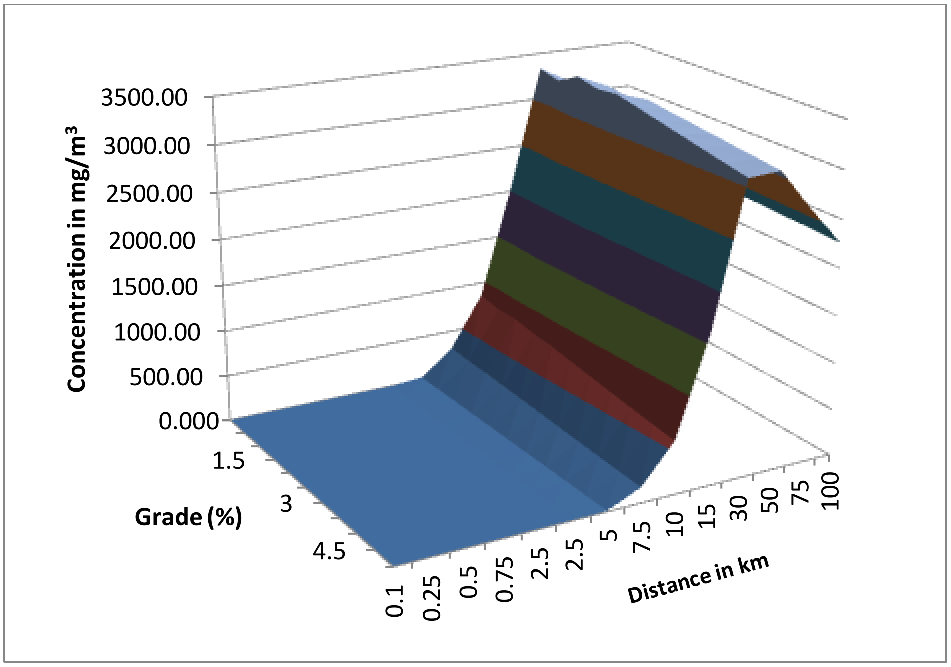

Figure 12 Change in Concentration with respect to Gradient and Downwind Distance

The distance covered by the pollutants in this case is much higher than that covered in simple line model. This result shows that the emitted particles can travel more than a hundred kilometers. This source is approximately 5 meters above the ground. Here, the pollutants cover approximately 5 km at the stack height and then start scattering. The reason for such a different pattern is that the point source model employs effective stack height that is based on Brigg's parameters discussed in theChapter 2: Literature Review. This height takes into account the effect of stack height, wind velocity and atmospheric stability, gas exit velocity gas exit temperature and ambient temperature in determining the spatial extent of the pollutants. This is the reason why pollutants cover a large distance before scattering particles into the space described by a Gaussian distribution.

4.6 Population Density

The population density affects the total health cost. A highway located in a densely populated area will have more people exposed to the pollutant. That results in more illness cases per square kilometers and higher total health cost due to the highway. The monetary cost variation with different categories of population density is presented in Figure 13. This figure shows the rate of hospitalization and emergency visits for low (100), medium (250) and high (500) population densities in persons/square kilometers. The monetary cost due to illnesses increases when the corresponding population density is high. The percentage increase in monetary cost depends on the unit cost of illness, the age of population and considered illness incidences.

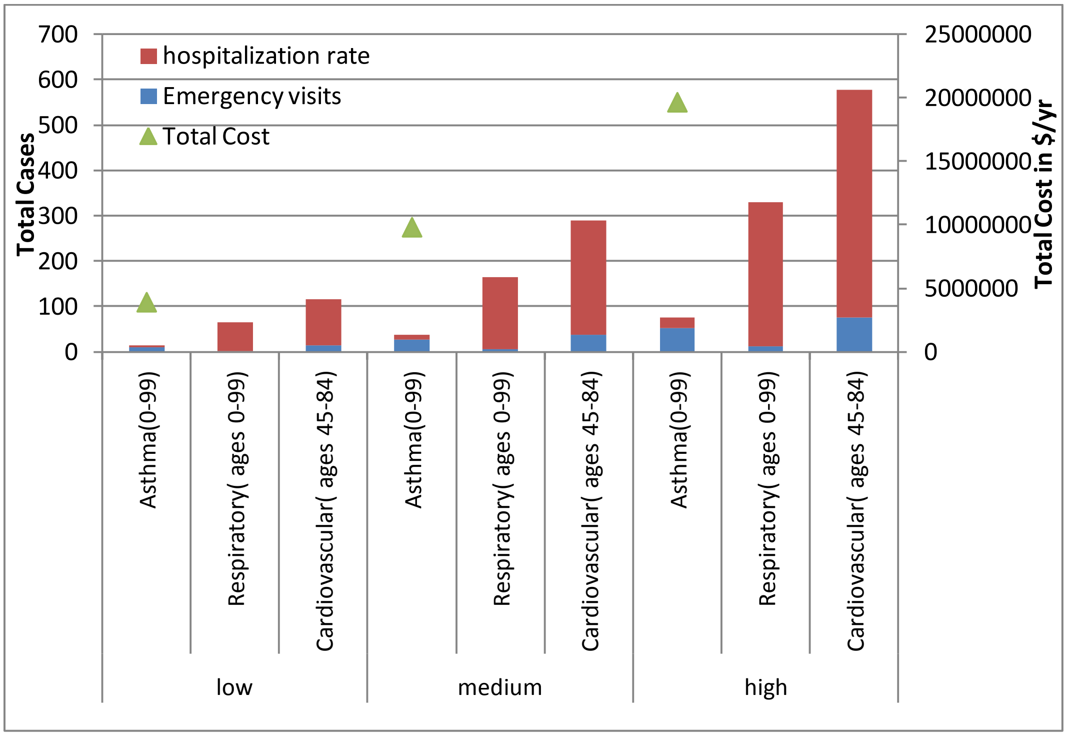

Figure 13 Total Cost with respect to Population Density

Chapter 5: Conclusions and Recommendations

The environmental costs of a new highway are important considerations in the evaluation process. The current practice in environmental impact analysis is to estimate the traffic emissions using an approved regulatory model and compare with the set standards. This regulatory method neglects the impact of pollutants such as increase in illnesses and health problems and instead relies on the standards values. That is why the task is undertaken to develop a methodology that is capable of estimating detailed environmental impacts of a new highway in an area. This task has been achieved by combination of dispersion and exposure modeling techniques used in environmental, epidemiology and public health areas. The main contribution of this study is to formulate an evaluation methodology for environmental impacts that integrates different steps which has been developed in previous studies, and adapt them for the highways.

The estimate of environmental health impacts starts where the emission estimate ends. The emission rate estimate is used in dispersion modeling. This dispersion concentration is used in estimating the health effects on the population in the vicinity of a proposed highway. The dispersion modeling approach varies with source type and surrounding area conditions. The dispersion model depends on a number of factors such as traffic volume, atmospheric stability, wind direction, and wind speed. A simple line model is used after trying a newer model for a highway source in rural area. The new model results show a composite distribution for pollutant dispersion which is unlikely and that is why simple line model is used.

The health risks are calculated in terms of population exposure. The population exposure modeling techniques differ with the type of health risk indicator such as mortality, illnesses, and morbidity. The health indicator plays an important role in estimation method of the health effects. The mortality risk is not estimated for all pollutants. The risk of illnesses is estimated using the data from epidemiological and public health studies. The proposed methodology is demonstrated for a defined case scenario.

The case considers the emissions estimate of a new highway with an existing highway at 1 kilometer from the new highway. The emissions filter through tree coverage before reaching the residential area. The results are shown for the emissions of a new highway and a neighboring highway. The risks associated with the new highway are estimated for premature deaths, hospitalization and emergency visits. The premature death risks are calculated using pseudorandom numbers. The hospitalization rate and emergency visits risks are estimated from the data available in references. The environmental cost in monetary terms is calculated as the sum of hospitalization rate and emergency visits. The estimate of total environmental cost in monetary terms is $0.36 million per year which is significant. The environmental cost for a design life of 20 years is $0.36*20 = $7.2 million. It is noteworthy that this estimate is for a one kilometer length of road.

The cost varies with distance from the highway, wind direction, vehicular traffic on the new and existing highway, and other factors. The analysis of results shows that the concentration is inversely related to the wind direction. The lowest dispersion value is observed for highest value of angle of wind direction. The traffic mix on the new highway changes the emission rate significantly, i.e. the percentage of vehicle type and fuel type induce different results of emission rates. The type of area considered, i.e. urban or rural, changes the modeling approach for air pollutant dispersion modeling. The changes in gradient show a direct relation between the gradient and the vehicle emission rate. And changing the gradient can alter the case scenario for pollutant dispersion modeling. The alteration in case scenario demonstrates change in dispersion concentration from the highway. The pollutant particles can travel distances as large as 100 kilometers when considered as elevated and continuous point source.

For future research work, the modeling approach can be used for new highway benefit cost analysis and for comparing the construction cost and user costs with the environmental costs. The environmental costs estimated here take into account some of the atmospheric conditions, different combinations of meteorological, traffic, and land use conditions will result in different results. The percentages of various vehicle types, presence of other pollutant sources in the area and urban or rural consideration will change the risks associated with air pollutants. A similar study can be conducted for other primary and secondary pollutants.

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