Health Benefits of Traffic-Related Particulate Matter Control Policies: The Case of Bangkok, Thailand

Ying Li

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Approved by:

Douglas J. Crawford-Brown

Richard N. L. Andrews

Richard M. Kamens

Brian J. Morton

Mort D. Webster

ABSTRACT

YING LI: Health Benefits of Traffic-Related Particulate Matter Control Policies: The Case of Bangkok, Thailand
(Under the direction of Douglas J. Crawford-Brown)

Urban air pollution is a major public health concern in some large developing metropolitan areas in Asia such as the capital city of Thailand, Bangkok. Since the 1990s, this area has been suffering from severe ambient particulate matter (PM) pollution mainly attributable to its wide use of diesel-fueled vehicles and motorcycles with poor emission performance. While the Thai government strives to reduce emissions from transportation through enforcing policy measures, the link between specific control policies and associated health impacts is inadequately studied. Furthermore, despite the fact that the actual effects of some mitigating policies, such as the in-use vehicle inspection and maintenance (I/M) programs, may be greatly uncertain, this uncertainty issue has traditionally been ignored in evaluating the impacts of a policy on public health.

This dissertation estimated the health benefits potentially achieved by the new PM-related I/M programs targeting all diesel vehicles and motorcycles, as well as several vehicle retrofitting or repowering programs in the Bangkok Metropolitan Region, linked the levels of PM emission reductions available from the I/M programs with the health benefits and costs of the programs, and quantified the impacts of key I/M design elements on emission reductions. The benefits were estimated by using the health benefit analysis framework that integrates air quality modeling, exposure assessment, exposure-response assessment and economic valuation, and the effectiveness of an I/M program was evaluated by utilizing a new analysis approach called "I/M Design". In addition, a stratified meta-analysis was conducted to develop a mobile source specific PM concentration-response coefficient appropriate for this study.

The results indicated that traffic-related PM control policies potentially yield substantial health benefits relative to the business-as-usual scenario. Particularly, it was found that with a high level confidence, the I/M programs will produce total health benefits that outweigh the large expenditures on policy implementation. Nevertheless, the uncertainty in the effectiveness of such programs needs to be taken into account so that the associated health improvement is appropriately evaluated. Finally, I/M design considerations should address some key design elements such as the problem vehicle identification rate in order to improve the effectiveness of the programs.

Dedicated to my mother, Fanghua Zheng, and to my father, Zhensheng Li,
for their love and support.

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LIST OF ABBREVIATIONS

BAU Business as usual

BMA Bangkok Metropolitan Administration

BMR Bangkok Metropolitan Region

CBA Cost-Benefit Analysis

CDF Cumulative distribution function

CEA Cost-Effectiveness Analysis

CI Confidence interval

CO Carbon monoxide

COI Cost of illness

CR Concentration-response

EIA (United States) Energy Information Administration

EPA (United States) Environmental Protection Agency

GDP Gross Domestic Product

HBA Health Benefit Analysis

HIA Health Impact Assessment

I/M Inspection and maintenance

NOx Oxides of nitrogen

O_3 Ozone

PCD (Thailand) Pollution Control Department

PDF Probability density function

PM Particulate matter

$PM_{2.5}$ Particulate matter having an aerodynamic diameter of less than or equal to 2.5 micrometers

PM_{10} Particulate matter having an aerodynamic diameter of less than or equal to 10 micrometers

SO_2 Sulfur dioxide

VOCs Volatile organic compounds

WTP Willingness to pay

Chapter 1 Introduction

1.1 Urban air pollution and its health impacts in developing countries

Air pollution has been a public health concern for several decades. In particular, in the world's megacities, generally defined as metropolitan areas with populations exceeding 10 million inhabitants, air pollution has become one of the most crucial problems due to the rapid growth of urban areas and the concentrations of people and activities (Molina & Molina, 2004). Air pollution has been implicated in widespread adverse health outcomes in these areas such as premature death, prenatal death and a variety of diseases. The problem is even worse in developing countries, where urbanization is progressing much faster than in developed countries, whereas the technology and pollution control in energy sectors are usually inferior (Katherine & Tatyana, 2000). Available data indicate that in mid-1990s, the ambient concentrations of pollutants in several developing megacities were a few times higher than the World Health Organization's (WTO) air quality standards (Katherine & Tatyana, 2000).

Fuel combustion by motor vehicles is one of the largest and fastest growing sources of air pollution in these developing urban areas. Automotive emissions are considerable health threats to local residents because they are emitted at the ground level. Despite the fact that motor vehicles are much more common in developed than in developing countries on the basis of the number of vehicles per capita as well as more common use of public transportation by the residents of developing nations, air pollution problems caused by motor vehicles in developing nations are even worse because vehicles in these regions tend to concentrate in a few large cities (Katherine & Tatyana, 2000). And some common problems include old vehicles with extremely high exhaust emissions, unpaved or poorly maintained roadways that suspend dust, insufficient enforcement of emission standards, congested roadways and public transportation systems as well as misuse of fuels (Chow et al, 2004). Along with the rapid urbanization and economic development in many large urban areas of the developing world, the change of residents' lifestyles toward greater mobility and the escalating demand for motor vehicles are also noticeable, exerting increasing stress on the ambient environment and public health. Therefore, control policy measures are imperative in these areas in order to tackle air quality problems and ensure sustained future development. At the same token, the scarcity of resources in developing countries makes it more crucial to set policy priorities that focus on strategies with high effectiveness but lower costs.

1.2 The case of Bangkok, Thailand

It is evident that air pollution has caused serious health damage throughout South Asia, and Thailand's capital city Bangkok stands out as one of the worst areas. While most of the growth in Thailand over the last thirty years was driven by increasing economic activities in and around the Bangkok area (Parsons International Ltd, referred to as Parsons later, 2001), this area, where approximately 10 million people live, has also been suffering from severe adverse health effects attributable to air pollution, and consequent economic loss as well. For instance, the U.S. Environmental Information Administration reported that in 2001, airborne particulate matter was estimated to have caused 3,300 premature deaths and to have led to almost 17,000 hospital admissions in Bangkok, at a total health care cost of up to $6.3 billion (Energy Information

Administration [EIA], 2003).

The Thai government is hoping that a stronger commitment to environmental protection will help the country cope with the environmental challenges that it now faces (EIA, 2003). Over the past decade, a series of programs were initiated to curb air pollution, particularly in urban areas. With the government's efforts, remarkable progress has been made in improving the country's ambient air quality. A World Bank report said that the ambient levels of key pollutants, including lead[1], particulate matter, sulfur dioxide and carbon monoxide, in Bangkok and other urban centers have fallen dramatically, and with the exception of particulate matter and ozone, all pollutants comply with the country's air quality standards by the time reported (World Bank, 2002). However, it was also reported that the ambient concentrations of particulate matter, the air pollutant of the greatest health impact, still remains high and exceeds Thailand's National Ambient Air Quality Standards (NAAQS)[2], in particular along traffic corridors in urban areas like Bangkok (World Bank, 2002).

Traffic congestion and poor air quality have traditionally been the major impediments to Bangkok's long-term sustainability (Parsons, 2001), resulting in damages to the economy and deterrent to tourism and foreign investments. Studies indicated that the major sources of air pollution in Bangkok are mobile sources, and among many pollutants emitted from motor vehicles, particulate matter is the most critical one (Thongsanit et al, 2003). Like in many large urban areas in developing countries, the slow development of transportation infrastructure in the city does not keep pace with the rapid increase in travel demand as the consequence of the growth in population and increase in income levels. Ubiquitous traffic congestion problems in this area are widely recognized. At the same time, old vehicles with higher exhaust emissions and unpaved or poorly

[1]Thailand has phased out leaded gasoline by January 1996.

[2]Thailand's NAAQS for PM_{10} are: daily -- 120ug/m^3; and annual -- 50ug/m^3 (World Bank, 2002).

maintained roadways that suspend dust are responsible for a substantial proportion of total emissions from mobile sources. To tackle these problems, a variety of policies targeting emissions control in the transport sector have been introduced or proposed in Thailand, while policy design and implementation still need to improve so as to ensure the effectiveness of policies. Controlling emissions from the transport sector is the focus of this research.

1.3 Study motivations

Since air pollution control usually imposes substantial costs on a society, an understanding of the link between specific control policies and associated health risk reductions as well as benefits would provide valuable information to decision-makers. While currently studies on this topic are not adequate in Thailand as well as many other less developed countries, and thus it is the objective of the present study to provide a better understanding of this topic through in-depth analyses, this study also aims at developing a general framework that quantifies the health impacts of an energy use sector such as transportation, and a framework that can be used elsewhere, in particular large developing urban areas with concentrated population and economic booming, to evaluate the benefits of air pollution control policies. A fundamental assumption of this framework is that the primary benefits associated with air pollution control are human health benefits, and the ability to reduce health risk is one of the major decision criteria in setting policy priorities. The rationale of this assumption lies in the assertion that it is important in developing countries to include human health protection as a key criterion when forming urban development policies and to make it an integral part of the public policy-making process (Li et al, 2004).

More importantly, a review of existing health benefit analysis literature reveals that an important uncertainty issue, namely, the uncertainty about the actual effects of pollution mitigating policies, has traditionally been ignored. By assuming full implementation of a policy measure, an evaluation may considerably overestimate the health benefits achieved by that policy, or simply shift the focal point of decision-making processes, if the emission savings are in fact considerably uncertain for decision makers or likely to be far less than anticipated. Consequently, some control measures, such as the vehicle inspection and maintenance (I/M) programs investigated in the present study, which have been traditionally ranked superior among various vehicle emission control measures by the results of Cost-Benefit Analysis (CBA) or Cost-Effectiveness Analysis (CEA)[3] may be suboptimal if the uncertainty issue mentioned above is taken into account. Given this, beyond a quantitative understanding of the potential health benefits of vehicle emission control policies in heavily polluted developing urban areas, further questions to be addressed here are how various program design considerations (e.g. participation levels or rigorousness) might affect the associated heath outcomes, and what are the minimum implementation requirements that at least ensure the benefits are greater than the costs of implementing the programs.

In this study, health benefits are evaluated in a classical health benefit analysis framework that integrates air quality modeling, exposure assessment, exposure-response assessment and economic valuation. And a new policy evaluation framework is employed and linked to the health benefit analysis framework to address the second policy question. The control policy measures studied here include new particulate matter (PM) oriented I/M programs targeting all diesel-fueled vehicles and motorcycles, and heavy-duty diesel vehicle retrofitting or repowering programs. Given the fact that in developing nations,

[3]In the air quality management field, a cost-benefit analysis usually calculates the net benefits or benefit-cost ratios of various emission control policy measures, whereas a cost-effectiveness analysis calculates the costs of unit emission reduced by various measures. Both methods can be used to compare and rank different control measures.

vehicles are often poorly maintained and there are still enormous old vehicles running on roads, these policy options have been proposed as imperative solutions to the air pollution problem resulting from mobile sources in these regions in addition to setting new vehicle emissions standards and enforcing fuel quality regulations.

Chapter 2 Literature Review

2.1 Particulate matter (PM) and its health effects

2.1.1 Pollutant of focus: particulate matter

Particulate matter is regarded as the key air pollutant linked with detrimental health effects because scientific studies have generally shown that PM is responsible for the largest attributable fraction of mortalities due to air pollution exposure (Wang & Mauzerall, 2006). In health impact assessment, there may be no need to include other air pollutants from mobile sources such as carbon monoxide (CO), hydrocarbon (HC), nitrogen oxides (NO_x) as the concentrations of these pollutants are often correlated with PM, and thus epidemiological studies cannot strictly allocate observed effects to single pollutants, but the synergy effects between PM and other pollutants instead (Kunzli et al, 2000). The inclusion of the impacts of all pollutants individually would grossly overestimate the health impacts (Kunzli et al, 2000). Another pollutant of health concern is ground-level ozone (O_3) resulting from chemical reactions between NO_x and HC in the presence of sunlight and warm temperatures. In the Bangkok area, ozone is currently not an air pollution problem (Parsons, 2001). Given this evidence, this study focuses solely on PM emitted from mobile sources. On the other hand, we may underestimate the total health benefits from pollution control in transportation sector without taking other pollutants into account. With this respect, the results of this study are considered as conservative estimates of the total benefits of emission control.

2.1.2 Definition and categories of particulate matter

Particulate matter is the term used for a mixture of tiny solid particles and liquid droplets found in the air (EPA, 1997a). Airborne particles come in all sorts of sizes, shapes, compositions and origins (Brunekreef & Forsberg, 2005). Some particles are large enough to be seen as dust or dirt, while others are so small that they can be detected only with an electron microscope. The range of particle sizes of concern for air emission evaluation is quite broad, and particles of different sizes behave differently in the atmosphere and in the respiratory system (EPA, 2006). The following table summarizes categories and definitions of particles used by the U.S. Environmental Protection Agency (EPA):

Table 2.1 U.S. EPA's Categories of Particles

Category	Definition
Total Suspended Particulate Matter (TSP)	Particles ranging in size from 0.1 micrometer to about 30 micrometer in diameter
PM_{10}	All particles having an aerodynamic diameter of less than or equal to 10 micrometers
$PM_{2.5}$	All particles having an aerodynamic diameter of less than or equal to 2.5 micrometers
Particles less than 0.1μm	Defined as the name
Condensable Particulate Matter	Particulate matter that forms from condensing gases or vapors by chemical reactions as well as by physical phenomena

Source: EPA, 2006

Another useful terminology differentiates "fine particle" and "coarse particle". The former usually refers to those in the range of 0.1 to 2.5 micrometers, and the latter refers to those in the range of 2.5 to 10 micrometers (EPA, 2006). Current regulation in the U.S. is primarily interested in those "inhalable particles", namely, PM_{10} and $PM_{2.5}$

(U.S. EPA, National Ambient Air Quality Standards[4]), because particles less than 10 micrometers are considered to pose a particularly greater risk to health due to their potentially long airborne retention time and the inability of the human respiratory system to defend itself against particles of this size (Dockery et al, 1993). On the contrary, those "coarser" particles ranging in size from 10 to about 30 micrometers in diameter are regarded not to be inhalable or can be exhaled quickly. In addition, most of these particles will deposit onto the ground quickly. For these reasons, they are less likely to be detrimental to public health. In 1987, EPA replaced the earlier total suspended particulate (TSP) air quality standard with a PM_{10} standard (EPA, 1996).

Inhalable particles are the focus of this study. Although the existing epidemiological studies suggest that the severity of the health effects of $PM_{2.5}$ may be much greater than that of PM_{10}, PM_{10} health effect studies are much more prevalent to date because PM_{10} concentrations are easier to measure in the ambient air (Li et al, 2004). As for the study country, Thailand, data are available mostly for PM_{10}. Given these reasons, this study employs PM_{10} as the pollution indicator.

2.1.3 Sources of particulate matter

Particles are emitted from a variety of both natural and anthropogenic sources, and particles from different sources differ in their size and composition. Examples of natural sources include evaporated sea spray, wind-borne pollen, windblown dust and volcanic or other geothermal eruptions (World Bank, 1999). PM from natural sources tends to be coarse, whereas almost all fine particulates are generated as a result of combustion processes, including the burning of fossil fuel for steam generation, heating and

[4]Available on U.S. EPA's official website: http://www.epa.gov/air/criteria.html#1, accessed October 12, 2006.

household cooking, agricultural field burning, engine combustion, and various industrial processes. Emissions from these anthropogenic sources tend to be in fine fractions, i.e. less than 2.5 micrometers in diameter and include a larger variety of toxic elements than particles emitted by natural sources. The particles may be primarily carbonaceous in nature, if they originate from combustion activities or mineral (Li et al, 2004). In addition, some particles are emitted directly from their sources (e.g., smokestacks and cars), and these are referred to as “primary” particles, whereas others are formed in the atmosphere from gases such as sulfur dioxide (SO_2), NO_x, and volatile organic compounds (VOCs) and other compounds in the air (Christakos & Serre, 1999). These are referred to as “secondary” particles. The sources and species characteristics of PM may vary significantly across locations, and the physical and chemical composition of PM emissions is determined by the nature of pollution sources. For instance, in a semi-arid area like Spokane, Washington, windblown dust contributes to a substantial fraction of the total PM concentration (Clainborn et al, 2000). In cold and temperate parts of the world, domestic coal burning for winter heating can be a major contributor to the particulate content of urban air. Traffic related emissions might make a substantial contribution to the concentrations of suspended PM in areas close to traffic. We need to understand the relative contributions from different sources in an area to set policy priorities and design cost-effective control strategies.

Studies consistently find that motor vehicles are the major sources of PM emissions in the Bangkok area, in particular, particles from heavy and light duty diesel-fueled vehicles and two-stroke motorcycles (Parsons, 2001). The following table summarizes the results from the most recent study of PM air emission sources in the Bangkok Metropolitan Region (BMR) by the Pollution Control Department (PCD) of Thailand.

Table 2.2 Particulate Matter Emission Rates and Proportion from Various Emission Sources in the Bangkok Metropolitan Region

Emission Source	1997		2002	
	Emission Rate (tons/year)	Percentage of Emission Rate (%)	Emission Rate (tons/year)	Percentage of Emission Rate (%)
Point Source	3,735	9.78	3,246	8.84
Mobile Source	20, 602	53.94	19,735	53.77
Area Source	13,855	36.28	13,724	37.39
Total	**38,192**	**100.00**	**36,705**	**100.00**

Note: Data source: Pollution Control Department, 2000. Point sources included industrial factories, funeral pyre and municipal incinerators; Mobile sources considered motor vehicle types including: large diesel vehicles, small diesel vehicles, gasoline vehicles and others; area sources included airport, petrol service stations, community or residential area, and industrial area. The 2002 data were predicted based on the 1997 data by taking various factors into account such as energy policy, changes in fuel composition and traffic.

Another study (Parsons, 2001) provided information on the share of total PM emissions by different vehicle types as follows:

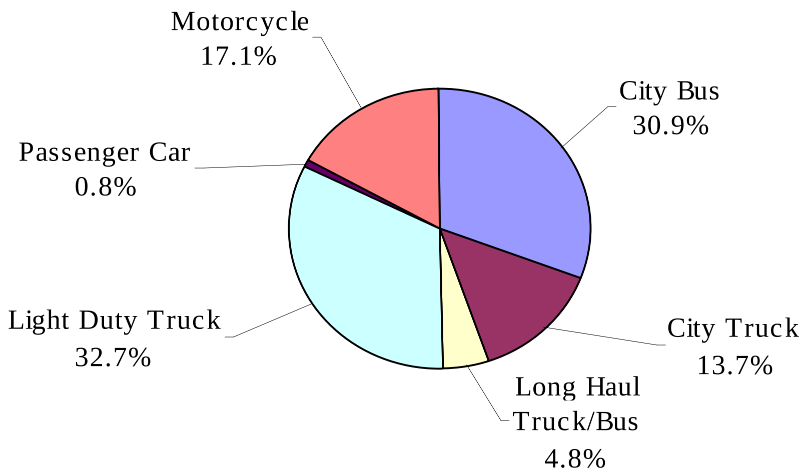


Figure 2.1: Distribution of Mobile Particulate Matter Emission Sources in the Bangkok Metropolitan Region (Source: Parsons, 2001)

These results illustrate that diesel-fueled vehicles (including various trucks and buses), along with motorcycles, consist of the vast majority (all together more than 99%) of traffic-related PM emissions sources in the study area.

2.1.4 Health effects of particulate matter

2.1.4.1 Overview

The casual relationship between air pollution and health effects is the primary justification of the investment in pollution control. Airborne particulate matter is considered as the air pollutant of the greatest health impacts due to the factor that it induces a series of adverse health effects, in particular premature mortality. Because these tiny particles are inhalable and they easily reach the deepest recesses of the respiratory systems of human being, scientific studies have linked breathing PM, particularly fine particles (alone or in combination with other air pollutants), with a variety of significant health problems. The U.S. EPA summarizes these detrimental health effects as follows (EPA, 1997a):

- Premature death;
- Respiratory related hospital admissions and emergency room visits;
- Aggravated asthma;
- Acute respiratory symptoms, including aggravated coughing and difficult or painful breathing;
- Chronic bronchitis;
- Decreased pulmonary function that can be experienced as shortness of breath; and
- Work and school absences.

And susceptible subjects include the elderly, children and individuals with preexisting heart or lung diseases. The Executive Summary of the most recent *Air Quality Criteria for Particulate Matter* states that "Associations of both short-term (usually days) and long-term (usually years) PM exposure with most of these endpoints have been consistently observed", and that "The general internal consistency of the epidemiologic database enhances the confidence accorded the reported results and has contributed to increasing public health concern" (EPA, 2004).

Concentration-response (CR) functions relate the change in the incidence of adverse health effects in a population to the change in pollutant concentrations experienced by that population (Ostro et al, 2006)[5]. Estimating CR functions represent a crucial step in health impact assessment and may involve significant uncertainty. Studies have relied on epidemiological studies for CR functions appropriate for benefit evaluation. Besides air pollution, human health conditions are potentially affected by a variety of factors, such as lifestyle (e.g. smoking habits, activity patterns, etc), weather, season, day of the week, working and living environment, pre-existing health conditions and health care status, infectious disease epidemics, and socioeconomic features. In general, epidemiological studies employ sophisticated regression techniques to isolate the impact of air pollution on health from that of others.

A large number of epidemiological studies on the associations between PM exposure and adverse health effects including mortality and morbidity have been conducted from around the world over the last two decades, mainly in the western developed countries but increasingly in less developed world. Dockery et al (1993) is a landmark study. Started in the 1970s, this study estimated the effects of air pollution on mortality by analyzing data from 14-to-16-year mortality follow-up of 8111 adults of six cities in the Northeastern and Midwestern U.S. They reported that long-term exposure to ambient fine particles ($PM_{2.5}$) was significantly associated with both all-cause mortality

[5]Another related term is the "exposure-response function", referred to as the relationship between the level of exposure to a pollutant and the probability and/or severity of the effect. Epidemiological studies on the health effects of PM considered here generally rely on ambient PM concentrations rather than concentrations of actual individual exposure to estimate the associations of PM exposure with adverse health effects. Therefore, it is more appropriate to use the term "concentration-response" for these estimates.

and cardiopulmonary-cause mortality. This study improved a few earlier cross-sectional studies on the association between mortality rates and particulate air pollution in the U.S. metropolitan areas (e.g. Evans et al, 1984, Ozkaynak et al, 1987) by directly controlling for the confounding effect of cigarette smoking as well as other health risk factors. Despite some limitations (e.g. residual confounding, statistical methods, and the lack of toxicological evidence), the consistency of findings by this study is remarkable and suggestive.

In general, epidemiological studies may be conducted in three ways (Crawford-Brown, 1999; Rothman & Greenland, 1998): (1) Ecological studies (including cross-sectional studies across different sites and time-series semi-ecological studies within a given site); (2) Cohort studies (usually classified into retrospective and prospective studies based on the temporal pattern of data collection); and (3) Case-control studies. The first two are the most commonly used strategies in the study of health effects of particulates, in particular, time-series studies. They are surveyed in the succeeding sections.

2.1.4.2 Time-series studies

(1) Mortality

Compared with other types of studies, time-series analysis is the most commonly used method of analysis. Numerous time-series studies on the associations of particulate air pollution with adverse heath effects have been performed around the world, from developed to developing countries and across a wide range of climate conditions. The most commonly used study region is a city, and fewer studies were also conducted in a larger region such as a state or a country. A full list of time-series studies on this topic can be found in databases like the EPA Criteria Document (EPA, 2004), Medline (National

Library of Medicine), and Current Contents (Institute for Scientific Information). Time-series studies usually examine the relationship between daily levels of ambient PM concentrations (24-hour average concentrations are used in most cases) in an area (a city, a county or others, assuming that the concentration at a time is identical across the study area) and daily occurrence of mortality or morbidity, by employing statistical techniques. The study population can be the general population exposed to the pollutant (e.g. Ostro, 1995; Lippmann et al, 2000; Goldberg et al, 2001; Gwynn et al, 2000; Simpson et al, 2000; among others), or some susceptible groups such as children (e.g. Conceicao et al, 2001; Zhang et al, 2002; Schwartz & Neas, 2000; Ransom & Pope III, 1992; among others) and the elderly (e.g. Rodes et al, 2001; Fischer et al, 2003; Liao et al, 1999; among others).

The key advantage of the time-series method is that it potentially reduces the confounding effects of many factors, which otherwise might be difficult for researchers to control. Specifically, several important confounding factors, such as smoking habits, health care status, activity patterns, working and living environment as well as socioeconomic status, do not vary considerably over time. Beside this, the data required for time-series studies are often more readily accessible, whereas a cross-sectional study requires information from an adequately large number of places, and a prospective cohort study requires continuously following a group of subjects for a long study period. However, the weakness of time-series analyses is that they can solely capture short-term health effects by examining the covariates of daily levels of PM concentrations and health endpoints. Therefore, they are less useful for investigating chronic effects.

Previous time-series studies generally observed significantly positive associations between particulates and adverse health effects (mortality or morbidity). However, the concentration-response relationship estimated by different studies has differed substantially. For instance, Levy et al (2000) surveyed 18 relevant studies and found that their estimates of the changes in daily mortality associated with a 10-$\mu g/m^3$ increase in PM_{10} ranged from –0.5% to 1.6% (two studies reported negative associations, whereas the other sixteen reported positive associations). EPA's criteria document summarizes that the changes in total non-accidental mortality associated with the same level of increase in 24-hour average PM_{10} range from 0.3% to 1.6% (EPA, 2004). Although some researchers argued that it was difficult to see the consistency of the magnitude of estimated health effects among different studies, we must admit that besides air pollution, people's health can be affected by a number of factors such as meteorological conditions, socioeconomic status (e.g. race, gender, education, income, etc), psychological conditions like stresses, underlying health status and genetics, use of health care, activity patterns and so forth. At the same time, the epidemiological studies differ in a number of ways such as research design and statistical methods. The variability in findings could be a function of site-specific differences, analytical decisions, or simply random variations (Levy et al, 2000).

In recent years, meta-analytic techniques, which involve quantitatively combining results from related but independent studies for the purpose of integrating findings, have been developed and increasingly applied in environmental health field to handle the heterogeneity problem. It is argued that compared with traditional reviews, meta-analysis provides a more objective summary of evidence, and it might be particularly useful to formally examine sources of heterogeneity and to clarify the relationship between environmental exposures and health effects (Blair *et al*, 1995). Although it still subject to controversy, meta-analysis is a useful tool to integrate results from a large collection of individual studies and to explore sources of heterogeneity. The following table summaries recent meta-analyses of time-series studies (studies carried out after 2000) on the associations between daily mortality and PM_{10} levels.

Table 2.3 Recent Meta-Analyses of Time-series Studies on the Associations Between Daily Mortality and PM_{10} Levels

Publication (year)	Study area	% Change in daily all-cause mortality per 10-µg/m^3 increase in PM_{10} (Mean estimate, 95% CI)	% Change in daily respiratory mortality per 10-µg/m^3 increase in PM_{10} (Mean estimate, 95% CI)	% Change in daily cardiovascular mortality per 10-µg/m^3 increase in PM_{10} (Mean estimate, 95% CI)
Braga et al (2000)	5 U.S. cities	0.78% (0.51-1.05%)		
Levy et al (2000)	19 U.S. studies	0.70% (0.54-0.86%)		
Schwartz (2000)	10 U.S. cities	0.67% (0.52-0.81%)		
Stieb et al (2002)	109 studies from around the world	0.64% (0.48-0.76%)		
Anderson et al (2004)	33 European studies	0.60% (0.40-0.80%)	1.3% (0.5-2.0%) (18 studies)	0.9% (0.5-1.3%) (17 studies)
Dominici et al (2005)	90 U.S. cities	0.21-0.41% depending on the model used		
Zeka et al (2005)	20 U.S. cities	0.52% (0.31-0.72%)		
Samet et al (2000)	20 U.S. cities	0.51% (0.07-0.93%)	0.68% (0.20-1.16%) (Two causes combined)	
Lvovsky et al (2000)	6 cities from around the world	0.84%		
Aunan and Pan (2004)	6 Chinese cities	0.3%	0.4%	0.6%

All these meta-analyses observed positive association between acute exposure to PM_{10} and daily mortality. Most meta-analyses were conducted in the U.S. and European countries. The study by Aunan and Pan (2004) was the only meta-analysis focusing on an Asian country. The magnitude of health effects (increase in daily all-cause mortality) approximately falls into the range of 0.5-1.0% for a 10-µg/m^3 increase in ambient concentrations of PM_{10}. The National Morbidity, Mortality and Air Pollution Study (NMMAPS) funded by the Health Effects Institute included the largest number of cities in one study (Dominici et al, 2005; Samet et al, 2000). They also controlled for the effects of other combustion-related criteria pollutants including O_3, NO_2, SO_2 and CO. Using advanced statistical techniques, they estimated the mortality risk associated with PM_{10} as well as other pollutants in 20 of the largest U.S. cities and extended the analysis to other 90 U.S. cities. The results of both the 20 cities and 90 cities analyses are generally consistent with an average approximate 0.5% increase in overall mortality for every 10-µg/m^3 increase in PM_{10} measured the day before death. In a revised analysis of the 90-city database, Dominici et al (2005) discovered smaller effects (varied from 0.21% to 0.41% increase in overall mortality for every 10-µg/m^3 increase in PM_{10} depending on models used) when more stringent convergence criteria were used. Nevertheless, the key scientific finding in earlier studies that there exists an association of PM_{10} with daily mortality 1-day later and that the association could not be attributed to any of the other pollutants being studied was confirmed in all these analyses. Compared with the studies in the U.S. and in Europe, Aunan and Pan's (2004) China study reported a lower pooled-estimate of the PM_{10}-mortality coefficient. The authors argued that one possible explanation is that scientific evidence has suggested the concentration-response relationship may become less steep as ambient concentration levels rise.

A review of more recent meta-analyses indicates that the PM-mortality concentration-response coefficients reported in these studies tend to be lower than those observed in earlier time-series studies. For instance, an earlier systematic review by Dockery and Pope (1996) reported an approximate 1% increase in daily mortality occurred for every 10-µg/m^3 increase in PM_{10}. The decrease in the estimates of effects might be the results of the development of better statistical techniques, decrease in measurement errors, and the use of more stringent criteria to evaluate studies.

(2) Morbidity

Compared with studies on the PM-mortality relationship, studies on the associations between PM and morbidity are much less comprehensive (Wang & Mauzerall, 2006). It is perhaps due to the difficulties in collecting accurate and sufficient data of illnesses. Most studies investigated the relationship between levels of PM and hospital admissions for certain types of diseases. Also, many studies focused on more susceptible subjects such as the elderly or children. The following table lists three recent meta-analyses of time-series studies on the associations between PM_{10} and daily morbidity.

Table 2.4 Recent Meta-Analyses of Time-series Studies on the Associations Between PM_{10} and Daily Morbidity

Publication (year)	Study area	% Change in daily hospital admissions per 10-µg/m^3 increase in PM_{10}	Age group
Samet et al (2000)	14 U.S. cities	1.19% (0.97-1.41%) CVD* 2.45% (1.75-3.17%) COPD** 1.9% (1.46-2.34%) Pneumonia	65+
Zanobetti et al (2000)	10 U.S. cities	1.27% (1-1.5%) CVD 2.5% (1.8-3.3%) COPD 1.95% (1.5-2.4%) Pneumonia	65+
Anderson et al (2004)	8 European studies	0.7% (0.2-1.3%) respiratory diseases	65+

* CVD: Cardiovascular disease

**COPD: Chronic obstructive pulmonary disease

The 10 cities studies by Zanobetti et al (2000) were actually included in the 14 cities studied by Samet et al (2000), and their estimates of effects were rather similar. A review of existing epidemiological time-series studies shows that, with a few exceptions (e.g. Tsai et al, 2003; Wong et al, 2002) that reported no significant association or even negative association, studies generally observed positive associations between PM levels and the incidence of respiratory and cardiovascular diseases. The relationship appears to be more pronounced for children and elderly. These findings are consistent with mortality studies.

In summary, a review of epidemiological time-series studies over the last decade suggests that there is a consensus on the positive associations between acute particulate exposure and excess mortality and morbidity, in particular, for respiratory and cardiovascular diseases and related deaths.

2.1.4.3 Cross-sectional studies

Cross-sectional studies compare the rates of detrimental health effects across different areas. The common problem with such a study is that there may be many differences other than exposure to air pollutants, which affect the rates of health effects (Crawford-Brown, 1999). Cross-sectional studies of PM health effects usually compare annual mortality rates with annual average concentrations rather than examining the daily level of incidence and air quality (Ostro, 1993). As such, they may incorporate effects of both acute and chronic exposure to air pollution (Ostro, 1993). Ostro (1993) reviewed three cross-sectional studies conducted in the U.S. After converting their results to a common metric, the health effects were summarized as follows:

Table 2.5 Cross-sectional Studies on the Associations Between Mortality and PM_{10} Concentrations in the U.S.

Publication (Year)	Study area	% change in annual mortality per 10-µg/m^3 increase in annual PM_{10} Concentrations
Ozkaynak and Thurston (1987)	100 metropolitan areas in the U.S.	1.49%
Evans et al (1984)	117 cities in the U.S.	0.84%
Lipfert et al (1988)	872 cities in the U.S.	2.3%

Source: Ostro, 1993

The weakness of these early cross-sectional studies was that they did not control for some important risk factors such as smoking. Nevertheless, they did provide some additional evidence to support the findings from time-series studies.

2.1.4.4 Cohort studies

In cohort studies, a group of initially healthy people is followed for a long period to observe how they develop diseases or die. The study by Dockery et al (1993) in six U.S. cities (Harvard Six Cities Study) followed 8111 individuals over 16 years. They reported that adjusted mortality-rate ratio for the most polluted of the cities as compared with the least polluted was 1.26, which means that after accounting for the effects of other risk factors, the long-term average mortality rates were 26 percent higher in the subjects living in communities with higher levels of PM ($PM_{2.5}$ was the pollutant they studied). As a follow-up of the Harvard Six Cities Study, Laden et al (2006) extended mortality follow-up for 8 years in a period of reduced air pollution concentrations and found an increase of 16% in overall mortality was associated with each 10-$\mu g/m^3$ increase in annual $PM_{2.5}$ levels. Another landmark cohort study is American Cancer Society Study (Pope III et al, 1995, 2002 and 2004; Krewski et al, 2000; Jerrett et al, 2005). Below is a listing of major cohort studies that address long-term exposures to $PM_{2.5}$ and premature death.

Table 2.6 Cohort Studies on the Relationship Between Long-Term $PM_{2.5}$ Exposure and Mortality

Study	Publication	Percentage increase in all cause mortality per 1-µg/m^3 increase in annual $PM_{2.5}$ (Mean, 95%CI)
Harvard Six Cities, original	Dockery et al 1993	1.40% (0.43-2.53%)
Harvard Six Cities, HEI reanalysis	Krewski et al 2000	1.51% (0.54-2.58%)
Harvard Six Cities, extended analysis	Laden et al 2006	1.6% (0.7-2.6%)
American Cancer Society (ACS), original	Pope III et al 1995	0.69% (0.37-1.06%)
ACS, HEI reanalysis	Krewski et al 2000	0.73% (0.41-1.10%)
ACS, extended analysis	Pope III et al 2002	0.58% (0.2-1.1%)
ACS intrametro Los Angeles	Jerrett et al 2005	1.7% (0.5-3%)
Postneonatal infant mortality, CA	Woodruff et al 2006	0.7% (-0.7-2.4%)
Adventist Health Study of Smog (AHSMOG), males only	McDonnell et al 2000	0.91% (-0.21-2.39%)
AHSMOG, females only	Chen et al 2005	4.2% (0.6-9%) for fatal coronary heart disease
Veterans Administration (VA)	Lipfert et al 2006	1.50% (0.48-2.54%)
11 California counties, elderly	Enstrom 2005	0.1% (-0.1-0.3%)
Netherlands, elderly (55-69)	Hoek et al 2002	1.17% (-0.19-3.03%), pollutant measured was black smoke particles[6]
French Air Pollution and Chronic Respiratory Diseases (PAARC) Survey	Filleul et al 2005	0.7% (0.3, 1.1%), black smoke
Cystic Fibrosis Foundation, patients enrolled	Goss et al 2004	2.1% (0.7-3.3%)

Note: the listing of studies was cited from an online document, *Published Studies on the Relationship Between Long-Term $PM_{2.5}$ Exposure and Mortality*, from the website of California Air Resource Board, http://www.arb.ca.gov/research/health/pm-mort/pm-mort.htm, accessed November 21, 2006.

Except for the study by Enstrom (2005), which observed only a slightly elevated relative risk attributable to $PM_{2.5}$ exposure, other studies generally reported increased mortality associated with increase in $PM_{2.5}$ levels. In addition, a recent expert judgment study (Industrial Economics, 2006) provides median estimates, which ranged from 0.7% to 1.6% change in annual, adult, all-cause mortality per 1-µg/m^3 change in annual average $PM_{2.5}$ in the U.S., based on personal interview with 12 health experts who have conducted

[6]In European countries, particles are often measured as black smoke (BS). Black smoke usually refers to particles less than or equal 4.5 micrometer in diameter. It is argued that black smoke may serve as a proxy for diesel exhaust particles (Gotschi et al, 2002).

related research. And experts in this study tended to be confident with a causal relationship between $PM_{2.5}$ and premature death. Note that the study by Jerrett et al (2005) reported an estimate significantly larger than other publications of the ACS study. Given that this study selected subjects from Los Angeles, the authors argued that the chronic health effects associated with within-city gradients in exposure to $PM_{2.5}$ can be even larger than previously reported across metropolitan areas.

Regarding source-specific chronic health risk, two studies examined mortality risk for traffic-related air pollution: Hoek et al (2002) assess the relation between traffic-related air pollutant and mortality in participants of the Netherlands Cohort study on Diet and Cancer, and observed that cardiopulmonary mortality was associated with living near a major road. And Finkelstei et al (2004) studied all natural-cause mortality in relation to residence within 50m of an urban road, and reported an increased risk of mortality for subjects living close to the major road as well as a 2.5 years mortality rate advancement period.

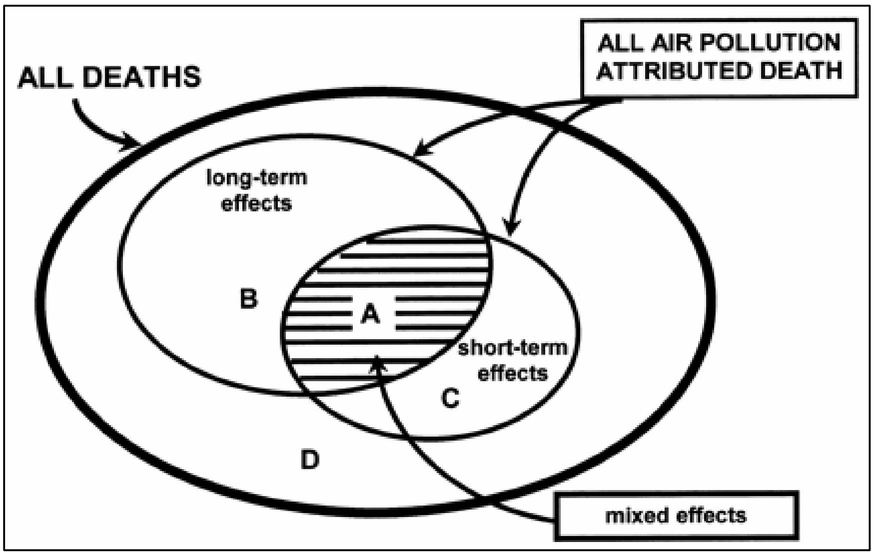
Cohort studies are usually time-consuming and expensive, but they are valuable in exploring chronic effects of exposure to PM over a long period of time. Compared with acute effects observed by time-series studies, long-term chronic effects observed by cohort studies are generally larger. Kunzli et al (2001) reported that the estimates of health effects attributable to air pollution based on cohort studies are 5-10 times larger than estimates based on time-series studies. However, results from studies of health effects of long-term PM exposure are still controversial and less conclusive than associations found in time-series studies, due to the difficulties in following the subjects and controlling various confounding issues over a long period. In addition, although some studies have argued that exposure to PM air pollution is an important environmental risk factor for lung cancer (e.g. Pope III et al, 2002), the effect is not determined yet. At present PM is not regulated as a carcinogen in the U.S.

2.1.5 Key uncertainty issues about the health effects of particulate matter

Although many epidemiological studies have shown significant associations of ambient PM levels with a variety of health endpoints, they can show only statistical associations that may not have causal relationship between exposure to air pollution and adverse health effects (Krupnick & Portney, 1991; Feinstein, 1988). Nonetheless, associations of both short-term and long-term PM exposure with adverse health outcomes have been consistently observed, which enhance the confidence accorded the reported results and has contributed to increasing public health concern (EPA, 2004). However, there still remain great scientific uncertainties regarding the PM health effects. EPA's criteria document states that among these uncertainty issues are "the shapes of PM exposure-response relationships, the magnitude and variability of risk estimates for PM, the ability to attribute observed health effects to specific PM constituents, the time intervals over which PM health effects (e.g., shortening of life) are manifested, the extent to which finding in one location can be generalized to other locations, and the nature and magnitude of the overall public health risk imposed by ambient PM exposure" and "understanding of underlying biologic mechanisms has not yet emerged" (EPA, 2004). In addition, some study has suggested considering the uncertainty in the slope of the concentration-response curve as the most important source of uncertainty in the exposure-health evaluation (Yeh & Small, 2002). In the following discussion, several key uncertainty issues are surveyed.

2.1.5.1 Short-term effects v.s. long-term effects

Although it is generally agreed that the associations of both short-term and long-term PM exposure with adverse health endpoints have been consistently observed, and thus neither of them should be ruled out in risk assessment and management, there is no consensus about their relationship and relative magnitude. Kunzli et al (2001) presents the following conceptual framework to describe the relationship between the deaths attributable to long-term exposures and those attributable to short-term exposures to fine particles:



Note: (1) Source: Kunzli et al, 2001; (2) Circle sizes do not reflect relative effects.

Figure 2.2 Graphic Illustration of Deaths Due to Ambient Air Pollution in a Population

Based on this graph, the authors argued that deaths attributable to air pollution can be categorized into four cases, corresponding to the letters A, B, C and D in the graph: A) air pollution increases both the risk of underlying diseases leading to frailty and the short term risk of death among the frail; B) air pollution increases the risk of chronic diseases leading to frailty but is unrelated to timing of death; C) air pollution is unrelated to risk of chronic diseases but short term exposure increases mortality among persons who are frail; and D) neither underlying chronic disease nor the event of death is related to exposure to air pollution. And the authors also compared the differences of these four categories of cases in the following table:

Table 2.7 A Comparison of the Four Categories of Death Cases Attributable to Air Pollution

Category of Cases	Impact of Air Pollution	
	Underlying Frailty Due to Air Pollution	Occurrence of Death (Event) Triggered by Air Pollution
A	Yes	Yes
B	Yes	No
C	No	Yes
D	No	No

Source: Kunzli et al, 2001.

Overall, it is difficult to derive an estimate of the total annualized mortality effect due to reductions in ambient PM levels that may reflect reductions in both short-term peak and long-term average exposures to PM (Industrial Economics Inc, 2006). Time-series studies analyze the impacts of daily variations in PM concentrations and can characterize the cumulative impact of exposure over a few days, but not over a longer period of time. And uncertainty also remains in the issues such as the time-lag of effects after exposure and the spread period of effects for short-term effects. Different studies have reported different time lag (usually spans from 0 to 5 days) and studies also showed that the effect was spread over several days and did not reach zero until a few days after the exposure (e.g. Schwartz, 2000). A very recent study argued that air pollution exposure with comparable total daily dose may have very different effects when occurring at high levels over a few hours as opposed to low levels over a longer time, implying that the effects of airborne PM on daily mortality may be underestimated, because the daily average describes chronic exposures but does not capture information about acute exposures, that is, large exposures occurring over only a few hours (Staniswalis et al, 2005).

On the other hand, cohort studies focus primarily on analyzing the impact of long-term exposures to PM but may also capture some of the impact of short-term variations in exposure during the cohort follow-up period (Industrial Economics Inc, 2006). In addition, although cohort studies have indicated larger effects on incremental mortality vis-à-vis short-term exposure, the actual effect is still not clear due to some limitations in the study design. For instance, EPA's criteria document argued that the chronic exposure relative risk estimates are based on PM concentration during the 5 or 15 to 20 years study periods and do not necessary reflect the full impacts of longer past PM exposure, which was likely to be much higher in the most highly polluted cities, resulting in overestimates of the relative risk attributable to long-term exposure to PM air pollution (EPA, 2004).

This study considers both the short-term and long-term effects of PM exposure on mortality. However, due to the factor that only short-term (time-series) studies on the association between mortality and PM levels are readily available in Thailand and the transferability of epidemiological studies from developed countries to developing countries is always an issue of concern, this study relies more on time-series studies conducted in Thailand. In sensitivity analysis, long-term effects are included using cohort studies performed in the U.S. and the magnitudes of both effects are compared. With respect to the long-term effects, given that the U.S.EPA recently revoked the annual PM_{10} standard (effective December 17, 2006) due to a lack of evidence linking health problems to long-term exposure to coarse particle pollution[7], only the change in mortality attributable to $PM_{2.5}$ is considered here. Given that $PM_{2.5}$ concentrations were not measured in Thailand for the baseline year 2000, the mass concentrations of $PM_{2.5}$ are

[7]Source: U.S. EPA, National Ambient Air Quality Standards (NAAQS), http://www.epa.gov/air/criteria.html, accessed November 9, 2006.

approximated by abstracting a reasonable ratio of $PM_{10}/PM_{2.5}$ from the literature.

2.1.5.2 Shape of concentration-response functions and threshold

The shape of the concentration-response curve for PM_{10} is another important source of uncertainty. For the health impact assessment and air quality management, one important question is whether there are threshold levels below which pollutants do not cause any harm (Forsberg et al, 2005). For particles, results from time-series studies of daily mortality and hospital admissions generally suggest no threshold for estimating health impacts (Forsberg et al, 2005). Adverse health outcomes can happen at a very low level of PM_{10}. For example, Braga et al (2000) found deleterious health effects of PM_{10} at levels below the air quality standard; Thurston's (1996) review confirmed that an acute pollution-mortality association could occur at routine ambient levels, and suggested that such effects extend below the present U.S. air quality standards, especially for susceptible subpopulations like the elderly and those with preexisting respiratory conditions; Daniels et al (2000) found a linear relation without indication of threshold for PM_{10} and relative risk of death for all causes and the cardio-respiratory cause; Schwartz et al (2002) found the relationship between $PM_{2.5}$ and total mortality persisted down to levels as low as 2 $\mu g/m^3$, with little evidence of a threshold; and Pope III (2000) argued that there was no threshold for exposure-response relation between PM and health effects.

Based on the assumption used by most epidemiological studies, the U.S. EPA suggests an exponential CR function to estimate relations between PM and health effects (Deck et al, 2001)[8]. An exponential CR function is:

$$y = B \times e^{\beta \cdot x} \tag{2-1}$$

[8]The author also noted that the one exception used in the risk analysis was for respiratory hospital admissions, for which a linear CR function was estimated (Deck et al, 2001).

Where x is the ambient PM level, y is the incidence of the health endpoint of interest at PM level x, β is the coefficient of ambient PM concentration (β is also called Poisson regression coefficient given that the Poisson regression is used to compute it), and B is the incidence at $x = 0$, that is, when there is no ambient PM. The change in health effects incidence from the baseline incidence, y to y_0 corresponding to a given change in ambient PM levels, $\Delta x = x_0 - x$, is:

$$\Delta y = y \times (e^{\beta \cdot \Delta x} - 1) \qquad (2\text{-}2)$$

Or, alternatively,

$$\Delta y = y \times (RR_{\Delta x} - 1) \qquad (2\text{-}3)$$

Given that relative risk (RR) is calculated from the Poisson regression coefficient β as follows:

$$RR_{\Delta x} = e^{\beta \cdot \Delta x} \qquad (2\text{-}4)$$

Where $RR_{\Delta x}$ is the relative risk associated with the change in ambient PM level, Δx.

Based on the assumption made in most epidemiological studies, this study applies an exponential CR function with no threshold for the health effects of PM.

2.1.5.3 Sources and components of particles

Particle size has been used to differentiate the toxicity of particulates, given that size is an important determinant of the site and efficiency of deposition (Laden et al, 2000). Fine particles ($PM_{2.5}$) have generally been considered to be more toxic and pose a much greater health risk for people than coarse-mode particles ($PM_{10-2.5}$), and most adverse health effects are considered to be induced by the fine fraction of PM_{10}. Particle size is also a surrogate for particle source and composition (Laden et al, 2000). On the other hand, some study also argued that neither fine nor coarse particles may be ruled out

(Lipfert et al, 2000). And while it is generally considered that $PM_{2.5}$ is a proxy of particles from anthropogenic combustion sources, in certain cases, natural sources can also be an important contributor to fine particles. For instance, Clainborn et al (2000) discovered that windblown dust contributes to a significant fraction of $PM_{2.5}$ concentrations during dust events in a semi-arid area like Spokane, Washington.

In recent years, studies are increasingly addressing the sources of particles as a factor that plays an important role in the health effects. A landmark study by Laden et al (2000) demonstrated that the combustion particles in the fine fraction ($PM_{2.5}$) from mobile and coal combustion sources are associated with increased mortality, but fine particles from crustal sources are not. And in comparing the health effects of fine particles from mobile and coal combustion sources, the former has a much stronger effect on increased mortality. The weakness of Laden's study was that it was not able to differentiate the time periods before or after the removal of lead from gasoline, given that the lead element in particles may be particularly toxic to people. Zeka et al (2005) found that when percentage of primary PM_{10} from traffic was added into the regression model, a higher relative risk was observed. The authors considered this as strong evidence that traffic particles are more toxic than those from other sources, and used it to explain the lower effect estimates reported from Eastern Europe, where coal combustion accounts for a much larger proportion of the particles than in Western Europe. Samet et al (2000) also use the sources of particles to explain some appreciable geographical variations in effect of PM_{10} on mortality in the U.S.: for the 90 cities, the largest effect was found in the Northeast where traffics are more concentrated than in other regions.

These studies provided evidence to support the importance of particle sources for health impacts, and in particular, the evidence that particulate matter from vehicle exhausts, which is the focus of the present study, is very likely to be more detrimental to health than PM from other sources. Given this evidence, it was argued that source-specific risk estimates need to be established for the most cost-effective abatement strategies (Forsberg et al, 2005). With respect to Bangkok, fine particulate matter in this area is characterized by diesel emissions, which contain mostly carbon dust and organic carbon compounds, as well as abundant ammonium sulfate salt formed from fuel sulfur and ammonia (Parsons, 2001). These particles are known to be toxic, leading to respiratory, neurological and carcinogenic health disorders (Parsons, 2001).

Currently the source-specific PM health risks are not adequately understood by scientific studies due to the difficulty in source apportionment. And previous epidemiological studies rarely separated the contributions of particles from different sources because in most cases, particles in a certain area originate from a mixture of sources. In order to understand the difference in mortality risk posed by particles from different sources, a stratified meta-analysis, referred to as a special type of meta-analysis that groups the estimates according to a particular feature or characteristic and then carry out separate meta-analyses within each group (Egger et al, 2001), is performed to compare PM_{10}-mortality coefficients associated with particles from different sources (details of the meta-analysis are described in Chapter 3).

2.1.6 Particulate matter health effects research in Bangkok

The critical PM air pollution problem in the city of Bangkok has been noticed world widely. In the late 1990s, the Pollution Control Department (PCD) of Thailand contracted Hagler Bailly Inc. to carry out a research project to estimate the effects of airborne PM on public health in Bangkok (Thanh & Lefevre, 2001). A final project report was submitted to the PCD in 1998 (Hagler Bailly, 1998), followed by a few publications. In the following section, existing studies on the health effects of PM pollution in Bangkok are surveyed.

Jinsart et al (2002) studied the roadside PM_{10} levels in six urban and suburban sites from December 1998 to March 1999, and found that almost all the PM_{10} levels in urban areas exceeded the Thailand National Ambient Air Quality Standards of 120 $\mu g/m^3$. Ostro et al (1999) and Vajanapoom et al (2002) examined the relationship between PM_{10} and daily mortality for the period of 1992-1995, and 1992 -1997, respectively, and both concluded that increasing PM_{10} levels were associated with the increase in daily mortality from all non-external causes, cardiovascular, respiratory and other diseases. The former study reported that a 10-$\mu g/m^3$ change in daily PM_{10} is associated with a 1-2% increase in all cause mortality, whereas the latter observed a weaker association (approximately 0.8% for the same level of change in daily PM_{10} level). The authors of the second study gave two reasons for the difference: data were gathered for a longer time period (1992-1997) and multi-site mean PM_{10} concentrations were used rather than using one monitoring site as in the study by Ostro et al (1999). Vichit-Vadakan et al (2001) conducted a study among three panels: school children, nurses (considered as a low exposed group) and non-smoking adults living and working within 2-km of an air quality monitoring station (considered as highly exposed adults). Through tracking the respiratory symptoms of all participants for 3 months in the winter season from late 1995 through early 1996, their study observed that an increase of 45-$\mu g/m^3$ in PM_{10} was associated with about a 50% increase in lower respiratory symptoms in highly exposed adults, about 30% in the children, and about 15% in nurse. The effect of lower respiratory symptoms found in this study seems to be much larger than the effect of incremental mortality, which usually falls into the range of 0.5-1% per $\mu g/m^3$ PM_{10}. Tamura et al (2003) designed a cross-sectional study to compare the prevalence of nonspecific respiratory disease (NSRD) among 1603 traffic policemen in Bangkok's heavily polluted, moderately polluted and suburban (low polluted) areas. They found that the prevalence of NSRD in the three areas was 13.0%, 10.9% and 9.4%, respectively, and the odds ratio for NSRD for each 10-$\mu g/m^3$ increase in PM_{10} was 1.11. This is supporting evidence that the increased prevalence of respiratory symptoms among traffic policemen was associated with urban traffic air pollution. Hagler Bailly (1998) examined the association between PM_{10} and hospital admissions and reported that for the elderly (age$\geq$65) the relative risk for 30-$\mu g/m^3$ PM_{10} was 1.05 and 1.01 for respiratory and cardiovascular hospital admissions, respectively.

In summary, the association between PM and adverse health effects has been repeatedly observed in Bangkok by different epidemiological studies, regardless of study designs. The consistency across different studies supports a cause-and-effect relationship between exposure to ambient particulates and health damage in the Bangkok area.

2.2 Health benefit analysis: A risk-based framework for policy formulation

A sophisticated tool, sometimes called Health Benefit Analysis (HBA) or Health Impact Assessment (HIA), is increasingly used to quantify the current cost of pollution (or potential benefits of control) (Ostro et al, 2006). HBA applies risk assessment techniques to identify the expected type and magnitude of health effects that have been linked with changes in emissions from air pollution sources, and ultimately estimates monetary values associated with these changes (Ostro et al, 2006). It is considered as an integrated assessment procedure that combines the knowledge in air quality modeling science, engineering, epidemiology and economics (Wang & Mauzerall, 2006). The results of a HBA are usually used as a component of cost-benefit analysis for a given policy, which involves estimation of the incremental costs and benefits associated with that policy (Levy et al, 2006). A general framework of HBA is summarized by Levy and Spengler (2002) as follows:

- Estimation/forecasting of the amount of goods produced;
- Evaluation of the required inputs to produce these goods;
- Emissions inventory/estimation;
- Dispersion modeling/exposure assessment;
- Estimation of impacts associated with pollutant exposures among at-risk subpopulations or systems; and
- Valuation and aggregation of impacts.

Levy and Spengler (2002) suggest that each of these components needs to be modeled with some precision and with adequate characterization of uncertainty. In recent years, increasing studies from all over the world have used this framework to analyze the health impacts associated with emissions from different sources. In the U.S., this framework has been utilized by the U.S. EPA in making regulatory decisions based on cost-benefit comparisons (e.g. The costs and benefits of reducing lead in gasoline, Schwartz et al, 1985; The costs and benefits of the Clean Air Act, EPA , 1999). And all of the National Ambient Air Quality Standards (NAAQS) in the U.S. include a benefits analysis (EPA, 1999a). Other examples of studies in the U.S. include an earlier study on assessing the health benefits of reducing PM air pollution in the whole U.S. (Ostro & Chestnut, 1998); Four recent studies on the health benefits of reducing ground-level ozone concentrations (Sanhueza et al, 2003; Levy et al, 2001; Ostro et al, 2006; Hubbell et al, 2005); A study on the health benefits of power plant emission control in Massachusetts (Levy & Spengler, 2002); And an evaluation of the health risk reductions associated with proposed PM standards in Philadelphia and Los Angeles (Deck et al, 2001). Examples of studies in Europe include Kunzli et al (2000), who assessed the public health impacts of traffic-related air pollution in three European countries; Rabl and Spadaro (2000) who examined the health effects of reducing energy consumption and shifting to cleaner energy sources in Europe. Some studies in Asia are Wang and Mauzerall (2006) who evaluated the impacts of adopting energy technologies on public health in eastern China; Li et al (2004) who investigated the human health benefits of establishing a new technology strategy for coal-fired power generation and a new industrial coal-use policy in Shanghai, China. Joseph et al (2003) estimated the health impact due to PM_{10} in an industrial, residential and commercial area of Mumbai, India. In addition, an example of studies in other regions is Cifuentes et al (2001)'s study on health benefits of urban air pollution reductions associated with climate change mitigation in three large cities in Latin America (Santiago, Sao Paulo, and Mexico City), as well as New York. The only studies found in Thailand using a similar framework is Thanh and Lefevre's (2001) study on the health benefits of controlling air pollution from a lignite-fired power plant. With respect to PM air pollution, an interesting finding is that the beneficial effects of reduced particulate levels appear to be far greater in rapidly developing nations than in developed nations, although they are substantial in both regions (Working Group on Public Health and Fossil-Fuel Combustion, 1997).

The existing studies either focus on the health benefits from achieving a target air quality standard (e.g. Ostro et al, 2006; Deck et al, 2001), or the benefits associated with specific policy measures targeting emissions from certain source(s) such as power plants and transportation sectors (e.g. Wang & Marzerall, 2006; Sanhueza et al, 2003), or health related environmental costs attributable to an energy use system as a whole such as transportation (e.g. Kunzli et al). This study falls into the second type since it examines hypothetical control scenarios targeting PM emissions from the transportation sector. Compared with studies focusing on health benefits from industrial emission mitigating policies, fewer studies have been performed to address control strategies targeting emissions from transportation. More importantly, this study mainly contributes to the health benefits analysis literature in two ways. First, although concerns have been aroused that with uncertainty about their actual effects, control policies may have much lower potential to reduce emissions, resulting in lower net benefits (Harrington et al, 1998), earlier health benefit analyses typically assumed that pollution control policies would achieve the expected emission reductions without taking into account this uncertainty issue. This study examines how uncertainty about the actual effects of policy imposition may affect expected health benefits, and further links the health benefit analysis with policy design considerations; Second, this study considers source-specific risk estimates for PM, whereas previous studies generally assumed every strategy to lessen PM levels having the same impact per unit change in the mass concentrations (Forsberg et al, 2005).

2.3 Policy measures to control air pollution from mobile sources

2.3.1 Control policies targeting emissions from transportation

As discussed before, motor vehicles are responsible for the largest sources of PM in the Bangkok area. Currently there are several key problems related to motor vehicle emissions in Bangkok, including:

- The number of vehicles traveling, as well as the demands for traveling are increasing rapidly, whereas the development of transportation infrastructure is too slow to accommodate the growth in vehicle population and increase in the number of trips. Traffic congestion has been a notorious problem in the Bangkok Metropolitan Region. Slow moving or idling vehicles generate much higher emissions during traffic jams.
- A large number of high emitters such as heavy-duty diesel trucks, old city buses, and two-stroke motorcycles are running on roads, which are responsible for a substantial proportion of the total motor vehicle emissions.

- Vehicle emissions have resulted in very high levels of PM on the areas close to traffic (so-called hot spots), which are dangerous for public health, in particular for those who work outdoors and those who work or live near roads but without air conditioning.

Given the evidence, earlier studies consistently recommended that in Bangkok policies should be directed at mobile sources to tackle the vital PM air pollution problem. Compared with industrial emissions control, the transportation field presents a range of environmental problems with considerable technical challenges: It presents more sociopolitical difficulties because monitoring and enforcement are difficult and because the owners and polluters are not professionals, which introduces several constraints (Sterner, 2003). The emission taxes or tradable permits that work well for stationary sources are not considered feasible for controlling mobile source pollution because technology is not available to monitor emissions at the source, i.e. to measure the emissions of each vehicle while traveling (Fullerton & West, 2002). At the same time, monitoring of individual vehicles is not only constrained by technologies and costs, but also resisted by vehicle owners (Fullerton & West, 2002). In addition, there are legal restrictions set against the search of a private vehicle. On the other hand, consumers' choices regarding gasoline and vehicle characteristics are heterogeneous, which may further increase the complexity of regulating motor vehicle emissions (Fullerton & West, 2002).

A World Bank publication (World Bank, 2001) categorizes measures to control emissions from motor vehicles into three types: (1) Fewer vehicle miles traveled in total; (2) Less fuel use per vehicle miles traveled (VMT); and (3) Less pollution per unit of fuel used. Based on these objectives, a variety of policy tools have been developed targeting traffic-related emissions. Each of the measures, more or less, seeks to achieve one or more of these objectives. The following table cited from the World Bank publication (World Bank, 2001) provides a full list of policy instruments targeting traffic-related emissions.

Table 2.8 Policy Instruments Targeting Vehicular Air Pollution

Policy Goals	Technological	Administrative	Economic
Reducing Vehicle Kilometers			
Increase private vehicle occupancy		High occupancy vehicle lanes; Parking priority to high occupancy vehicles; Encouraged car sharing.	Congestion pricing; Tax incentives.
Restrain demand		Vehicle use limitation; Parking policies.	Road pricing; Fuel tax; Parking pricing; Taxing vehicles by distance run.
Increase public transport share	Dedicated bus ways.	Bus priorities; Public transport regulatory reform.	Subsidy to public transport.
Reducing Fuel Used Per Vehicle Kilometer			
Improve fuel economy	Increase engine efficiency; Reduce vehicle size.	Fuel economy standards.	Fuel taxation.
Encourage nonmotorized transport	Investment in nonmotorized transport infrastructure.	Protection of nonmotorized transport in road use.	
Improve traffic management	Intelligent traffic system technology.		
Reducing Emissions Per Unit of Fuel Used			
Improve fuel quality		Tighter diesel fuel standards; Bans on leaded gasoline.	Differential taxation.
Improve vehicle maintenance		Age restriction on vehicles; Inspection/Maintenance programs.	Differential vehicle taxation and fines.
Improve conventional diesel technology	Four stroke; Electronic fuel injection; Oxidation catalyst (with diesel containing 0.05% sulfur or less).	Tighter emission standards for in-use vehicles; Diesel sulfur reduction to enable catalyst adoption.	
Improve two- and three-wheeler technology	Higher quality lubricant for two-stroke engine; Four-stroke engine; Pre-mix fuels for two-strokes.	Tighter 2T lubricant standards; Ban new two stroke-engines; Tighter two/three-wheeler emission standards.	Differential taxation.
Use alternative fuels	Investment in compressed natural gas distribution.	Much tighter PM standards; Mandate use of gas.	Differential fuel and vehicle taxation.
Switch to "clean diesel" technology	Ultra-low sulfur fuel (0.005% or less); Particulate trap.	Much tighter PM standards.	Higher tax on conventional diesel.

Source: World Bank, 2001

Similar to policy tools for controlling stationary emission sources, generally there are two distinct types of policy instruments targeting motor vehicle emissions: direct regulations and economic incentive (EI) instruments. Both types of policy tools have advantages and disadvantages. The key difference between them is the amount of discretion granted to pollution sources in determining their pollutant discharges (Harrington et al, 2004)[9]. In a regulatory policy the discretion belongs mainly to the regulator because for each regulated source the regulation normally specifies the limits for each pollutant or the technology used to achieve those limits, whereas an economic incentive policy provides incentives to abate but does not specify the abatement methods or even the quantity that is permissible to discharge because the discretion belongs to the regulated source (Harrington et al, 2004).

Although economic incentive instruments are increasingly used to control emissions in the industrial field and have achieved a prominent place among the tools used by government particularly in the regulation of industrial emissions (EPA, 2001), to date the most common policies targeting road transportation emissions are still regulatory (Sterner, 2003), mainly because of the difficulties and great uncertainties in implementing EI instruments due to the uniqueness of mobile emission sources. Moreover, it is believed that considerable health and environmental benefits can be gained through simple regulatory measures, particularly if uncertainty is relatively small (Sterner, 2003). Nevertheless, a switch of policy focus from regulatory to EI instruments in regulating vehicle emissions has been proposed (Harrington et al, 1994), considering the merits of EIs over direct regulations in terms of potential lower total control costs. In addition, it was also argued that the difference between economic incentive policies and more rigid regulatory policies would depend on how well the incentive policies are targeted at

[9]In the case of vehicle emissions control, the sources of pollution are vehicles, and motorists can be considered as polluters.

reducing emissions (rather than other goals such as raising revenues) and on their ability to influence behaviors (Harrington, et al, 1998).

Major regulatory instruments for reducing motor vehicle emissions include vehicle emission standards, fuel quality regulation (e.g. unleaded gasoline and limits on diesel sulfur), mandatory vehicle inspection and maintenance, etc; and major economic incentive instruments include environmental road pricing, taxes on fuels or car emissions, marketable permit systems for vehicle manufacturers, etc. Given the considerable uncertainty issues associated with instituting EI instruments and current lacking of international experience and best practices, direct regulatory measures are considered more practical in Thailand, and are the focus of this study.

In the Bangkok Metropolitan Region, primary programs and activities currently undertaken targeting vehicle PM emissions include (World Bank, 2002):

- An inspection and maintenance system for all diesel vehicles, taxis, tuk-tuks, light-duty gasoline vehicles and motorcycles;
- Gradually phasing out two-stroke motorcycles by enforcing higher taxes;
- Converting all taxis to run on clean-burning liquefied petroleum gas (LPG);
- Imposing stricter emission standards for new vehicles;
- Black smoke public complaint hotlines;
- Paving, sweeping and washing major roads;
- Prioritizing bus lanes;
- Developing convenient public transportation infrastructure (e.g. sky-trains and subways).

Currently the major control measures are direct regulations, and the only economic incentive policy tool adopted is taxes, but more are in the government’s policy agenda. For example, it has been proposed to designate high pollution areas in which fees are collected for vehicles to enter (United Nations Environment Program, 2004). Cost-Benefit Analysis is an essential method used by the Thai government to compare and prioritize various air pollution control measures[10].

2.3.2 Defining policy scenarios in this study

This study considers the business-as-usual (BAU) scenario, or the scenario with no further control policies relative to the baseline year 2000, and two control policy scenarios: (1) Enhanced PM-related vehicle inspection and maintenance (I/M) programs targeting all diesel-fueled vehicles and motorcycles. The cut-points of vehicle inspections are set to be 70-75% of the current average emission rates for each type of vehicles, aiming at reducing total mobile source PM_{10} emissions by approximately 25%; (2) Programs of retrofitting or repowering equipment including installing diesel oxidation catalytic converters or particulate traps (these two exhaust after treatment systems are exclusive to each other), and switching to clean fuels such as compressed natural gas (CNG) or liquefied petroleum gas (LPG) for certain types of diesel vehicles. The programs of retrofitting or repowering equipment are proposed to be introduced in addition to the I/M programs in order to further reduce PM_{10} emissions through technology innovation whereas assuring emission control equipment is functioning properly through the I/M. The PM-oriented I/M programs are the main focus of this study.

2.3.2.1 Vehicle inspection and maintenance program

Motor vehicle I/M programs have been considered to be cost-effective in the U.S. and are required by the Clean Air Act Amendments of 1990 in regions with the most challenging air pollution problems (Eisinger, 2005). The Transportation Research Board analyzed 20 mobile source control measures and found that I/M was the most

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[10]Conversation with Ms. Mingquan Wichayarangsaridh, Director, Air Quality and Noise Management Bureau, Pollution Control Department of Thailand, July 2005.

cost-effective, averaging $2000/t of pollution lessened, a value considered good in the air quality management field (Transportation Research Board, 2002). The main purpose of I/M programs is to encourage better maintenance for in-use vehicles and to assure the vehicle emission control systems are functioning properly through periodic inspection. The rationale for an I/M program is that the emission distribution among the vehicle population is highly skewed: a small portion of vehicles (estimated at 5-10%), sometimes called gross polluter vehicles, is responsible for a substantial fraction (variously estimated at 50 to 80%) of total vehicle emissions (Harrington, 1997; Beaton et al, 1995). Moreover, not only old vehicles can be gross polluters, but also vehicles of all model years may include some proportion of gross polluters (Beaton et al, 1995), due to the factor that vehicle emission levels are heavily dependent on maintenance. This problem can be even more highlighted in developing countries, where vehicles have a long lifetime and are often poorly maintained (Sterner, 2003). In this case, upgrading maintenance practices and replacing the worst engines should be considered first before moving on to better technology (World Bank, 2001). In addition, despite technological and regulatory advances, new vehicle standards are not sufficient to achieve pollution abatement goals if vehicles deteriorate rapidly, resulting in increasing emission rates (Sterner, 2003). Therefore, to control rapidly growing vehicle emissions, governments must not only affect the behavior of vehicle manufacturers and fuel suppliers, but also the actions of drivers in terms of how well they maintain their vehicles regardless of their vehicle ages (PA Government Services, Inc, 2004). However, although simple in concept, the detailed design and implementation of I/M programs is challenging. For example, when emission control equipment malfunctions, vehicle performance may be unaffected, hence the driver has no private incentive to seek repairs, and demanding private expenditures of money and time by vehicle owners will create the usual tensions that lead many actors to try to evade the regulation in numerous ways (PA Government Services, Inc. 2004). The practices of I/M programs in the U.S. have shown various barriers that may cause failure of these programs to generate the emission reduction originally anticipated by policy makers. For instance, motorists may have many opportunities to evade required repairs such as testing vehicles numerous times until they happen to pass. By and large, launching an effective I/M program requires massive behavior change among the drivers of a region (PA Government Services, Inc. 2004)

Despite the remaining controversies, the I/M programs in the U.S., in particular the enhanced I/M established by the U.S. EPA in 1992, have played an important role in assisting states to achieve air quality standards. For example, in the state of California, the enhanced I/M programs were responsible for about 25% of the emission reductions required by 1999 under the State's 1994 Ozone State Implementation Plan (California Air Resource Board, 2000). In developing countries, the I/M programs in Mexico City, developed with the assistance of the World Bank, are regarded as one of the most successful cases. The case of Mexico City demonstrates that substantial and sustained mobile source emissions reductions and air quality improvements are possible through effective I/M programs that are characterized by committed leadership, the right institutional design and the right incentives (PA Government Services, 2004). At present, I/M programs are considered to be a core air pollution control measure in severely polluted urban areas (Eisinger, 2005). Moreover, traditional I/M programs focus primarily on gasoline-fueled vehicles. However, given the increasing public awareness of the health and environmental threats posed by emissions from diesel-fueled vehicles, there is a growing interest to introduce equivalent programs and test protocols also for diesel-fueled vehicles in order to reduce emissions, particularly fine particulate matter emitted from these vehicles[11]. Testing equipment for PM emissions is less mature and more expensive compared with those for CO, HC and NO_x (main air pollutants in gasoline exhaust). Currently, the most commonly used procedure is opacity testing, but its accuracy in terms of measuring levels of PM emissions is facing a lot of challenges. Nevertheless, the ongoing development in test procedures and technologies (e.g. new PM meters capable of measuring PM vehicle emissions) makes PM-oriented I/M programs more attractive for policy makers to consider.

In Bangkok, the existing I/M system was initiated in 1994, and applies to all six provinces in the Bangkok Metropolitan Region (defined as the city of Bangkok and its surrounding five provinces, details will be provided later), which is a hybrid vehicle inspection system comprising government-operated centralized inspection stations and privately operated inspection stations (Parsons, 2001). Parsons' report states that Bangkok's existing I/M programs are ineffective at detecting vehicles that do not meet the emission standards due to various reasons, including common use of dysfunctional test equipment, non-consistent administration system, lack of skilled test workers, non-automated facilities, poorly managed data records, and so forth, resulting in continued excessive emissions. Another study by Pala-En (1998) also found that 17 of 30 (56.7%) selected private motorcycle inspection centers in Bangkok did not follow the standard testing procedure of the Department of Land Transport to issue a certificate of compliance and more than 60% of these inspection centers had technician practices scores in low levels, which were in the range of 4.50-13.25 out of a total of 25 points.

Upgrading the existing I/M programs is needed to improve the effectiveness of the programs. Furthermore, the main reason that makes an effective diesel-related I/M program desirable in Bangkok is, as discussed before, that the principal sources of PM

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[11]Source: Clean Air Initiative, Air quality information systems, http://www.cleanairnet.org/infopool/1411/propertyvalue-17722.html, accessed November 1, 2007.

pollution in this area are diesel-fueled vehicles (including most city buses, city trucks, light duty trucks, vans, etc), in particular those old ones lacking emissions control, which can contribute more than 50% of the total PM_{10} emissions. Given that pollution levels from diesel vehicles are heavily dependent on maintenance, and a well-maintained diesel vehicle will generally retain a good emissions performance throughout its operating life, it is expected that in Bangkok considerable reductions in PM emissions from mobile source will be delivered through effective I/M programs targeting related vehicles.

Parsons' final report on the Bangkok Air Quality Management Project recommends an effective I/M system targeting all diesel-fueled vehicles and motorcycles to be the most cost-effective control measures to mitigate PM pollution in the BMR. Overall, they estimate that I/M programs will cut current mobile source emissions by up to 25%. However, a key gap in Parson's report is that in estimating the emission reduction benefits associated with the proposed I/M programs, they did not take into account the uncertainty about the actual effects of the programs in reducing PM emissions. Instead, they only considered the ideal case that all violating vehicles would be identified and properly repaired. Based on the past experience in the U.S., the actual effectiveness of I/M programs in reducing PM emissions can vary significantly, or be much lower than the levels anticipated. For instance, five studies have estimated the reductions in light-duty motor vehicle hydrocarbon exhaust emissions from California enhanced I/M vary in the range of 12-34% (summarized by Eisinger, 2005). In theory, vehicular emission reductions available from I/M programs are mainly determined by the failure thresholds, or cut points, used to identify problem vehicles, but are also quite sensitive to a variety of factors, such as the actual percentage of problem vehicles identified, the percentage of problem vehicles waived from repairs or operating illegally, the emission reduction achieved by repairs, the durability of repairs, and so on (Eisinger, 2005).

Given the considerable uncertainty involved in the total emissions reductions achieved by I/M programs, this study takes into account this issue and examines its effect on the total benefits (health benefits are considered here) of the programs. Furthermore, in order to understand how I/M program design considerations, i.e. the change in factors that affect the total reductions in vehicle emissions achieved by an I/M program, would alter the total health benefits of the program, an I/M program-evaluating tool developed by Eisinger (2005) will be employed and linked to health benefit analysis[12]. This tool incorporates a complete list of factors that affect the effectiveness of an I/M program and allow the users to predict the total percentages of vehicle emission reductions achieved by the program. The theoretical basis and modeling details of this tool is discussed in Chapter 3.

2.3.2.2 Programs of retrofitting or repowering equipments

Besides I/M programs to assure emission control equipment is functioning properly, further emission reductions can be delivered through retrofitting or repowering in-service vehicles, in particular heavy-duty vehicles, with best available control technologies or cleaner fuels. The second policy scenario considered in this study is a policy package that combines I/M measure with the installation of feasible PM reduction technologies in primary diesel-fueled vehicles including buses and trucks and switching to cleaner fuels.

Diesel engines have a higher mass of particles in the exhaust mainly due to less clean fuels and no catalytic treatment of the exhaust gases (Forsberg et al, 2005). Two common exhaust after-treatment technologies focusing on diesel PM emissions are

[12]This spreadsheet-based modeling tool, called I/M Design Tool (version 5.5) has been provided by the author, Dr. Douglas S. Eisinger of Sonoma Technology, Inc, with the permission to use for research and policy analysis purposes.

oxidation catalytic converters and particulate traps (Parsons, 2001). Using these devices as retrofit devices to reduce emission levels from in-use vehicles is rapidly gaining acceptance (Parsons, 2001). For example, in 1997, the U.S. EPA approved a certification system for diesel oxidation catalytic converter. The retrofit kit used under this program demonstrated a 25% reduction in PM for 1979 to 1993 Detroit Diesel Corporation (DDC) 2-stroke engines (EPA, 1998). And in 2000, the Government of California announced their plan to diesel particulate traps, which can reduce diesel PM emissions by at least 85 percent, to retrofit existing diesel vehicles (California Air Resources Board, 2000).

Improving fuel quality is another key component of reducing motor vehicle emissions (Parsons, 2001). Reducing sulfur levels in diesel fuel will lower particulate emissions in exhaust, and also has an important ancillary benefit of enabling the use of highly effective exhaust after-treatment such as continuously regenerating particulate traps (Parsons, 2001). Gaseous fuels including compressed natural gas (CNG) and liquefied petroleum gas (LPG) typically generate particulate emission rates at least ten times lower than diesel, but converting an existing diesel engine to operate on these fuels requires some major internal modifications and consequently high costs (Parsons, 2001).

However, at present there are several barriers that make the adoption of some control measures not feasible or highly cost-ineffective in the BMR. First, particulate traps is not currently considered feasible to be adopted widely due to the requirement of fuel with sulfur content around ten times lower than the present levels, although it may be feasible for the operators of large, well-controlled fleets to procure suitable fuel and use it for their own vehicles; second, converting a diesel engine to operate solely on a gaseous fuel (CNG or LPG) would probably suffer from the high costs (a minimum of around $20000 per vehicle). In addition, CNG is not currently supplied by commercial infrastructure in Bangkok, and LPG has an obvious advantage over CNG in terms of the size and weight of storage tanks (a CNG tank is around three times larger than a LPG tank); third, although light-duty diesel vehicles are the highest polluting group among all diesel vehicles, their low annual kilometers traveled cause that the capital cost of retrofitted equipment is amortized very slowly (Parsons, 2001). Given these factors, Parsons' technical report recommended that the most feasible as well as cost-effective package of policy measures is to combine together effective I/M programs, installing oxidation catalyst converters on heavy-duty diesel vehicles (heavy trucks and buses), and installing LPG and diesel bi-fuel fueling systems (typically in the ratio of 80% diesel to 20% gaseous fuel) on vehicles equipped with oxidation catalytic converters. The total annual health benefits of this policy package are analyzed in Chapter 4.

Furthermore, although some highly effective measures (e.g. particulate traps, CNG or LPG fueled engine) is currently not feasible in the BMR, rapidly increasing demand for motor vehicles will make the investments on fuels and technologies with strong environmental credentials ultimately imperative to mitigate air pollution from mobile sources. However, due to the high costs of adopting these control measures, governments in developing countries may be inclined to invest in these measures. This study aims at demonstrating the potential health benefits associated with some control measures that can potentially reduce vehicle emissions to a very large extent in order to help government to make decisions about the investments on these measures in the future. Therefore, health benefits of these measures are also estimated in addition to the policy package mentioned above.

2.3.3 Base year and baseline emissions

The year 2000 is selected as the base year to establish the emission baseline because the nature of the available data needed for the modeling purpose of this study.

More importantly, in the prior year 1999, which was named by Bangkok Metropolitan Administration (BMA) as the Air Pollution Mitigation Year (United Nations Environment Program, 2002), thirteen pollution mitigation measures were implemented in the BMR, including:

1. Provide car engine tune-up service stations to the public for free;
2. Publish car engine maintenance manuals for distribution to the public;
3. Set up black smoke inspection points in 50 districts jointly between traffic police and BMA;
4. Set up mobile black smoke inspection units comprising 6 cars operating in 6 areas;
5. Set up motorcycle's white smoke and noise level inspection units within the inner area of Bangkok;
6. Air quality report of the critical area with cooperation from PCD through display boards and air quality report for the pollution-free streets;
7. Designate pollution-free streets that prohibit single occupant (originally there were three streets and expanded to eight streets);
8. Pave street shoulders to reduce road dust;
9. Enforce wind screen for buildings under construction;
10. Enforce dust control for trucks to use cover and wheel washing;
11. Put up campaign boards to inform public on various measures being done;
12. Designate car-free streets for selected suitable streets to reduce air pollution;
13. Improve fuel quality by joint efforts to reduce air pollution.

These measures did significantly contribute to curbing Bangkok's severe PM pollution in the last decade, whereas ambient PM concentrations continue to exceed the air quality standards. Given this, the goal of this study is, on the basis of these control measures, to examine the possibility of further emission reductions relative to the 2000 levels through introducing new abatement policy measures.

Chapter 3 Methodology

The methodological steps of this research are shown in the following graph, and will be discussed in detail in this Chapter.

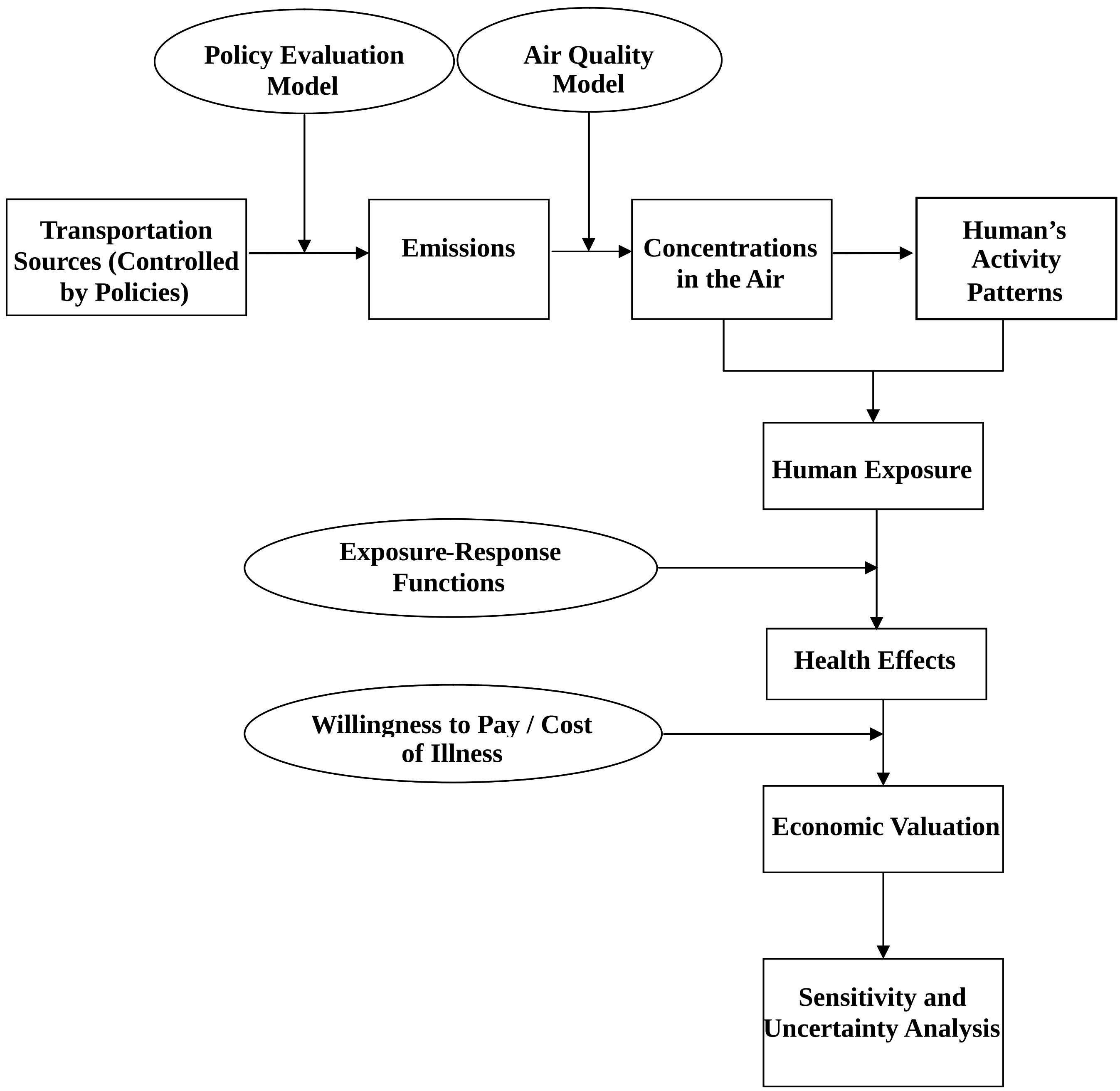

Figure 3.1 Methodological Steps

3.1 Modeling human exposure to ambient particulate matter

3.1.1 Simulating ambient PM_{10} concentrations

The estimation of human exposure to outdoor air pollution often involves two key elements: assessing changes in ambient concentrations of pollutants under the policy scenarios considered, and estimating the population exposed to these changes of pollutant levels (Ostro et al, 2006). To simulate the transport, dispersion, and related processes of a pollutant in the atmosphere and estimate the air pollution concentrations at interested sites, usually referred to as receptors, scientists have developed air quality models. These models are valuable tools for supporting policy and regulatory decisions. Spengler and Wilson (1996) summarized the models used in air pollution studies as usually being variations of three types:

- The simple proportional model: Based on the idea that the amount of a reduction in ambient concentrations is proportional to the emission reduction. This type of modeling was applied extensively in the 1960s and early 1970s in the U.S. as local and state air pollution control agencies devised fuel sulfur regulations and transportation control plans.
- Numerical simulation models: Use derivations of the Navier-Stokes equation for fluid flow. Concentrations are estimated by a stepwise finite differentiation scheme. This type of model has been extensively used in photochemical modeling, where the intermediate concentrations of a number of reactive pollutants are needed.
- Gaussian plume or Lagrangian models: Most widely used models that are based upon the idea that the eddies or turbulence in the air act on the emitted pollution in such a way that the concentrations will be distributed normally around a peak centerline concentration. This type of models is used for primary pollutants such as PM, NO_2, SO_2, and air toxics.

In practice, the U.S. EPA categorizes the most commonly used air quality models into three types[13]:

- Dispersion Modeling: These models are typically used in the permitting process to estimate the concentration of pollutants at specified ground-level receptors surrounding an emissions source;
- Photochemical Modeling: These models are typically used to simulate the impacts from all sources by estimating pollutant concentrations and deposition of both inert and chemically reactive pollutants over large spatial scales; and
- Receptor Modeling: These models are mathematical or statistical procedures for identifying and quantifying the sources of air pollutants at a receptor location.

The most appropriate air quality model for the present study may be a dispersion model, since it primarily aims at understanding the human health impacts attributable to air pollution in a metropolitan area. And unlike particulates from industrial point sources or natural sources, which may be transported over very long distances with high stacks or climate conditions, the particulate emissions from mobile sources considered in this study are expected to have much greater impacts at the local scale than at the regional scale[14]. Particulates from vehicle exhaust have been regarded as a significant contributor to local PM concentrations (Forsberg et al, 2005). On the other hand, ignoring the impacts at regional scale may underestimate the total benefits from pollution control. The modeling domain of this study is the Bangkok Metropolitan Region (BMR, also known as Bangkok Metropolitan Area), which includes Bangkok and its surrounding five provinces, Samut

[13]Source: U.S. EPA website, Air Quality Models, http://www.epa.gov/scram001/aqmindex.htm, accessed October 30, 2006.

[14]Studies on air quality have classified the impacts into local and regional scales due to the difference in the way pollutant plumes disperse in the atmosphere at these scales: at the local scale, chemical reactions in the atmosphere have little influence on the concentrations of pollutants whereas at the regional scale, chemical conversion and dry and wet deposition processes need to be taken into account to determine pollutant concentrations (Thanh & Lefevre, 2001).

Prakarn, Nonthaburi, Pathum Thani, Nakhon Pathom, and Samut Sakhon, altogether covering an area of 7761.50 square kilometers (the area of Bangkok is 1568.20 km^2), or about 1.5% of the area of Thailand, and with a population of 9.9 million (as of November 2006), or about 15% of the total population of Thailand, with population density of 1279 per square kilometer[15]. The reason that Bangkok's peripheral provinces are also included is that these areas are closely linked in terms of traffic and economic development (Oanh & Zhang, 2004). Bangkok is a tropical metropolitan area characterized by a fairly warm and humid climate. The Meteorology Department of Bangkok defines seasons in this city as: Winter – October 16 to February 15; Summer – February 16 to May 15; and Rainy – May 16 to October 15 (Ostro et al. 1999). The temperatures remain high over the entire year, for instance, the temperature in Bangkok during the period 1992 through 1997 varied from about 25ºC (approximately 77F) to 32ºC (approximately 90F), with the mean 28.8ºC (approximately 83.9F) (Vajanapoon et al, 2002).

U.S.EPA's preferred dispersion models include AERMOD, a steady-state plume model, and CALPUFF, a non-steady-state puff dispersion model. These models require source location and parameter data, receptor locations, and meteorological data as input, and generate ambient pollutant concentrations as output. However, for the present study, it would be rather difficult to run such a dispersion model to obtain a reasonable estimate of PM concentrations in the study domain, because the PM emission rates for different automotive categories are different (e.g. fleet-average emission rate of city buses/trucks can be more than four times higher than light-duty trucks), and the distribution of vehicle categories in the study area are not identical or consistent, plus adequate data for the distribution of vehicle types are not available at present. This study, therefore, assesses the projected change in ambient PM_{10} concentrations at monitoring sites associated with

[15]Source: Wikipedia, http://en.wikipedia.org/wiki/Bangkok_Metropolitan_Area, accessed November 15, 2006.

the imposition of control strategies based on the idea of the simple proportional model. The approach used is discussed below.

It is assumed that the shares of different sources for the ambient concentrations of PM_{10} are proportional to their relative contribution to the emission inventory of PM_{10}. For example, if PM_{10} emitted from transportation account for 50% of the total PM_{10} emissions from all sources, then 50% of ambient PM_{10} concentrations (daily 24-hour average PM_{10} concentrations are considered in this study) are contributed by transportation sources. This assumption is expressed in formula as follows:

$$F_T^C = \frac{C_T}{C_T + C_A} = F_T^E = \frac{E_T}{E_T + E_A} \qquad (3\text{-}1)$$

Where,

F_T^C : Fraction of ambient PM_{10} concentration contributed by transportation sources

F_T^E : Fraction of total PM_{10} emissions from transportation sources

E_T : PM_{10} emissions from transportation sources

E_A : PM_{10} emissions from all other sources

C_T : Ambient PM_{10} concentration contributed by transportation sources

C_A : Ambient PM_{10} concentration contributed by all other sources

The simple proportional model states that the amount of a reduction in ambient concentrations is proportional to the emission reduction. Based on this, the reduction in ambient PM_{10} concentrations contributed by transportation sources (ΔC_T) is proportional to the projected emission reduction in this source (ΔE_T). That is to say, in the case that the magnitude of transportation emission sources is reduced by a certain percentage (e.g. 25% under the I/M scenario), the ambient PM_{10} concentrations contributed by this type of sources will be reduced by 25% accordingly. In addition, it is assumed that except for mobile sources, other PM_{10} emission sources are unaffected by policies under consideration, i.e. E_A and C_A remain unchanged with policy. Therefore, the change in observed ambient concentrations (ΔC) equals the projected change in concentrations contributed by mobile sources, expressed in formula as follows:

$$\Delta C = \Delta C_T \tag{3-2}$$

This approach allows us to calculate the changes in daily PM_{10} concentrations observed at monitoring sites that reflect the imposition of hypothetical control policies by using Bangkok's PM_{10} emission inventory data and daily monitoring data for the year 2000. Emission inventory provides information on fractions of PM_{10} from various sources. Here the assumption is that that Bangkok's inventory contains all natural (e.g. forest burning in the study region Southeast Asia) and anthropogenic (e.g. transportation, industry, incinerators, etc) PM_{10} sources that contribute to the observed daily PM_{10} concentration (C) at monitoring stations. Hence, we have:

$$C = C_T + C_A \tag{3-3}$$

From equations (3-1) and (3-3) we derive:

$$C_T = F_T^C \times C = F_T^E \times C \tag{3-4}$$

And based on the idea of the simple proportional model we have:

$$\Delta C = \Delta C_T = C_T \times \frac{\Delta E_T}{E_T} \tag{3-5}$$

Where, ΔE_T is the change in PM_{10} emissions from transportation sources, and $\frac{\Delta E_T}{E_T}$ is the expression of the projected percentage change in PM_{10} emissions from transportation sources after policy.

Finally, from equations (3-4) and (3-5) we derive:

$$\Delta C = C_T \times \frac{\Delta E_T}{E_T} = F_T^E \times C \times \frac{\Delta E_T}{E_T} \qquad (3\text{-}6)$$

Therefore, by obtaining F_T^E from the PM_{10} emission inventory for the study area, and C, daily monitoring PM_{10} concentration, as well as the percentage change in PM_{10} emissions from motor vehicle sources under the policy scenarios considered, we will be able to estimate the change in ambient PM_{10} concentrations at the sites where monitoring data were obtained.

This approach is applied to compute the daily reductions in PM_{10} concentrations at all monitoring stations in Bangkok where daily PM_{10} monitoring data are available in the base year 2000. The current air quality monitoring network operated by the Pollution Control Department consists of seventeen automated stations located in the BMR (Parsons, 2001), but among them only eight have PM_{10} data in 2000. The names, IDs and locations of these stations are listed below.

Table 3.1 Eight Monitoring Stations in Bangkok with Daily Ambient PM_{10} Concentrations in 2000

Station Name	ID	Longitude*	Latitude*
Ramkhamhaeng University	09T	100.62	13.75
Klongjun National Housing Authority	10T	100.66	13.77
Huai Khwang National Housing Authority	11T	100.57	13.77
None-tree Vitaya School	12T	100.55	13.70
Singharatpitayakom School	15T	100.45	13.68
Thon Buri Substation Electricity	52T	100.52	13.80
Traffic Police Residence	53T	100.60	13.79
Dindang Housing Authority	54T	100.56	13.76

*Source of the geographic location information: Puangthongthub, 2006

There were also days when monitoring data were missing. Daily PM_{10} concentrations on these days are estimated based on data of the previous and next days using a linear interpolation approach. In the case that monitoring data were missing for ten or more continuous days ($n \geq 10$), the seasonal average PM_{10} level is utilized as an approximation of the daily concentrations[16].

The currently available PM_{10} emission inventory in Bangkok was examined. Usually the sources of particles include local and regional contribution to total PM_{10} mass concentrations (Forsberg et al, 2005)[17]. However, in the BMR there still exists significant uncertainty in the relative share of contribution to the total PM_{10} emissions by different sources. A recent World Bank study found that emission inventories prepared and reported by different organizations can differ by a factor of twenty, and the existing inventories in the BMR do not account for pollution generated outside its airshed (World Bank, 2002). The following table summarizes the results of motor vehicle emission rates and percentages of total PM_{10} emissions studied by different organizations.

Table 3.2 Motor Vehicle PM_{10} Emission Rates by Different Studies

Study Organization	Year	Emission Rate (ton/year)	Percentage of Total PM_{10} Emissions (%)
Pollution Control Department	1997	20,602	53.94
Pollution Control Department	2002	19,735	53.77
Radian International LLC	1997	14,043	22.8
Parsons International Limited	1999	14,595	N/A

Sources: Pollution Control Department (2000), Radian International LLC (1998), and Parsons International Limited (2001).

The table shows that the annual emission rates of PM (in units of tons per year) estimated by different studies differ by a factor of about 1.5. Regarding the percentage of

[16]Seasonal average rather than annual average concentrations are used here to reflect the possible significant difference among seasons throughout a year. The seasons in Bangkok include Winter, Summer and Rainy, as defined before.

[17]Regional contribution refers to long-range transported particles from large sources on the continent, including both primary particles and secondary particles formed from gaseous precursors, whereas local contribution mainly consists of primary particles.

total PM_{10} emissions from mobile sources, five source categories were considered in Radian's study: re-entrainment, industrial/commercial boilers, mobile sources, power plants and construction, whereas three categories were included in PCD's analysis: point sources, mobile sources and area sources (details of each were described in Chapter 2). It seems that PCD's study included more sources of PM except for re-entrainment. Radian's study estimated that re-entrainment accounted for 33.2% of total PM emissions, although PM from re-entrainment in later years was very likely lower than in 1997 due to the control strategies implemented (World Bank, 2002).

This study employs the annual emission rate (in units of tons per year) estimated by Parsons (2001) due to the factor that this is the only study that estimated the fleet-average PM_{10} emission rates (in units of grams per vehicle-kilometer) for different categories of vehicles in the BMR. However, Parsons' study does not provide an estimate of the contribution of mobile sources PM to the total PM emissions in the study area. Therefore, this information has to be estimated based on other available studies. Base on the information in Table 3.2, the total annual PM_{10} emissions in the BMR are derived as follows.

Table 3.3 Estimated Total Annual PM_{10} Emissions from All Human Sources in the BMR (Year: 2000)

Study Organization	Year of Study	Estimated Annual PM_{10} Emissions from Motor Vehicles (2000, ton/year)*	Percentage of Total PM_{10} Emissions (2000)	Total Annual PM_{10} Emissions in the BMR (2000, ton/year)
Pollution Control Department	1997, 2002	20,077	53.84%	37,290
Radian International LLC	1997	14,043	22.76%	61,700
			Mean Estimate	**49,495**

**Note*: The original studies provided estimates for the years of 1997 and 2002. The 2000 estimate was calculated by assuming a constant annual emission decrease rate for the period of 1997-2002.

To take into account the uncertainty in the emission inventory, this study uses the mean estimate of the two study (49,495 tons per year as shown in Table 3.3) to represent the total human source PM_{10} emissions in the study area. And since the annual emission rate provided by Parsons' study is employed, the percent contribution of mobile sources to total PM_{10} emissions is estimated as follows.

Table 3.4 Motor Vehicle PM_{10} Emission Rates and Percentages Used in This Study (Year: 2000)

Study Organization	Estimated Annual PM_{10} Emissions from Motor Vehicles (2000, ton/year)*	Total Annual PM_{10} Emissions in the BMR (2000, ton/year)	Percentage of Total PM_{10} Emissions (2000)
Parsons International Limited	15,650	49,495	**31.62%**

**Note*: The estimated annual emission from motor vehicles in 2000 is estimated based on the available 2000 vehicle emission factors, 1999 vehicle registration data, estimated vehicle annual growth rates and estimated average annual vehicle kilometer traveled. The details of calculation are provided in Chapter 4.

31.62% is considered as the mean estimate of the relative contribution of mobile sources to the total PM_{10} emissions in the BMR. In the uncertainty analysis, this parameter is assumed to be uniformly distributed, and the upper and lower bounds are assumed to be 1.5 times (47.43%) and half (15.81%) of the mean estimate, respectively.

Moreover, if the PM_{10} emission inventory used does not account for pollution generated outside its airshed, i.e. regional sources such as the forest burning particulates in Southeast Asia, an estimated regional background level of PM_{10} should be subtracted from the measured concentrations. To my knowledge, currently there are very few studies on the regional background levels of PM_{10} in Thailand. The study by Puangthongthub (2006) estimated the regional PM_{10} background in different regions of Thailand by using the 5th percentile of monitoring data collected in each corresponding region during 1998-2003. As a result, the regional background levels in different regions in Thailand were: 3.02 μg/m^3 in the BMR and the central region, 7.42 μg/m^3 in the Northern region, 12.33 μg/m^3 in the Southern region, and 7.65 μg/m^3 in the Northeastern region. Overall, these findings indicate that the regional background levels of PM_{10} in Thailand are generally small. The estimate for the BMR (3.02 μg/m^3) is employed here, assuming that over time, the regional background level of PM_{10} remains unchanged in the study domain. In the uncertainty analysis in Chapter 6, this estimate is considered to be the mean of the background PM_{10} level, the lower bound is 0, and triangular distribution is assumed due to the lack of further information on the PDF of this parameter. Further more, based on the mean and the lower bound values, the upper bound value is estimated to be 6.04 μg/m^3.

Another source of uncertainty is that in different parts of the study domain, the relative contribution from different sources to total PM emissions is expected to be different. For instance, a larger percentage of PM from mobile sources is likely to be found in inner than outer Bangkok. However, information on this is not available to reliably estimate the relative source contribution to PM exposure in different parts of the study area. In modeling the air quality, identical relative contributions from different sources across the study area are assumed. More specifically, the percentages of mobile sources listed in Table 3.3 apply to the entire study area.

3.1.2 Assessing human exposure to particulate matter

In most observational epidemiological studies of air pollution and health effects, the ambient pollutant concentrations obtained from fixed site air monitoring networks are typically used as the surrogate of the concentrations to which people are exposed in their daily lives (Watchalayann et al, 2005). Usually the individual average daily exposure to pollution depends upon people's activity patterns. For example, those who work or live close to traffic areas, such as shop house dwellers, street vendors, traffic police, and those who work or live in places without air-conditionings are exposed to much greater air pollution levels than those protected from exposure by air conditioning (Parsons, 2001). In general, the level of air conditioned housing in Thailand is still relatively low despite of its tropical climate conditions, and thus indoor and outdoor mingling of particulates is believed to be rather complete as particles penetrate most indoor environments readily especially when the windows and doors are frequently opened, which is the case in Thailand (Seaton et al, 1995). As for the transportation sector, it was estimated that about 60% of the trips are taken in non-air conditioned buses (Parsons, 2001). Moreover, Watchalayann et al's (2005) study on exposure to PM_{10} of central Bangkok's shop house dwellers found that the average of personal exposure lies in between indoor and outdoor PM_{10} concentrations, but personal exposures are significantly correlated with both indoor and outdoor concentrations. This is further supporting evidence for the argument that ambient PM_{10} concentrations could reliably be used as representative of personal exposure in the setting and living style of Bangkok residents. Given this, ambient PM_{10} concentrations are used in this study as a proxy for individual exposure.

In order to estimate the number of people affected by the changes in PM_{10} levels at a monitoring site, the total population in the BMR in the base year 2000 is equally assigned to each of the eight stations that have daily PM_{10} concentration data. Here it is assumed that the population in Bangkok is equally distributed around each station. Due to the lack of PM_{10} monitoring data in Bangkok's surrounding five provinces, daily PM_{10} concentrations averaged across the eight stations are used as estimates for the daily concentrations in these provinces.

3.2 Estimating the change in health effects attributable to particulate matter exposure

3.2.1 Health endpoints considered

As discussed in Chapter 2, a range of adverse health effects has been associated with exposure to ambient PM. Previous health benefit studies consistently suggested that death usually represents the dominant impact in economic terms, and therefore is of particular interest for policy-makers (Kunzli et al, 2001). Nevertheless, for a comprehensive evaluation of the health benefits associated with pollution mitigation, both mortality and morbidity effects are included in this study. Those health endpoints for concentration-response functions are not available in existing literature are excluded. Based on U.S. EPA's list of PM health effects and other relevant literature (e.g. Ostro & Chestnut, 1998; Thanh & Lefevre, 2001; Cifuentes et al, 2001), the following health endpoints are included in this study.

Table 3.5 Health Endpoints Considered in This Study

Health Endpoint	Pollution Indicator	Study Type
Acute adult mortality (AM)	PM_{10}	Time-series
Acute postneonatal infant mortality (AIM)	PM_{10}	Time-series
Chronic bronchitis (CB)	$PM_{2.5}$	Cohort
Respiratory hospital admissions (RHAs)	PM_{10}	Time-series
Cardiovascular hospital admissions (CHAs)	PM_{10}	Time-series
Emergency room visits (ERVs)	PM_{10}	Time-series
Acute asthma attacks (AAAs)	PM_{10}	Time-series
Acute respiratory symptom days (ARSDs) (1)	PM_{10}	Time-series
Respiratory related restricted activity days (RADs) (2)	PM_{10}	Time-series
Sensitivity analysis: Estimating mortality using cohort studies		
Chronic adult mortality (CM)	$PM_{2.5}$	Cohort
Chronic postneonatal infant mortality (CIM)	$PM_{2.5}$	Cohort

Note:

(1) Acute respiratory symptom days are days when a person has one or more common respiratory such as sinus congestion, cough, or wheeze. And epidemiology studies of these kinds of symptoms are typically based on self-reported daily incidence during an extended study period

(Hagler Bailly, 1998)

(2) Restricted activity days are defined as days where a respondent was forced to alter his normal activity and indicates the most generalized health outcome, and its measure includes days of work loss or bed disability as well as more minor restrictions (Ostro, 1987).

3.2.2 Concentration-response (CR) functions

As mentioned before, assessing CR functions is a key step in estimating heath benefits of air pollution control. In the following sections, the CR functions for each health endpoint used in this study are discussed.

3.2.2.1 Premature mortality

The literature review in Chapter 2 indicates that for short-time acute exposure to PM, an increase of 0.5-1% in all-cause mortality was generally observed for an increase of 10-$\mu g/m^3$ PM_{10}, based on studies in the U.S. as well as in Europe. However, it is also generally accepted that given the disparity in particle characteristics and various factors that influence public health status, results from studies conducted in an area should not be extrapolated to another area without cautious examination of the transferability.

With respect to the factors that affect the associations between ambient PM_{10} and health, developed and developing countries might be very different. For example, in less developed areas, people may use less health care to prevent illness, they may spend longer time outdoors, there may be less prevalence of air conditioning to reduce exposure to outdoor air pollution, and so forth. Given these reasons, this study focuses on epidemiological studies conducted in Thailand or other developing areas in Asia if possible.

To date two epidemiological studies have been conducted to examine the associations between daily PM_{10} concentrations and daily mortality in Bangkok (Ostro et al, 1999; and Vajanapoom et al, 2002). The following table summarizes their findings.

Table 3.6 Studies on the Associations Between Daily PM_{10} Concentrations and Mortality in Bangkok

Study	Study Period	Change in Daily Mortality per 10-$\mu g/m^3$ Increase in Daily PM_{10}		Weight
		Best Estimate	95% CI	
Ostro et al, 1999	1992-1995	1.0%	0.4-1.6%	0.3
Vajanapoom et al, 2002	1992-1997	0.77%	0.43-1.1%	0.7
Weighted Average		**0.84%**		

These results indicate a large level of congruence in CR functions for acute mortality from Thailand and the U.S. as well as Europe. A weighted average of these two studies is used in my analysis. As the second study estimated the CR coefficient based on data over a longer time period, and used multi-site mean monitoring data whereas the first one used only data from one monitoring site, a larger weight was assigned to the second study.

However, these studies did not consider source-specific PM health risk. Given that this study aims at estimating the health risk reductions associated with the decrease of PM emissions from mobile sources, the CR functions obtained by these studies may not accurately capture the changes in health outcomes associated with the policy scenarios considered here. To investigate whether particles from distinct sources pose different risks on exposed population, a stratified meta-analysis approach was used to pool estimates from similar studies. This analysis was performed below.

3.2.2.2 A stratified meta-analysis of time-series studies on PM_{10} and acute mortality

In recent years, related research is increasingly addressing the contribution of the characteristics of particles (size, shape, composition and origin) to the heterogeneity of PM health effects found across different studies, in particular, the role of the sources of particles on the severity of health impacts. Some studies argued that combustion-related sources particles were exclusively associated with adverse health effects such as increased daily mortality, whereas particles from natural sources can be ruled out (Laden et al, 2000). A review of previous epidemiological studies indicates that there is a consensus that particles from motor vehicle sources pose a greater risk for public health than particles from other sources. However, a quantitative estimate of the distinction between source-specific PM_{10} risks is not yet clear. The purpose of performing a meta-analysis in this study is to fill this gap.

Meta-analysis is a two-stage process involving the calculation of an appropriate summary statistic for each of a set of studies followed by the combination of these statistics into a weighted average (Egger et al, 2001). Furthermore, two important terms are *fixed effect model* and *random effects model*. In a fixed effect model, it is assumed that the true effect of treatment is the same value in each study, or fixed, and each effect estimate is a random sample from a single underlying distribution $N(\theta, \sigma^2)$, and thus only within-study variability is considered, whereas in a random effects model, each effect estimate is drawn from $N(\theta_i, \sigma_i{}^2)$, where θ_i are random values drawn from $N(\theta, \tau^2)$, accounting for both between-study and within-study variability (Levy, et al, 2000). There is a great deal of debate about whether it is better to use fixed or random effects models, and the two models can provide very different answers. Generally speaking, random-effects meta-analyses are more conservative (the confidence intervals are wider) than fixed effect meta-analyses (Egger et al, 2001).

The general formula for deriving a summary (pooled) estimate in meta-analysis is (Egger et al, 2001, p. 291-298):

$$\theta = \frac{\sum w_i \theta_i}{\sum w_i} \tag{3-7}$$

Where,

θ: Pooled estimate;

w_i : Weight for each estimate (noted as w_i' in random effects model);

θ_i : The natural logarithm of relative risk in this study.

For fix effect model (inverse variance method):

$$w_i = \frac{1}{SE(\theta_i)^2} \tag{3-8}$$

And for random effects model (DerSimonian and Laird random effects model):

$$w_i' = \frac{1}{SE(\theta_i)^2 + \tau^2} \tag{3-9}$$

Where, τ^2 is the inter-study variation, given by:

$$\tau^2 = \frac{Q-(k-1)}{\sum w_i - (\frac{\sum w_i^2}{\sum w_i})} \tag{3-10}$$

The w_i are calculated in Equation (3-8). And Q is the heterogeneity statistic, given by:

$$Q = \sum w_i (\theta_i - \theta)^2 \sim \chi^2_{k-1} \tag{3-11}$$

Where, k is the number of studies included in the meta-analysis.

Finally, the null hypothesis (H_0) is:

H_0: $\theta_1 = \theta_2 = ... = \theta_i = \theta$

H_1: At least one θ_i is different from θ.

A stratified meta-analysis is a special type of meta-analysis, which involves grouping the estimates according to a particular feature or characteristic of some explanatory variable, and carrying out a separate meta-analysis within each group. This study groups health effect estimates by the major sources of PM_{10} in each studies. Based on the most commonly reported sources in epidemiological studies, in conducting the meta-analysis this study categorizes sources of particles into three categories: mobile, industrial and crustal sources, and performs a separate meta-analysis within each source group.

In performing this meta-analysis, the author reviewed all accessible time-series studies on PM_{10} and acute mortality from Medline (National Library of Medicine) database, U.S. EPA's PM research publication list, as well as references of published articles (including individual analyses and meta-analyses). There was no restriction on the location of studies. The criteria to evaluate and select studies to be included in the meta-analysis are:

- Including studies in which a single major source of PM_{10} was reported in the study area, but excluding studies that did not report the sources of PM_{10} as well as studies that reported more than one major source;
- Coal combustion is considered as industrial sources;
- Including studies on mortality of all ages (excluding studies focusing on infants, children or elderly mortality);
- Including studies that reported both the mean estimates of effect and the precision of estimation (confidence intervals);
- If more than one study have been conducted using the same population, include the most recent one, regardless whether the research groups are the same.

In general, a minimum of four available estimates is required to conduct a meta-analysis. In addition, the estimated health effects of PM_{10} were converted into a common metric of the percentage change of mortality per 10-$\mu g/m^3$ increase in ambient PM_{10} concentrations.

The literature survey indicated, not surprisingly, that most epidemiological studies did not report or separate PM_{10} from different sources in examining its health effects, particularly studies before 2000. Moreover, many studies reported more than one major PM_{10} sources in the study area. As a result of data collection and evaluation, four to six studies were included in each source group as follows:

Table 3.7 A Meta-Analysis of Time-Series Studies on Daily PM_{10}-Mortality Associations Stratified by Particulate Sources

Publication (Year)	Location	Period	% Change in Daily Mortality per 10-μg/m^3 Increase in PM_{10} (95% CI in Parenthesis)
Mobile Sources			
Verheoff, A. P. (1996)	Amsterdam, Netherlands	1986-1992	0.58% (-0.10%, 1.31%)
Wong, T. W. (2002)	Hong Kong, China	1995-1998	0.80% (0.10%, 1.40%)
Bremner, S. A. (1999)	London	1992-1994	1.29% (0.28%, 2.32%)
Laden F. (2000)	Six U.S. cities	1979-1988	3.40% (1.70%, 5.20%)
Industrial Sources			
Venners, S. (2003)	Chongqing, China	1995	0 (-0.72%, 0.68%)
Peters, A. (2000)	The coal basin in Czech Republic	1982-1994	0.94% (0.07%, 1.81%)
Goodman, P. G. (2004)	Dublin, Ireland	1980-1996	0.4% (0.3%, 0.6%)
Tsai, S. (2003)	Kaohsiung, Taiwan	1994-2000	0 (-0.91%, 0.82%)
Pope III, C. A. (1992)	Utah Valley, USA	1985-1989	1.50% (0.87%, 2.09%)
Laden F. (2000)	Six U.S. cities	1979-1988	1.1% (0.3%, 2.0%)
Crustal Sources			
Ostro, B. D. (2000)	Coachella Valley, California	1989-1992	0.41% (-0.42%, 0.81%)
Pope III, C. A. (1999)	Salt Lake City	1985-1995	0.8% (0.3%, 1.3%)
Slaughter J. (2005)	Spokane, Washington	1995-2001	0 (-1.21%, 1.19%)
Laden F. (2000)	Six cities U.S.	1979-1988	-2.3% (-5.8%, 1.2%)

The results of pool-estimates and the Q heterogeneity statistics from the stratified meta-analysis are summarized in the following table and graph.

Table 3.8 Pooled Estimates and Tests of Heterogeneity

Sources	Fixed Effect Model	Random Effects Model	Q statistic	p	Null Hypothesis (α =0.05)
Mobile	0.96% (0.54%, 1.38%)	*1.22%* (0.41%, 2.04%)	9.402	<0.025	Rejected
Industrial	0.45% (0.31%, 0.58%)	*0.58%* (0.08%, 1.07%)	15.769	<0.01	Rejected
Crustal	*0.55%* (0.19%, 0.92%)	0.47% (-0.046%, 0.99%)	4.435	>0.10	Not rejected

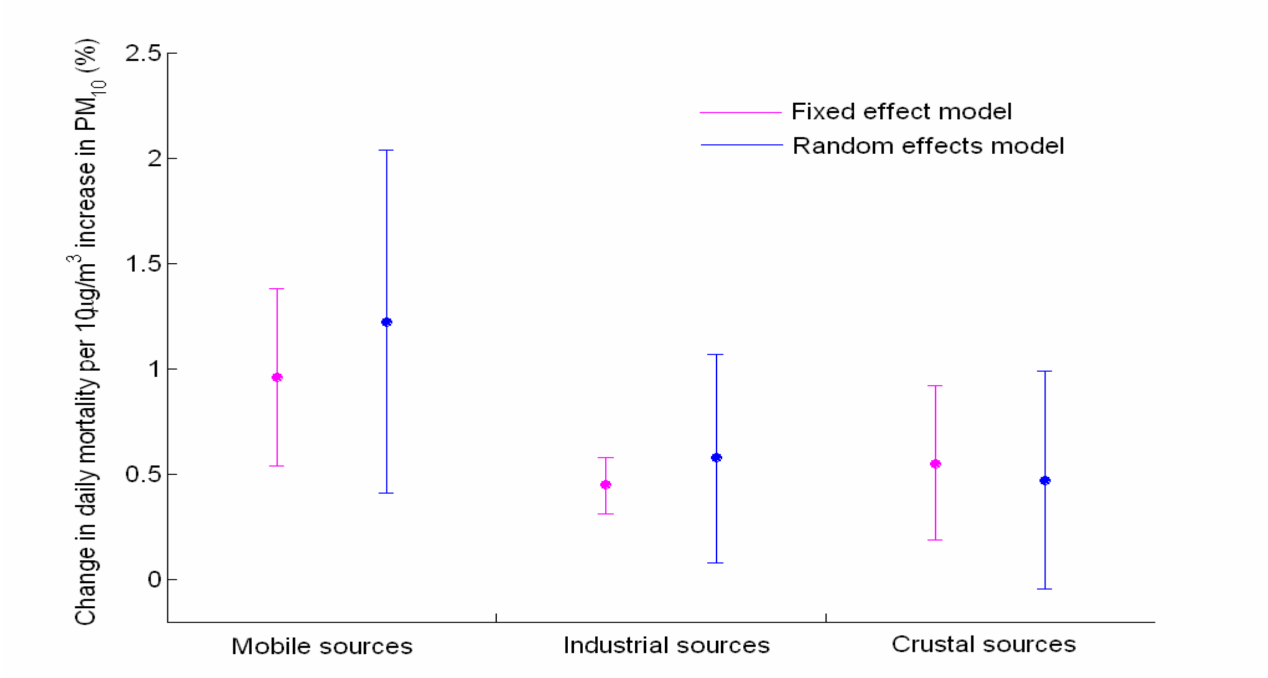

Figure 3.2 Pooled Estimates of 10-μg/m^3 Increase in PM$_{10}$ from Different Sources

The criterion used to select the statistical model (fixed effect or random effects) is that if the null hypothesis, i.e. the studies included in meta-analysis share a common mean value, is rejected, random effects model is adopted; otherwise, a fixed effect model is used. Based on this, the final results of pooled estimates are as follows:

Table 3.9 Pooled Estimates of PM_{10}-Mortality Associations by the Stratified Meta-analysis

Sources	Pooled Estimates (95% CI in Parenthesis) of the Change in Daily Mortality per 10-μg/m^3 Increase in PM_{10} Concentrations
Mobile	1.22% (0.41- 2.04%)
Industrial	0.58% (0.08 - 1.07%)
Crustal	0.55% (0.19- 0.92%)

The results indicate that the effects of PM_{10} from mobile sources are about two times larger than those of PM_{10} from industrial and crustal sources, whereas no appreciable difference between the effects of industrial and crustal sources was found. This implies that traffic-related PM_{10} in the air poses the greatest risk on human health. By and large, the effects of PM_{10} from mobile sources are approximately 1.5 times larger than the overall health effects of PM_{10} (0.5-1% per 10-μg/m^3 increase in PM_{10}). Based on the results by the meta-analysis, the estimates of PM_{10}-mortality in Bangkok from previous epidemiological studies are multiplied by a factor of ***1.5*** to obtain the health effects per unit exposure associated with mobile sources, given that the earlier epidemiological studies in Bangkok were based on the mixed effects of PM_{10} from all sources. The selected epidemiological studies as well as the adjusted estimates used in this study are listed in the following table.

Table 3.10 Studies on the Associations Between Daily PM_{10} Concentrations and Mortality in Bangkok

Health Endpoints	Reference	Study Area	Study Period	Change in Daily Mortality per 1-µg/m^3 Increase in Daily PM_{10}		Weight
				Best Estimate	95% CI	
Acute Adult Mortality	Ostro et al, 1999	Bangkok, Thailand	1992-1995	0.10%	0.04%-0.16%	0.3
	Vajanapoom et al, 2002		1992-1997	0.077%	0.043%-0.11%	0.7
	Weighted Average			0.084%	0.042%-0.125%	
	Adjusted Estimate (*1.5)			**0.13%**	**0.063%-0.188%**	
Acute Infant Mortality	Loomis et al, 1999	Mexico City, Mexico	1993-1995	0.69%	0.25%-1.30%	
	Adjusted Estimate (*1.5)			**1.04%**	**0.375%-1.95%**	

In addition, the reductions in PM_{10} levels and their health effects are estimated on the same day regardless of the possible time-lag after exposure, since the time-lag of health effects does not affect the aggregated estimate over a year. Acute health effects for each day are aggregated over an entire year to obtain the annual estimates.

3.2.2.3 Morbidity

Compared with epidemiological studies on mortality, studies on the association of morbidity and air pollution exposure are much less comprehensive (Wang & Mauzerall, 2006). Furthermore, studies on a few health endpoints are not available in Thailand. In this case, a relevant study in another developing country in Asia (e.g. China) is used. If both types of studies are not available, a study conducted in the western developed world (e.g. the U.S.) is selected. To reflect the source-specific PM morbidity risks, in the absence of information specific for morbidity, the same adjustment factor found for mortality (1.5) is used, assuming a linear relationship between mortality and morbidity. Although this assumption may introduce bias, given that morbidity is only responsible for a small portion of benefits, it is expected that it does not affect the overall estimates of health benefits considerably. The following table lists the references from which the CR function for each health endpoint are abstracted.

Table 3.11 Concentration-Response Coefficients for Morbidity in This Study

Health Endpoints	Pollutant Indicator	Percentage Change (95% CI) per 1-μg/m^3 Change in the Pollutant (1)	Adjusted CR Coefficient (*1.5)	Age Group	Reference	Study Area	Study Type
Chronic bronchitis	$PM_{2.5}$	0.45% (0.13-0.77%)	0.68% (0.20-1.16%)	All ages	Chen et al, 2002	China	Cohort
Respiratory hospital admissions	PM_{10}	0.18% (0.09-0.27%)	0.27% (0.18-0.36%)	All ages	Hagler Bailly, 1998	Bangkok	Time-series
Cardiovascular hospital admissions	PM_{10}	0.18% (0.10-0.26%)	0.27% (0.19-0.35%)	All ages	Hagler Bailly, 1998	Bangkok	Time-series
Emergency room visit	TSP (2)	0.006% (-0.003-0.015%)	0.015% (0.006-0.024%)	All ages	Xu et al, 1995	China	Time-series
Acute asthma attacks	PM_{10}	0.39% (0.19-0.59%)	0.59% (0.39-0.79%)	Adults >= 15	Chen et al, 2002	China	Time-series
		0.44% (0.27-0.62%)	0.66% (0.49-0.84%)	Children < 15			
Acute respiratory symptom days (3)	PM_{10}	0.3 (0.22-0.74)	0.45 (0.33-1.11)	All ages	Hagler Bailly, 1998	Bangkok	Time-series
Restricted activity days (4)	PM_{10}	0.058 (0.029-0.091)	0.087 (0.044-0.137)	Age >=18	Hagler Bailly, 1998; Ostro, 1987	USA	Time-series

Note:

(1) Concentration-response coefficients for air pollutants are usually considered to be normally distributed in literature (e.g. Blackman et al, 2000)

(2) A study suggests a PM_{10}/TSP ratio of 0.6 in Bangkok (Hagler Bailly, 1998).

(3) Units of acute respiratory symptom days and restricted activity days are per capita concentration-response for 1-μg/m^3 change in annual average PM_{10}. These risk factors are abstracted from Hagler Bailly, 1998. These factors are estimated by using CR coefficients from the original regression analysis and baseline incidence rates, but they are not provided in the documents. Therefore, the risk factors are applied directly here.

(4) The original study only surveyed adults of ages 18-65, who were working at the time when the study was carried out, for three reasons: first, restrictions in activity are easier to detect for workers since their time is usually more structured than that of non-workers; second, workers daily activity patterns tend to be similar; and third, the time and length of exposure to a given outdoor air pollutant tend to be similar. In anticipation that PM air pollution will lead to reduced activities, if the effect exists, among the general population rather other among the working ages exclusively, the CR coefficient is applied to all age groups, assuming that the effect is the same among the entire population.

3.3 Estimating the adverse health outcomes

The selected CR functions for each health endpoint are used in health impact functions to calculate the changes in health outcomes. As discussed in Section 2.1.5, this study employs an exponential CR function for PM health effects[18]. Therefore, the health impact function is:

$$\Delta y = y_0 \times (e^{\beta \cdot \Delta x} - 1) \qquad (3\text{-}12)$$

(Deck et al, 2001; Hubbell et al, 2005; Ostro et al, 2006; Sanhueza et al, 2003)

Where, Δy is the change in the number of each health endpoint (i.e. mortality or morbidity); y_0 is the baseline incidence, equal to the baseline incidence rate (I) times the potentially affected population POP:

$$y_0 = I \times POP \qquad (3\text{-}13)$$

And β is the concentration-response coefficient estimated from a corresponding damage function, and Δx is the estimated change in the ambient PM levels (daily 24-hour average measure for acute effects and annual average measure for chronic effects). The baseline incidence rates of some health endpoints are not available in Thailand. In this case, a similar strategy as in obtaining the CR functions is used. That is, a baseline incidence rate is in another developing country in Asia has the priority to be

[18]Some studies use a linear relationship of health response to air pollution at each health endpoint (e.g. Li et al, 2004, and Wang & Mauzerall, 2006). A linear health impact function is: $\Delta y = y_0 \times \beta \times \Delta x$, where all parameters are defined the same as in the health impact function derived from an exponential model.

selected to approximate the rates in Thailand. Otherwise, a rate in a western developed country is used. The following table summarizes the baseline incidence rates included in this study and their references.

Table 3.12 Baseline Mortality and Morbidity Incidence Rates (Year: 2000)

Health endpoints	Rate (1)	Reference	Study area
Adult natural death (2)	0.00371	Thailand Population Statistics 2000	Bangkok, Thailand
	0.00423		Samut Prakarn, Thailand
	0.00404		Nonthaburi, Thailand
	0.00367		Pathum Thani, Thailand
	0.00431		Nakhon Pathom, Thailand
	0.00494		Samut Sakhon, Thailand
Infant death	0.022		Thailand
Chronic bronchitis	0.0139	Chen et al, 2002; Wang & Mauzerall, 2006	China
Respiratory hospital admissions	0.0031	Hagler Bailly, 1998	Bangkok, Thailand
Cardiovascular hospital admissions	0.0028	Hagler Bailly, 1998	Bangkok, Thailand
Emergency room visit	0.238	Metzger et al, 2004	Atlanta, USA
Acute asthma attacks (<15)	0.0693	Chen et al, 2002; Wang & Mauzerall, 2006	China
Acute asthma attacks (>=15)	0.0561	Chen et al, 2002; Wang & Mauzerall, 2006	China
Acute respiratory symptom days	N/A		
Restricted activity days	N/A		

Note:

(1) Units of rates are cases *per year per person* in the exposed population.

(2) Natural deaths are all deaths other than accidents, suicides and homicides. In Bangkok, about 93% of deaths are natural (Hagler Bailly, 1998)

Premature mortality is also measured based on the years of life lost (YOLL), due to the factor that when an individual dies prematurely due to long-term exposure to air pollution, he or she may lose only a few years of his or her life (Rabl, 2003; Wang & Mauzerall, 2006). It is argued that depending on whether economic valuation is based on the number of lives lost or YOLL, the perceived health benefits of pollution mitigation may vary sufficiently to alter the results of a cost-benefit analysis (Wang & Mauzerall, 2006). However, given that there is no consensus on a methodology for estimating the economic value of a YOLL (Wang & Mauzerall, 2006), this study only considers the number of lives lost in economic valuation. The following section presents the details.

3.4 Economic valuation

To translate the benefits of reduced health risks associated with reductions in ambient concentrations of air pollution into economic terms, it is well accepted that the most appropriate economic valuation method is willingness to pay (WTP) (e.g. Freeman, 1993; Hubbell et al, 2005; Levy et al, 2001; Sanhueza et al, 2003). However, WTP estimates generally are not available for some health effects (Hubbell et al, 2005). In the absence of WTP estimates, cost-of-illness (COI), which is more accessible from previous studies (e.g. Smith et al, 1997; Stanford et al, 1999), is often used to evaluate health effects. A COI study estimates two types of costs of an illness: the direct costs (medical and non-medical) associated with the illness, and the indirect costs associated with lost productivity due to morbidity or premature mortality (Haddix et al, 1996). The weakness of COI is that it generally underestimates the true value of reducing the risk of a health effect, because it does not reflect the value of avoided pain and suffering (Hubbell et al, 2005). Therefore, COI should be used along with WTP to provide reliable economic measures. In addition, in the case that WTP for a certain health outcome is absent, WTP can be estimated based on empirical evidence for the ratios between WTP and COI. For instance, Rowe and Chestnut (1986) in their study of both WTP and COI for changes in asthma symptoms found that the ratio of WTP to COI ranged from 1.3 to 4. Levy *et al* (2001) suggested that we could calculate the COI of each health endpoint and regard it as a lower bound and place an upper bound at four times the COI and assign the probability weight 0.25 to both the lower and upper bounds, and 0.5 to the central value. These approaches, although involving uncertainties, can provide good estimates of the true economic values and their uncertainties in the case that WTP studies are lacking.

Another issue of concern is that currently there is no single WTP study available in Thailand to determine the economic values to reduce a risk of being affected by air pollution (Thanh & Lefevre, 2001). Since usually WTP is significantly affected by some factors such as income, values of WTP may not be readily transferred from a country to another before some proper adjustment is done (Thanh & Lefevere, 2001). However, scholars have developed several approaches to make such a transfer. The simplest one corrects only for the income difference between countries (Thanh & Lefevere, 2001). That is, in transferring the values from a country to another, only the difference in per capita income is accounted for and the values are assumed to be proportional to income. Therefore, based on the disparity in per capital incomes of the U.S. ($29760, Bureau of Economic Analysis, 2003) and Thailand ($1345, National Statistical Office Thailand, based on the exchange rate of 1:43.23 in 2000[19]), a ratio of 0.046 is derived. However, economists have suggested that income elasticity of WTP is less than one, which means that 'other things being equal, WTP in low-income countries is lower than WTP in high-income countries, but proportionately less than the income differentials' (Alberini et al, 1997; Thanh & Lefevere, 2001). Thanh and Lefevere (2001) suggested a ratio of 0.285 to be used to extrapolate U.S. values to Thailand. This study uses a ratio of 0.2 to extrapolate values of all health end points from U.S. studies to the current Thailand context. The same ratio was used in previous World Bank reports on Thailand (Hagler Bailly, 1998). The WTPs or COIs for each health endpoints used in this study, as well as their probability weights, are abstracted from existing literature. The following table

[19]Source of exchange rate: Bank of Thailand. (2007). Average exchange rate: Historical data. http://www.bot.or.th/BOTHOMEPAGE/Databank/EconData/EconFinance/Download/Tab50-1_E.xls, accessed January 18, 2008.

summarizes the values of mortality and morbidity in this study.

Table 3.13 Valuation of Health Endpoints in This Study

Health Endpoints	US Values (2000$) (1) (Selected probability weights in parenthesis)			Thailand Values (2000$) (2) (Selected probability weights in parenthesis)			Type of Estimate
	Low (0.33)	Central (0.5)	High (0.17)	Low (0.33)	Central (0.5)	High (0.17)	
Mortality	$2,372,835	$4,067,717	$8,248,425	$474,567	$813,543	$1,649,685	WTP
Chronic bronchitis	$169,488	$248,583	$440,669	$33,898	$49,717	$88,134	WTP
Respiratory hospital admissions	$7,909	$15,819	$23,728	$1,582	$3,164	$4,746	Adjusted COI (3)
Cardiovascular hospital admissions	$8,474	$16,949	$25,423	$1,695	$3,390	$5,085	Adjusted COI
Emergency room visit	$294	$588	$881	$59	$118	$176	Adjusted COI
Acute asthma attacks	$21	$55	$63	$4.2	$8.4	$12.6	WTP
Acute respiratory symptom days	$7	$14	$19	$1.4	$2.7	$3.8	Adjusted COI
Restricted activity days	$35	$70	$105	$7	$14	$21	WTP and Adjusted COI

Note:

(1) The source of the values and their selected probability weights is Ostro and Chestnut (1998), with the exception of the value of acute asthma attacks, which is abstracted from EPA (1997b). Inflation is calculated using the Inflation Calculator by U.S. Department of Labor, Bureau of Labor Statistics: http://data.bls.gov/cgi-bin/cpicalc.pl.

(2) Thailand value = US value X 0.2.

(3) Adjusted COI is COI multiplied by 2 to approximate WTP, calculated by the authors of the reference.

The economic value associated with change in health outcome (V_i) is given by

$$V_i = A_i \times WTP_i \qquad (3\text{-}14)$$

Where, A_i is the change in the number of a specific health endpoint i, and WTP_i is the willingness to pay to avoid the corresponding health endpoint.

3.5 Evaluating the effectiveness of vehicle inspection and maintenance programs: A policy model

I/M programs are one of the essential policy tools to control emissions from in-service vehicles. An I/M program has the potential to reduce emissions in a number of ways, such as better maintenance of vehicles by motorists as a result of the program, repairs made in anticipation of an I/M inspection (referred to as pre-test repairs) or as a result of failing the test, and early scrapping of vehicles that are not worth repairing (National Research Council, 2001). The following graph illustrates the sources of emissions reductions resulting from an I/M program.

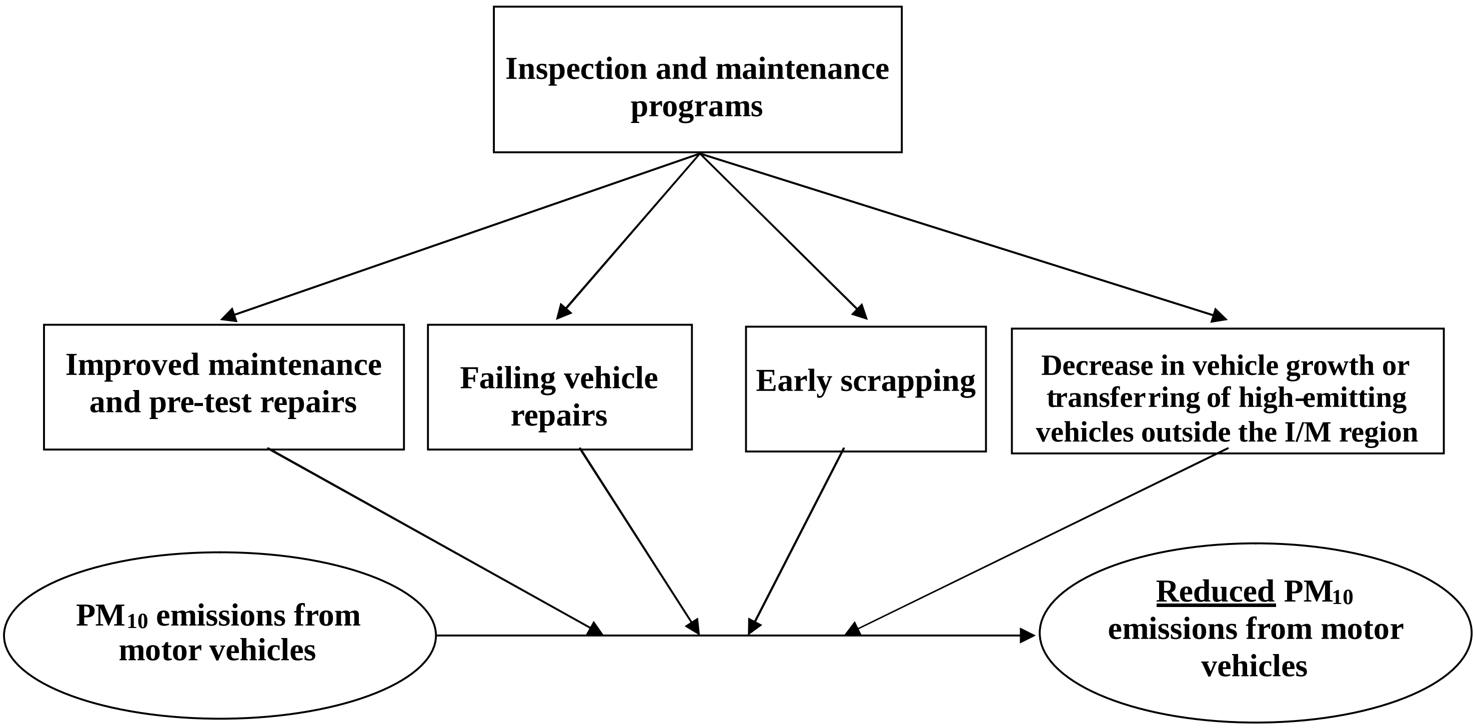

Figure 3.3 Sources of Emission Reductions from I/M Programs (Partial Information Source: National Research Council, 2001)

Based on this conceptual framework, Eisinger (2005) developed a spreadsheet tool to evaluate the effectiveness of I/M programs, i.e. levels of emission reductions by those programs. The theoretical basis of this tool is that the amount of vehicle emission reductions resulting from an I/M program is a function of the following variables (Eisinger, 2005):

- Pre-I/M test repair work: potential vehicle repairs motivated by instituting I/M for the vehicles that would otherwise be identified as problems.
- Post-I/M test repair work: emission reductions from repair work for the vehicles identified as problems by inspections. This variable itself is a function of the number (or the percentage) of the problem vehicles identified by I/M and the number (or the percentage) of identified problem vehicles being repaired effectively.
- Vehicles scrapped: emission reductions will be generated from scrapping gross polluting vehicles that would otherwise continue to be used, and replacing them by

low emission vehicles. This variable also includes high-polluting vehicles that are transferred outside the I/M region due to the implementation of the programs.

Appendix I includes the complete conceptual descriptions and mathematical formulas of the tool. Overall, the I/M program design spreadsheet tool developed by Eisinger (2005) allows users to adjust the values of parameters in the model and obtain the resulting percentage of emission reductions in total vehicle emissions. Although the tool is developed in the U.S. for the most common gasoline vehicle I/M programs, the fundamental ideas of vehicle I/M programs are universal and thus the theoretical modeling framework applies to I/M programs targeting other pollutants in other regions. However, the values of the variables must reflect the specific contexts and issues of concern with respect to the interested area and programs[20].

The main purpose of applying this state-of-art I/M policy model in this study is to link I/M design considerations with health benefits associated with the programs in order to understand the impacts of some key issues regarding I/M design, such as compliance rates, testing cut-points and effectiveness of repairs, on the potential health benefits of the programs.

In addition, the policy model Eisinger (2005) is a static model which only considers the current fleets, whereas the impacts of I/M programs on the growth of vehicle population over time is beyond the scope of the original study. However, given that this study aims at evaluating the future I/M benefits as the features of vehicle fleets change over time, the impacts of control policies on vehicle growth in future years need to be taken into account. Further details on the modifications of the original model are discussed in Chapter 5.

[20]Conversation with the author of the paper, Dr. Douglas S. Eisinger.

Chapter 4 Health Benefits of Control Policies in Transportation Sector

4.1 Health damage in the BMR attributable to PM_{10} from motor vehicle sources in 2000

Based on the assumption that motor vehicle sources account for 31.62% of total PM_{10} emissions in the BMR in the year 2000 (this is the mean value used in this study, as discussed in Chapter 3), the total health damage costs due to exposure to traffic-related fine particulate matter are approximately 2678 million 2000 dollars in that year. Premature deaths account for 1369.2 million dollars or 51.1% of the total health damage costs. The estimated health impacts (number of cases) and health damage costs in 2000 are summarized in Table 4.1.

Table 4.1 Health Damage Due to Exposure to Traffic-Related PM_{10} in the Bangkok Metropolitan Region (Year: 2000)

Health Endpoints	Total Number of Cases	Health Damage Costs (Million 2000 U.S.$)
Death	1683	1369.2
Chronic bronchitis	17304	860.3
Respiratory hospital admissions	1564	4.9
Cardiovascular hospital admissions	1382	4.7
Emergency room visits	10682	1.3
Acute asthma attacks	67911	0.6
Acute respiratory symptom days	80562555	218.5
Restricted activity days	15575427	218.2
	Total costs	**2677.7**

The results indicate that the economic loss due to exposure to PM_{10} emissions from transportation is considerable in the BMR, which accounts for about 2.35% of Thailand's Gross Domestic Product (GDP) in 2000 (113.73 billion dollars, National Statistical Office Thailand)[21].

4.2 Health benefits associated with PM_{10} emission reductions from mobile sources

When the potential health benefits associated with projected emission reductions in future years are simulated, the possible changes in the key parameters over time need to be included in the model. The following parameters are expected to change significantly over time: (1) The total number of vehicles is likely to increase rapidly in a developing area such as the BMR; (2) The average PM emissions per vehicle-km are likely to go down due to the tighter emission standards set for new vehicles and the improvement in emission control technologies in modern vehicles; (3) Vehicle kilometers traveled per vehicle in the study area may increase as a result of the increase in demand for travel; and (4) Population are likely to grow fast as the result of rapid urbanization.

4.2.1 Projected changes in vehicle population and total vehicle emissions in the business-as-usual scenario

Using 2000 vehicle emission factors from the PCD, 1999 vehicle registration data from the Land Transport Department (LTD) and annual kilometers traveled data from the Radian International LLC (1998), Parsons (2001) estimated the 1999 mobile source components of PM emissions (Table 4.2)[22].

According to the Road Transport Statistics of Thailand, the vehicle fleet in the

[21]As the economic center of Thailand, the city of Bangkok itself produces 40-50% of the country's total GDP. Source: http://en.wikipedia.org/wiki/Bangkok, accessed October 24, 2007.

[22]The study assumed the vehicle emission rates (g/km) were the same in 1999 and 2000.

BMR grew at a rate of 6.2% per year on average over 1983 to 1999 (Parsons, 2001). The annual growth rate in vehicle registrations in the BMR is much higher than other more saturated markets. For example, the annual vehicle registration growth rate in the U.S. during the same period 1983-1999 was 1.7% (Federal Highway Administration, 2005). When the types of vehicles are examined for growth, the average annual growth rates vary significantly among different subcategories of vehicles. For example, light-duty trucks are the fastest growing category, growing at about 16% per year over the last ten years, motorcycles and passenger cars have increased by approximately 6% per year, whereas the total number of registered buses has remained virtually constant (Parsons, 2001). The annual growth rates for each automotive category used to estimate the vehicle population in future years are summarized in Table 4.2. It is assumed that the growth rates in the past will continue into the near future.

Table 4.2 Vehicle Population, Annual Growth Rates and Emissions Data in the Bangkok Metropolitan Region in 1999 (Source: Parsons, 2001)

Vehicle Type	Number of Vehicles (1999)	Average Annual Mileage (km)	PM Emission Rate (g/km) (2000)	Annual PM Emissions (tons/yr)	Share of Total PM Emissions (%)	Annual Growth Rate
City Bus	24928	97525	1.855	4510	31	1%*
City Truck	67253	16000	1.855	1996	14	3.3%*
Long Haul Truck/Bus	31819	12000	1.855	706	5	3.3%*
Light Duty Truck	664080	18075	0.398	4777	33	16%
Passenger Car	1317062	17171	0.005	113	1	6%
Motorcycle	1660119	10000	0.150	2490	17	6%
			TOTAL	**14595**	100	

**Note*: Thailand data, source: Pollution Control Department website[23].

The 2000 vehicle PM emission rates in Table 4.2 are the baseline emission rates in this study. As mentioned before, in the business-as-usual (BAU) scenario, although no further policy to control emissions from in-service vehicles is implemented, the

[23]PCD website, Thailand and BMR Diesel Vehicle Fleet. http://infofile.pcd.go.th/air/DIESEL2_Sayeg_Section_2.pdf, accessed October 16, 2007.

fleet-average PM emissions per vehicle-kilometer should decline over time due to the factors that: (1) Newer vehicle models enter the fleet and older ones are scrapped; (2) Over time, tighter emission standards for new vehicles are enforced, and emission performance in modern vehicles is greatly improved. For example, the emission standards for new light-duty vehicles in Thailand from 1995-2014 are summarized in the following table.

Table 4.3 Emission Standards for New Light-Duty Vehicles in Thailand (as of May 2007) (Source: Clean Air Initiative for Asian Cities, 2007)

Period	Emission Standard
1995-2000	Euro 1*
2001-2003	Euro 2
2004-2011	Euro 3
2012-2014	Euro 4

**Note*: Euro 1-4 stand for a series of European emission standards, which are sets of requirements defining the acceptable limits for exhaust emissions of new vehicles sold in Europe Union member states. The emission standards are defined in a series of European Union directives staging the progressive introduction of increasingly stringent standards. (Source: Wikipedia, http://en.wikipedia.org/wiki/European_emission_standards, accessed January 2, 2008) European emission standards are widely adopted in Asian countries.

However, Table 4.2 is the only available information on fleet-average emission rates for different types of vehicles in the BMR. It is rather difficult to estimate the change in average emission rates in future years resulting from the improvement in emission performance associated with new vehicles without conducting the same type of studies for the years after 1999. Moreover, fleet-average emission rates for in-use vehicles as well as their changes over time are barely studied to date. Some research was conducted in Beijing, China, for vehicle pollutants including NO_x, CO and HC, but not PM, due to the lack of PM emission data in the past (He et al, 2002). Given these facts, this study assumes a 5% annual PM emission factor decrease rate for all types of vehicles in the BMR[24]. The following graph summarizes the projected fleet-average PM_{10} emission rates in future years using the 1999 data and the 5% decrease rate assumption.

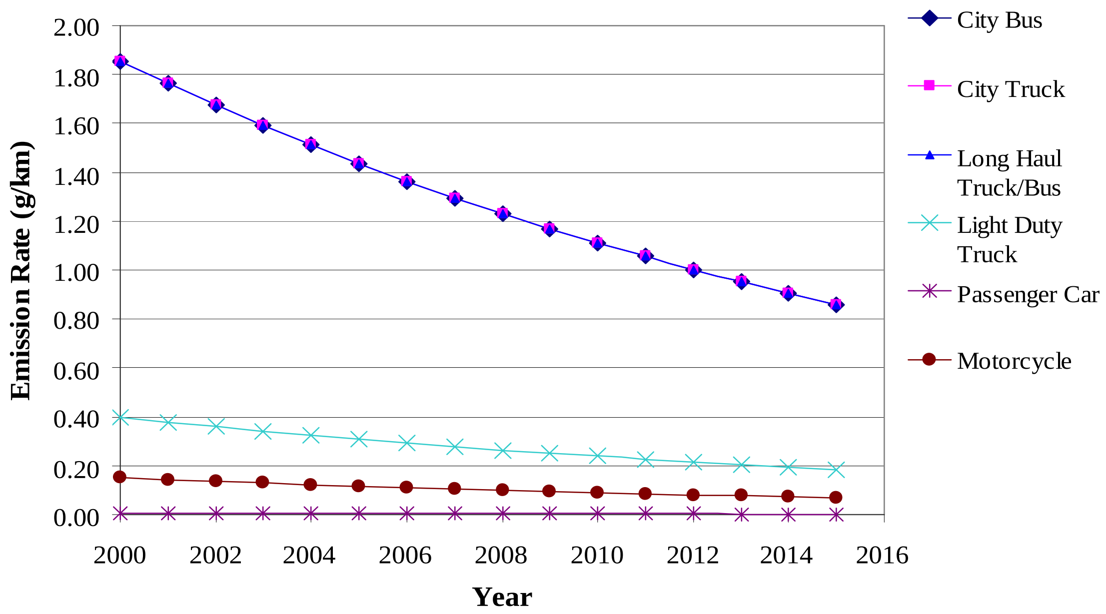

Figure 4.1 Fleet-Average PM_{10} Emission Rates Projection in the BMR, 2000-2015

Another issue is that the annual vehicle kilometers travelled (VKT) per vehicle may increase, as people's travel needs increase as the result of rapid economic growth and urbanization, particularly in a developing country, resulting in a potential increase in total emissions from motor vehicles if VKT growth exceeds the decline in emissions per kilometer. However, the increase in VKT per vehicle is more likely to happen among private vehicles such as private trucks, cars and motorcycles, since a large portion of commercial vehicles such as city buses and trucks usually run on fixed routes. Currently accurate information regarding the change in VKT per vehicle over time is not available in the BMR. I compared the 1999 data in Table 4.2 with the VKT per vehicle in Bangkok for several vehicle types in 2005 reported by Perera (2006). For available data on city buses, passenger cars and motorcycles, there is no significant increase in VKT per vehicle

[24]The same annual PM emission decrease rate was assumed in a World Bank study on the reductions in emissions from motorcycles in Bangkok (World Bank, 2003).

found between the two years. Given this finding, this study assumes that the average annual mileage per vehicle for all vehicle types remain unchanged over time.

The projected total annual PM_{10} emissions from motor vehicles in future years in the study area are estimated using the projected average emission rates, VKT, and projected number of vehicles. Based on the available data in Table 4.2, the projected vehicle population and emissions in the baseline year of this study, i.e. 2000, are estimated and summarized as follows.

Table 4.4 Baseline Vehicle Population and Emissions in the BMR (Year: 2000)

Vehicle Type	Number of Vehicles (1)	Average Annual Mileage (km) (2)	PM_{10} Emission Rate (g/km)	Annual PM_{10} Emission (tons/yr) (3)	Share of Total PM_{10} Emission (%)
City Bus	25177	97525	1.855	4555	29.1
City Truck	69472	16000	1.855	2062	13.2
Long Haul Truck/Bus	32869	12000	1.855	732	4.7
Light Duty Truck	770333	18075	0.398	5542	35.4
Passenger Car	1396086	17171	0.005	120	0.8
Motorcycle	1759726	10000	0.150	2640	16.9
			Total	**15650**	100

Note:

(1) The numbers of vehicles are estimated by using the annual growth rates in Table 4.2.

(2) Average annual mileages are assumed unchanged as discussed before.

(3) Total annual PM_{10} emissions = Number of vehicles X Average annual mileage X Emission rate.

The following graph summarizes the projected total particulate matter emissions from motor vehicles in the BMR under the BAU scenario during 2000-2015.

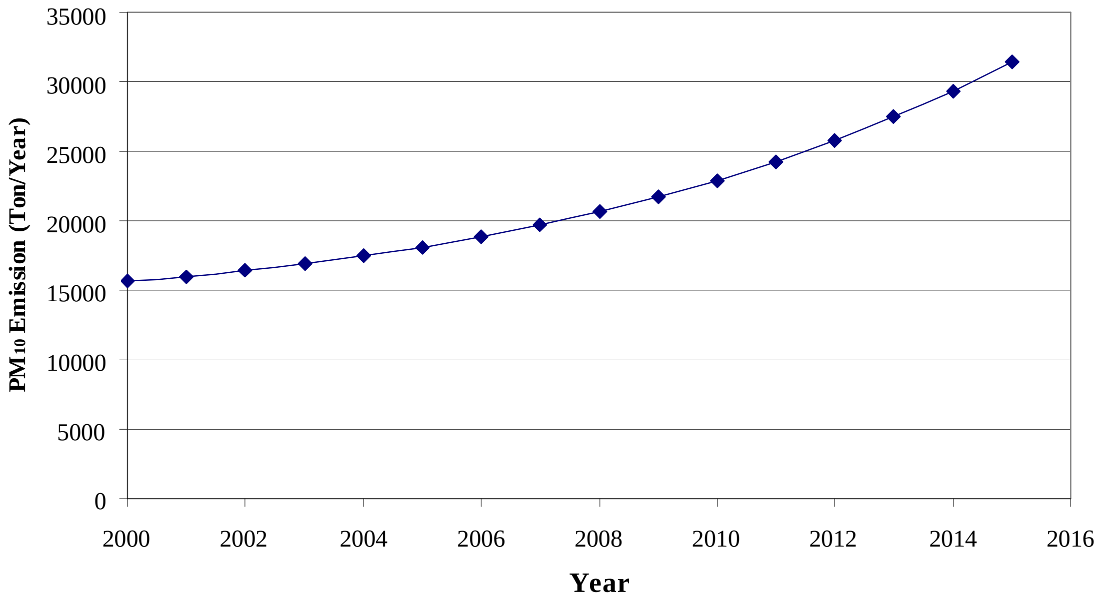

Figure 4.2 Projected Total PM_{10} Emissions from Motor Vehicles in the Bangkok Metropolitan Region: Business-as-Usual Scenario, 2000-2015

The graph indicates that under the BAU scenario, despite that the emission rates per vehicle are expected to go down over time as the result of improved vehicle performance, total vehicle PM emissions in the BMR will still increase, mainly attributable to the rapid growth in vehicle population in this developing urban area. For instance, total PM emissions from motor vehicles in 2008 are listed in the following table.

Table 4.5 Projected Vehicle Population and Emissions in the Bangkok Metropolitan Region in 2008: Business-as-Usual Scenario

Vehicle Type	Number of Vehicles	Average Annual Mileage (km)	PM_{10} Emission Rate (g/km)	Annual PM_{10} Emissions (tons/yr)	Share of Total PM_{10} Emissions (%)
City Bus	27263	97525	1.231	3272	15.8
City Truck	90077	16000	1.231	1774	8.6
Long Haul Truck/Bus	42618	12000	1.231	629	3.0
Light Duty Truck	2525471	18075	0.264	12053	58.4
Passenger Car	2225149	17171	0.003	127	0.6
Motorcycle	2804736	10000	0.100	2791	13.5
			Total	**20646**	100

Therefore, under the BAU scenario, BMR's total annual PM_{10} emissions from motor vehicles are estimated to be 20646 tons in 2008, approximately a 32% increase over PM_{10} emissions in the baseline year 2000. In particular, the relative contribution of light-duty trucks to total automotive PM emissions will increase from 35.4% in 2000 to 58.4% in 2008 due to the fact that they are the fastest growing vehicle category.

4.2.2 The impacts of emission control policies on the growth of vehicles and emissions

If new inspection and maintenance programs targeting all diesel-fueled vehicles and motorcycles are implemented, vehicle growth rates may diminish due to the regulation. However, data on new vehicle registrations in Bangkok in the past decade indicate that vehicle increase rates in this rapidly growing area are more significantly affected by the overall economic situation than by environmental regulation. The following figure illustrates the numbers of new vehicles registered in Bangkok during 1993–2005.

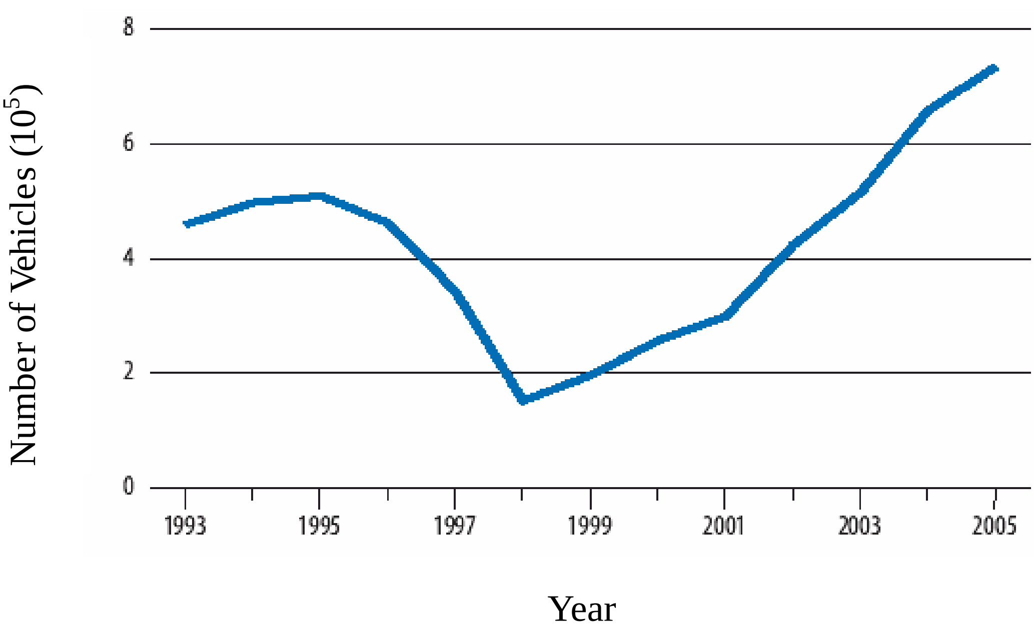

Figure 4.3 Numbers of New Vehicles Registered in Bangkok, 1993–2005
(Source: Asian Development Bank, 2006)

The decrease in new vehicle registration during 1993-1998 was largely due to the

Asian economic crisis in this period (Asian Development Bank, 2006). Figure 4.1 illustrates that although a series of emission control policies were introduced during 1999-2000, there was no significant decline in new vehicle registration observed in this period. New vehicle registration started to increase in 1999 at an average rate of approximately 22% annually, with more than 700,000 new units added in Bangkok alone in 2005 (Asia Development Bank, 2006). Figure 4.3, therefore, suggests that vehicle growth rates are driven by economic prosperity, rather than being related significantly to the presence or absence of regulatory controls.

When the subcategories of vehicles are examined, the trend seems to be similar. For example, the new motorcycle registrations as showed in the following graph.

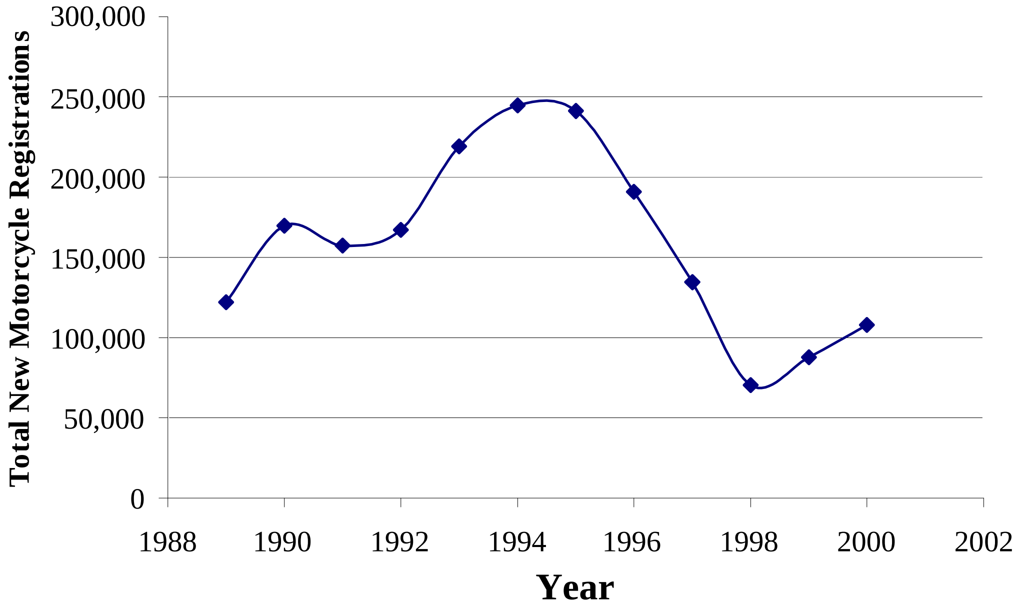

Figure 4.4 New Motorcycle Registrations in Bangkok, 1989-2000

Nevertheless, government policy interventions may still alter the trend in vehicle growth. Parsons (2001) reports that during 1990-1991, there was a decline in new vehicle registration in Bangkok, whereas new vehicle registrations in Thailand increased during the same period. They argue that the changes in Bangkok may be due to some policies implemented exclusively in the city, such as a change in the licensing fee and local value added tax.

Currently there is few empirical evidence on the impact of inspection and maintenance programs on vehicle growth in Asian developing countries. Beijing launched I/M programs targeting CO, HC and NO_x emissions from in-use vehicles in 1999 (Hao et al, 2006). Vehicle registration data in Beijing show that the average annual vehicle growth rate during 1999-2005 after the implementation of the programs was 12.1%, whereas the average rate during 1993-1998 before the programs was 18.3%, which means that there was a 34% decrease in the average annual growth rate after the programs[25]. However, there were other key vehicle emission control measures being launched in Beijing in 1999, such as the introduction of a more stringent (Euro 1) emission standard for new vehicles, completely phasing out lead in gasoline, etc (Hao et al, 2006). Therefore, it is difficulty to separate the impact of I/M from that of others.

Based on these limited evidence, this study assumes that a 10% decrease, or approximately one-third of the overall 34% decrease, in annual growth rate is attributable to the I/M programs. The assumption is used to project the change in annual growth rate for all diesel-fueled vehicles and motorcycles after the new PM-related I/M programs targeting these vehicles are introduced. The impacts of this assumption are examined in Chapter 5. And the growth rate for passenger cars is assumed to be unchanged since they are excluded from the PM-related I/M programs[26].

The levels of vehicle emission reductions achieved by an I/M program are heavily dependent of the design (e.g. emission testing cut-points) and implementation (e.g.

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[25]Vehicle registration data source: Beijing Transportation Center, 2005 Annual Transportation Report for Beijing (in Chinese).

[26]There may also be changes in the passenger car fleet. For example, some travelers are likely to switch to passenger cars if other vehicles are regulated more rigorously. However, these effects are not likely to have significant impacts on the total vehicle PM emissions since the emission projection in the BAU scenario shows that passenger cars only contribute to less that 1% of total vehicle PM_{10} emissions over time. Given this, assuming the same the growth rate of passenger cars under both the BAU and the I/M scenarios will not have significant impacts on the projection.

participation or compliance rates) of the program. The impacts of these factors on emission reductions, and consequently on the total benefits of the program, are discussed in Chapter 5. The analysis in this Chapter addresses the question: What level of vehicle emission reduction needs to be achieved by the I/M considered in order for the benefits of reducing health damages to outweigh the program implementation costs?

Parsons (2001) proposed PM_{10} emission reduction targets for each type of vehicle to be achieved by the new I/M programs targeting all diesel-fueled vehicles and motorcycles. They are summarized in the following table.

Table 4.6 PM_{10} Emission Reduction Targets by the PM-Related Inspection and Maintenance Programs in the Bangkok Metropolitan Region (Source: Parsons, 2001)

Vehicle Type	Percentage Reduction in Overall PM_{10} Emissions from a Specific Type of Vehicles	
Euro 0 Bus	25	20*
Euro 1 Bus	20	
Euro 2 Bus	15	
City Truck	25	
Light Duty Truck	25	
Long Haul Truck/Bus	25	
Motorcycle	30	

**Note*: 20% average reduction for all buses is an estimated value assuming that the number of vehicles and emission rates of the three bus types are identical. The assumption is made due to the lack of information about number of vehicles and average emission rates for each type of buses.

If the emission reduction targets can be achieved, i.e., the average PM_{10} emission rates of all vehicle types will be reduced to the target levels on the basis of the BAU scenario, the vehicle particulate emissions in 2008 under the I/M scenario would be:

Table 4.7 Vehicle Particulate Matter Emissions in the BMR under the Inspection and Maintenance Scenario (Year: 2008)

Vehicle Type	Number of Vehicles (2008)	Average Annual Mileage (km)	PM_{10} Emission Rate (g/km)	Annual PM_{10} Emissions (tons/year)	Share of Total PM_{10} Emissions (%)
City Bus	27236	97525	0.985	2615	17.0
City Truck	89789	16000	0.923	1326	8.6
Long Haul Truck/Bus	42481	12000	0.923	471	3.1
Light Duty Truck	2490636	18075	0.198	8915	57.9
Passenger Car	2225149	17171	0.003	127	0.8
Motorcycle	2788860	10000	0.070	1943	12.6
			Total	**15396**	100.0

The results in Table 4.7 indicate the target levels of PM_{10} emission reduction for each type of vehicles participating in the I/M programs (diesel vehicles and motorcycles) will result in an overall PM_{10} emission reduction of about 25% from motor vehicles relative to the BAU scenario in 2008.

4.2.3 Projected increase in population in the Bangkok Metropolitan Region

Population data are used to estimate the size of population exposed to air pollution. In projecting into the future years, the trends in population change in the past are usually used to forecast future population. The following table summarizes the average annual population growth rates in Bangkok and its five surrounding provinces over the period of 1999-2005.

Table 4.8 Average Annual Population Growth Rate in the Bangkok Metropolitan Region, 1999-2005

Province	Average Annual Population Growth Rate, 1999-2005
Bangkok	0.003%
Samut Prakan	1.59%
Nonthaburi	2.50%
Phathumthani	4.49%
Nakhorn Pathom	0.71%
Samut Sakorn	1.07%
Five Provinces	**2.10%**
BMR	**0.81%**
Thailand	**0.207%**

Source: Bangkok Metropolitan Administration, Strategy and Evaluation Department. 2006. Statistical Profile of Bangkok Metropolitan Administration 2005 (simplified as Bangkok Statistics, 2005).

These data indicate that the population growth rates of the five provinces are much greater than that of Bangkok. This may happen due to the trend that residents are moving out from the populous central city area to more suburban areas. On the other hand, the growth rate for the BMR as a whole is still greater than the rate for Thailand due to the rapid urbanization process occurring in the country. The average annual growth rates over 1999-2005 period for each province are used to predict the population in future years based on the population data in the base year 2000. The following graph shows the projected population growth in the BMR used in this study.

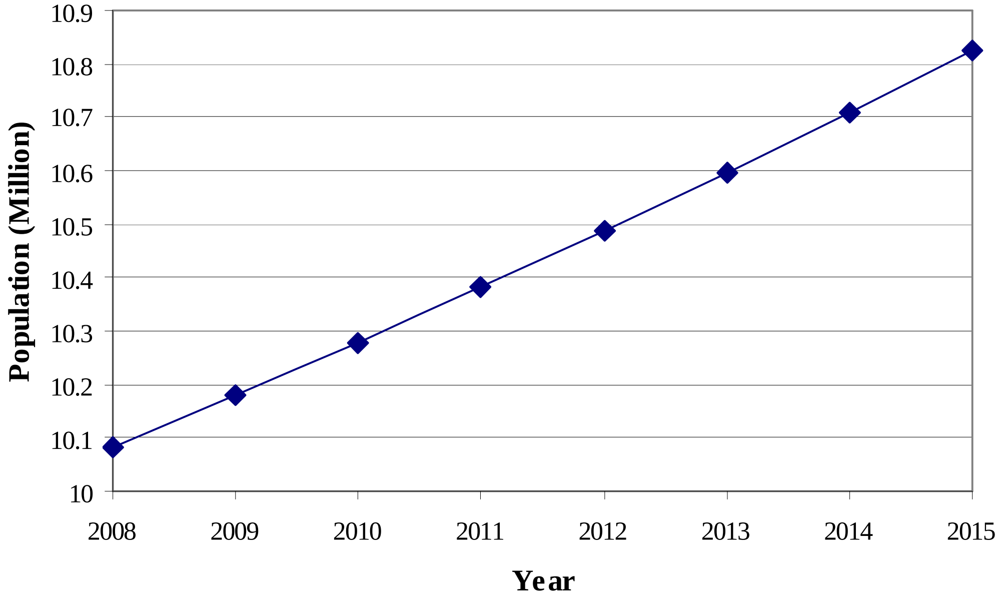

Figure 4.5 Projected Population in the Bangkok Metropolitan Region: 2008-2015

Moreover, in order to forecast the infant mortality due to PM pollution exposure in future years, the number of births needs to be predicted. Available data indicate that in Bangkok the average birth rate over the period of 2000-2005 increased 0.43% per year (Bangkok Statistics, 2005). Due to the lack of data for the surrounding five provinces, the birth rate increase in Bangkok is applied to all the six provinces in the BMR to predict the increase in the number of total births in future years.

4.2.4 Health benefits of the PM-related I/M programs

4.2.4.1 Health damage costs of PM_{10} from mobile sources in the BMR

As shown in Figure 4.2, with no further control policy, total PM emissions from motor vehicles in the BMR will increase rapidly mainly as the result of rapid growth in total number of vehicles. Consequently, the health damages in the BMR attributable to exposure to traffic-related particulate emissions are expected to increase substantially overtime relative to the base year, due to the increase in vehicle emissions as well as increase in population exposed to pollution. The approach, data, and mathematical equations using to estimate health damage costs of PM emissions from motor vehicles or the benefits of reducing emissions are described in detail in Chapter 3. Table 4.9 summarizes the annual health damages during 2008-2015, followed by a graph that illustrates the annual economic costs.

Table 4.9 Estimated Annual Health Damages Attributable to Particulate Matter Emissions from Motor Vehicles in the BMR: Business-as-Usual Scenario (Year: 2008-2015)

Year	2008	2009	2010	2011	2012	2013	2014	2015
Health endpoints (Cases)								
Death	2395	2549	2724	2923	3148	3404	3695	4025
Chronic bronchitis	25002	26673	28572	30728	33178	35962	39128	42730
Respiratory hospital admissions	2235	2379	2542	2725	2933	3167	3432	3730
Cardiovascular hospital admissions	1975	2103	2246	2409	2592	2799	3033	3296
Emergency room visits	15131	16073	17135	18330	19672	21179	22870	24767
Acute asthma attacks	98313	104928	112451	121006	130735	141803	154408	168776
Acute respiratory symptom days	114015832	121094401	129065313	138028833	148097364	159396930	172068849	186271617
Restricted activity days	22043061	23411584	24952627	26685574	28632157	30816740	33266644	36012513
Economic valuation (Million 2000 U.S. dollars)								
Health damages (mortality)	1948.3	2073.8	2216.3	2378.0	2561.4	2769.5	3005.8	3274.5
Health damages (morbidity)	1877.5	2000.0	2138.8	2295.9	2473.8	2675.2	2903.2	3161.5
Total health damages	**3825.7**	**4073.8**	**4355.1**	**4673.9**	**5035.2**	**5444.7**	**5909.1**	**6436.0**

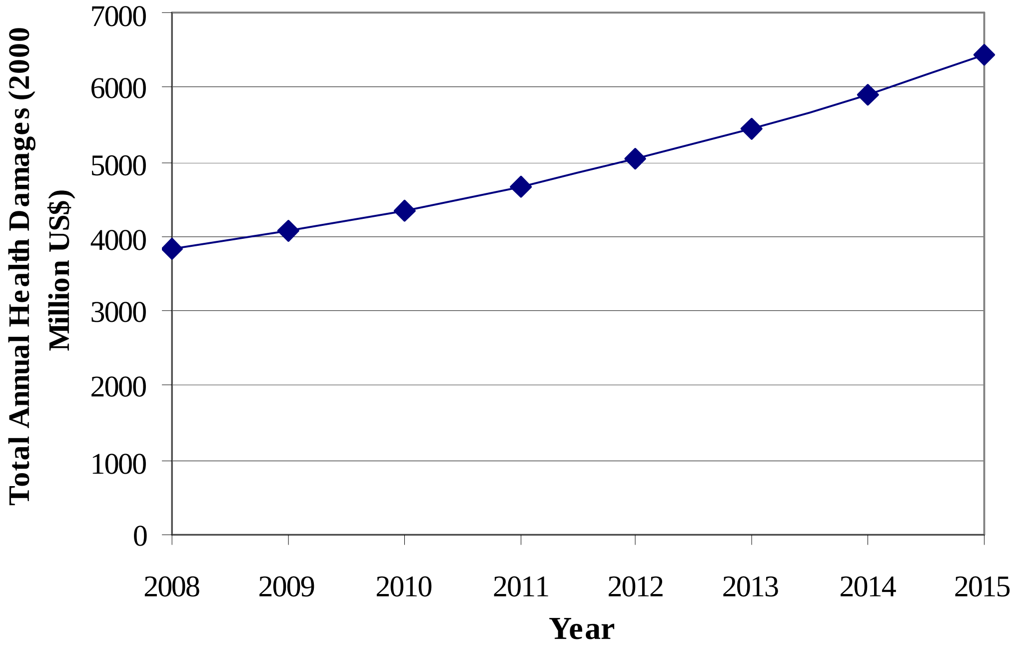

Figure 4.6 Health Damages Attributable to Particulate Matter Emissions from Motor Vehicles in the BMR: Business-as-Usual Scenario (Year: 2008-2015)

The results in Table 4.9 indicate that health damages due to PM_{10} from transportation sources will considerably increase relative to the baseline year 2000 if no control policy is implemented. For instance, the total damage cost in 2008 (3825.7 Million $) is estimated to about 1.5 times greater than that in 2000 (2677.7 Million $), whereas in 2015 (6436.0 Million $), the cost will increase by about a factor of 2.5 times relative to the base year of 2000.

However, implementing some emission control policies such as the PM-oriented I/M programs, if successful, is anticipated to considerably reduce PM_{10} emissions from the existing vehicle fleets and consequently produce significant health benefits. This is discussed next.

4.2.4.2 Potential health benefits of PM-related I/M programs

As discussed in Section 2.2, I/M programs targeting all diesel-fueled vehicles and motorcycles can potentially reduce 25% of PM_{10} emissions from mobile sources in the BMR. In reality, the actual levels of emission reductions by an I/M program may vary significantly, depending on a number of factors related to the actual performance of the program. This Chapter focuses on addressing the question of how large the benefits are if specific target emission reductions are achieved, and what level of vehicle emission reductions need to be achieved in order for the benefits to be greater than the costs. I first estimate the health benefits associated with the I/M programs if the emission reduction target is accomplished fully, i.e. a 25% vehicle PM_{10} emission reduction relative to the baseline. The following table summarizes the results for 2008-2015, followed by a graph that compares the total health damages attributable to vehicle PM emissions under the BAU and the I/M scenarios.

Table 4.10 Potential Health Benefits of I/M Programs Targeting 25% Particulate Matter Emission Reductions from Motor Vehicles in the Bangkok Metropolitan Region (Year: 2008-2015)

Year	2008	2009	2010	2011	2012	2013	2014	2015
Health endpoints avoided (Cases)								
Death	563	597	636	679	728	783	844	913
Chronic bronchitis	5863	6234	6653	7125	7656	8254	8927	9685
Respiratory hospital admissions	542	576	614	658	706	761	822	890
Cardiovascular hospital admissions	479	509	543	581	624	672	726	787
Emergency room visits	3772	4007	4271	4568	4901	5275	5695	6166
Acute asthma attacks	22937	24390	26031	27880	29964	32311	34952	37924
Acute respiratory symptom days	28503958	30273600	32266328	34507208	37024341	39849233	43017212	46567904
Restricted activity days	5510765	5852896	6238157	6671394	7158039	7704185	8316661	9003128
Economic valuation (Million 2000 U.S. dollars)								
Health benefits of avoided deaths	457.9	485.7	517.1	552.5	592.2	636.8	686.8	743.0
Health benefits of avoided illness	450.0	478.3	510.2	546.1	586.5	632.0	683.1	740.5
Total health benefits	**907.9**	**964.0**	**1027.3**	**1098.6**	**1178.7**	**1268.8**	**1369.9**	**1483.5**

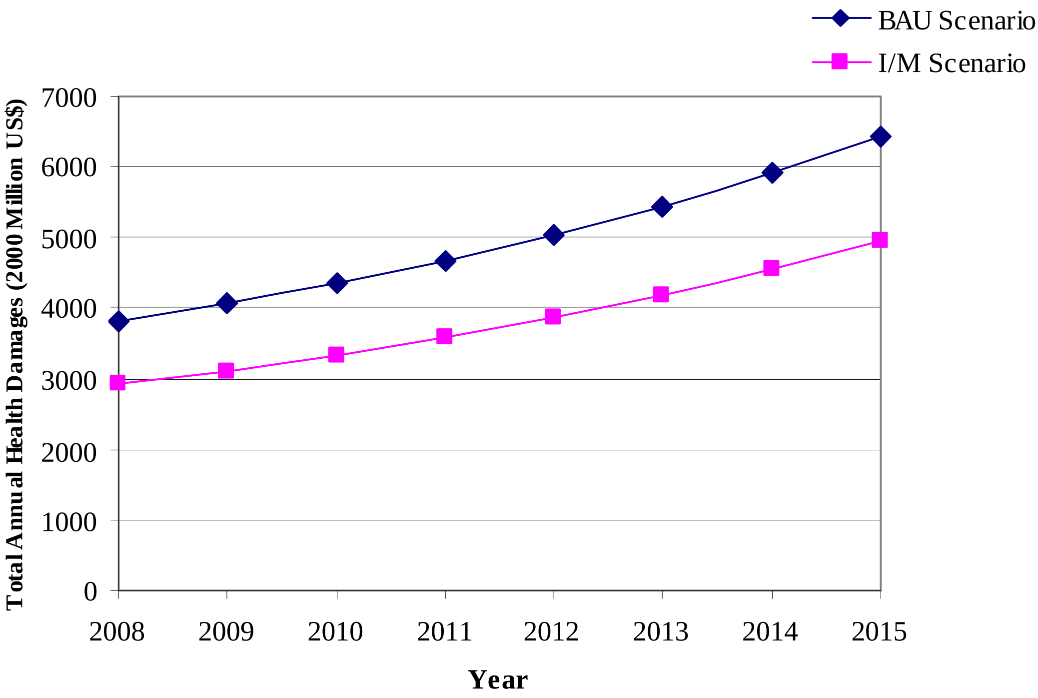

Figure 4.7 Total Annual Health Damages Attributable to PM_{10} Emissions from Motor Vehicles: BAU v.s. I/M (25% Emission Reductions) Scenarios

The results in Table 4.10 indicate that implementing I/M programs targeting all diesel-fueled vehicles and motorcycles in the BMR will produce significant benefits to public health resulting from air pollution mitigation, if the programs can achieve the target emission reductions from motor vehicles. For example, in 2008, a total of 907.9 million 2000 US$ health damages can be avoided in the BMR by the I/M programs. Avoided premature deaths account for approximately half (457.9 million 2000 US$) of the total benefits.

However, as discussed in Chapter 2, the actual effects of I/M programs might be highly uncertain, resulting in much smaller health benefits associated with the programs depending on the levels of emission reductions achieved. Health benefits of I/M programs as a function of PM_{10} emission reductions from the programs are analyzed in next section.

4.2.5 Costs of I/M programs

The overall costs, or social costs of an I/M program include the full resource costs that are paid by all parties affected by the program (National Research Council, 2001). In general, the costs of I/M programs fall into three categories: costs of finding failing vehicles, vehicle repair costs and costs of program administration and oversight (National Research Council, 2001). The following table by National Research Council (2001) summarizes the major cost components of I/M programs.

Table 4.11 Cost Components of I/M Programs

Cost Category	Components of Cost
Cost of finding failing vehicles	Test or inspection cost (e.g., in I/M lane, by remote sensor, or on-board diagnostic readout)
	Motorist cost including travel time and queuing time (for lane inspection)
Vehicle repair costs and associated fuel economy improvements	Resource cost of repair (if done at repair shop)
	Expenditures on parts and value of time (for self-repair)
	Cost of reinspection
	Fuel economy savings
Costs of program administration and oversight	Costs of administering program (aside from direct cost of testing)
	Enforcement costs
	Evaluation costs

Source: National Research Council, 2001.

More practically, the primary cost components of an I/M program consist of fixed costs including program start-up costs, and variable costs including operating and maintenance costs and vehicle emission repair costs[27]. Parsons (2001) estimated the annual costs of the proposed I/M programs targeting diesel-fueled vehicles and motorcycles in the BMR. However, their estimates of costs only included start-up costs (capital and land use costs) and inspection costs (operating and maintenance costs), but excluded problem vehicle repair costs, which can be a substantial portion of the total

[27]In the case of an I/M program, variable costs depend on the number of vehicles that are inspected and repaired. Generally, the more vehicles are inspected or repaired, the larger the variable costs are.

social costs of I/M programs (National Research Council, 2001). In order to derive the total social costs of the I/M programs considered here, the repair costs are estimated and aggregated with the other costs estimated by Parsons (2001).

Currently information about vehicle repair costs as well as effectiveness of repair associated with I/M programs is rather incomplete in the U.S. and elsewhere (National Research Council, 2001), in particular for PM-related I/M programs, which are still in their infancy. A recent study on reducing emissions from motorcycles in Bangkok (World Bank, 2003) reports 60% of motorcycles that fail the emission test will pass the test after a minor tune-up, and the remaining 40% that fail the initial test will need major repairs, remanufacturing to meet current emissions standards, or scrapping. The study also reports the costs of repairs: the average minor repair cost is 30 baht per vehicle ($0.75, based on the exchange rate of 39.79 in 2000); major repair costs vary from 100 baht to 10000 baht, depending on the motorcycle conditions (on average 2204 baht, or $58.3); and engine remanufacturing costs from 5000 to 7000 baht (on average 6000 baht, or $150.8). Since their study was conducted in 2000, it is expected that much less engine remanufacturing is needed to satisfy the emission standards in 2008 and the years afterward. Here only minor and major repairs are considered, as it is assumed that engine remanufacturing has already been accomplished for the large majority of candidate vehicles.

Given the lack of repair costs for all other vehicle types in Thailand, relevant information in the U.S. is examined. A study at Colorado State reports that estimated average repair cost is $465 per failing vehicle for heavy-duty diesel vehicles including heavy trucks and buses (Clean Air Fleets [CAF], 2002). Furthermore, Ando et al (1999) reports that the mean repair costs for light trucks are about $110 per vehicle[28]. In order to apply the U.S. vehicle repair costs to Thailand, the values need to be adjusted to reflect

[28]The value is for traditional I/M programs targeting emissions of CO, HC and NO_x. In lack of information about costs to reduce PM emissions from light-duty trucks, this value is adopted in this study.

the fiscal difference between the two countries. In general, vehicle repair costs include parts and labor costs and it is assumed that each accounts for 50% of the total repair costs. Regarding vehicle parts, the World Bank study reports that a motorcycle spark plug costs 36 baht or approximately $1, whereas its counterpart in the U.S. costs about $2 per sparkplug[29]. Based on this, this study assumes that the costs of vehicle parts in Thailand are half of the costs in the U.S. Second, in comparing the labor costs, it is found that the monthly salary of skilled laborers and technicians in Thailand falls into the range of $215-$286 (Thailand Board of Investment, 2006[30]), whereas the monthly salary of similar working groups in the U.S. falls into the range of $2500-$3000 (U.S. Department of Labor, 2006[31]). Based on this information, this study assumes the labor costs in Thailand are one tenth of the costs in the U.S. Therefore, a Thailand/U.S. ratio of 0.3 $(0.5 \times 0.5 + 0.1 \times 0.5 = 0.3)$ is used to extrapolate the U.S. costs to Thailand.

Empirical evidence in the U.S. indicates that the percentage of failing vehicles ranges from 10% to 25% in traditional I/M programs targeting light-duty vehicles such as passenger cars and light trucks (Parsons, 2001). Given this, an assumption is made that 17.5% of the light duty trucks are problem vehicles and thus need repairs. For motorcycles, this value is 25% based on the findings by World Bank (2003)[32]. Moreover, heavy diesel I/M programs are relatively new (e.g. presently in the U.S., only 15 States have diesel I/M of certain types, Duleep, 2004). Past experience shows that typical failure rate was 12-15% at the beginning, and declined to 5-6% after some years of

[29]Online source: http://www.hectorshardware.biz/shop/product.asp?dept_id=324&sku=577529&, accessed January 20, 2007.

[30]Source: http://www.boi.go.th/english/how/labor_costs.asp, accessed January 20, 2007.

[31]Source: http://www.bls.gov/oes/current/oes_nat.htm#b49-0000, accessed January 20, 2007.

[32]The study found that 65% motorcycles in Bangkok were 5 or less years old, and 20% of them failed the emission test, whereas among the remaining 35% motorcycles that were more than 5 years old, 35% failed the test. Based on this, it is assumed that 25% motorcycles fail the emission test $(0.65 \times 0.2 + 0.35 \times 0.35 = 0.25)$.

implementation (Duleep, 2004). This implies that the percentage of failing heavy diesel vehicles may be lower than that of failing light diesel vehicles. Based on this, this study makes an assumption that 10% of the buses and heavy trucks in the BMR are problematic.

The following table summarizes the estimated costs of I/M programs used in this study.

Table 4.12 Social Costs of Inspection and Maintenance Programs in the Bangkok Metropolitan Region

Vehicle Type	Annual Testing and Administrative Cost (Million 2000 US$) (1)	Repair Costs					Total Annual Cost (Million 2000 US$)
		Average Repair Cost per Vehicle in the U.S. (2000 US$)	Average Repair Cost per Vehicle in Thailand (2000 US$)	Percent of Repairs	Total Number of Repairs	Total Annual Repair Cost (Million 2000 US$)	
City Bus	4.995	465 (2)	140	10%	4766	0.380	5.37
City Truck	3.19	465 (2)	140	10%	15713	1.253	4.44
Long Haul Truck/Bus	7.53	465 (2)	140	10%	7434	0.593	1.35
Light Duty Truck	65.24	110 (3)	33	17.5%	435861	14.383	79.62
Motorcycle	32.93	Minor repair (4)	0.75	15%	418329	0.314	49.50
		Major repair (4)	58.3	10%	278886	16.259	
Total	**113.89**					**33.18**	**147.07**

Sources: (1) Parsons, 2001; (2) CAF, 2002; (3) Ando et al, 1999; (4) World Bank, 2003.

4.2.6. The net benefits of the PM-related I/M programs in the BMR

Cost-Benefit Analysis can be used to determine whether the investment of human and capital resources required for an I/M program are beneficial and to make a decision about whether to launch an I/M program or improving any existing I/M program designs. In general, it is expected that a program would produce at least a positive net benefit in order to ensure that social resources are not poorly spent. To examine the relative magnitude of benefits and costs of the I/M programs considered, both the total annual benefits and the total annual costs are estimated and then compared, using the year 2008 as an example. Here the only category of benefit included is heath benefit, the primary benefits of air pollution control. As mentioned before, the health benefits of the programs might vary considerably depending on the actual emission reduction accomplished. Table 4.13 summarizes the total annual health benefits of the I/M programs in the BMR associated with different levels of PM_{10} emission reductions resulting from the programs.

On the other hand, compared to the benefits, the total costs of the I/M programs may vary to a smaller extent as the level of emission reduction changes. In particular, besides the fixed costs, studies have found that there is little correlation between repair costs and emissions reduced per vehicle (National Research Council, 2001). However, it is expected that the total costs of I/M affect the aggregate level of emission of reductions available from the programs to a certain extent. For instance, a common way to find more gross polluters is to require more vehicles to be inspected. As a result, the total inspection costs will be increased. While an in-depth analysis of the relationship between I/M costs and emission reductions is beyond the scope of this study, the goal here is to provide reasonable estimates of the net benefits of the programs. The total annual cost estimation provided in Table 4.12 is considered to fall into the upper range of the program’s potential cost, since costs were estimated (by Parsons, 2001) based on the assumption that the I/M programs will be fully successful, i.e. achieve 25% overall PM_{10} emission reductions from motor vehicles. The program's total cost may decrease as the aggregate level of emission reductions decline. In order to provide conservative estimates of the net benefits, the total cost estimate in Table 4.12 is applied regardless of the overall level of emission reduction from vehicles. By subtracting the costs from the benefits, the annual net benefits of the I/M programs are illustrated in Figure 4.8.

Table 4.13 Annual Health Benefits as a Function of Percentage Particulate Matter Reduction from Motor Vehicles in the BMR (Year: 2008)

% PM_{10} Emission Reduction from Motor Vehicles	Annual Health Benefits (Million 2000$)	% PM_{10} Emission Reduction from Motor Vehicles	Annual Health Benefits (Million 2000$)	% PM_{10} Emission Reduction from Motor Vehicles	Annual Health Benefits (Million 2000$)
0%	0	***9%***	323.40	***18%***	650.64
1%	35.75	***10%***	359.57	***19%***	687.24
2%	71.54	***11%***	395.78	***20%***	723.89
3%	107.38	***12%***	432.05	***21%***	760.59
4%	143.26	***13%***	468.36	***22%***	797.33
5%	179.20	***14%***	504.72	***23%***	834.13
6%	215.18	***15%***	541.12	***24%***	870.98
7%	251.20	***16%***	577.58	***25%***	907.87
8%	287.28	***17%***	614.08	***26%***	944.81
Total Annual Costs		**147.1** (Million 2000$)			

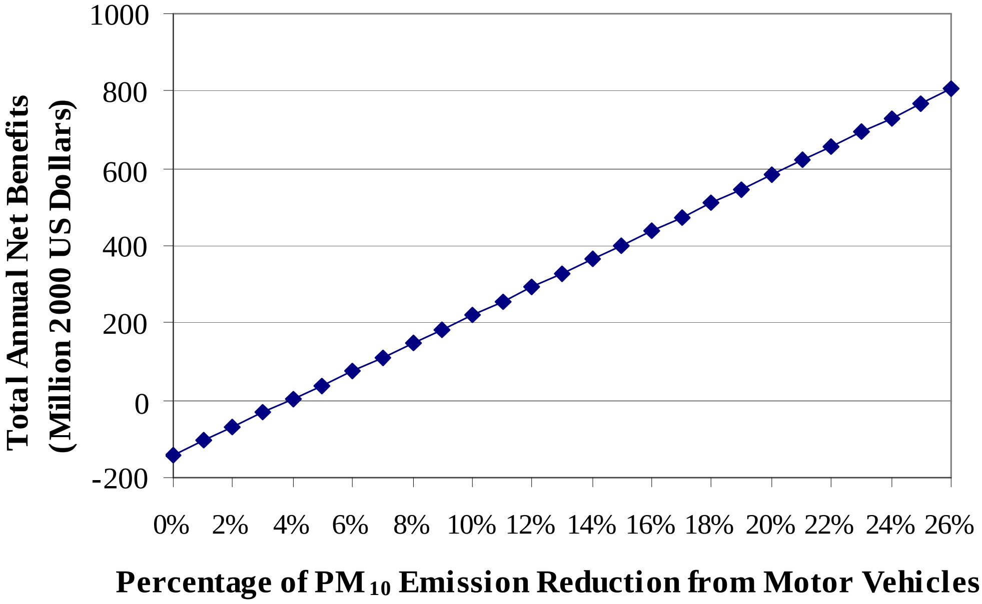

Figure 4.8 Total Annual Net Benefits of the I/M Programs as Functions of the Percent of Overall PM_{10} Emission Reductions from Vehicles (Year: 2008)

Figure 4.8 shows that the total net benefits are approximately a linear function of the percentage of PM_{10} emission reduction from motor vehicles. And a minimum of about 4% aggregate level of PM_{10} emission reduction is required to ensure the net benefits are positive, i.e. the total benefits of the programs considered outweigh the total costs. In the next Chapter, the minimum requirements for the key variables in I/M program designs, such as the fraction of problem vehicles identified and the fraction of effective repairs, in order to reach the 4% emissions reduction goal, are analyzed.

4.2.7 Health benefits of vehicle retrofitting and repowering programs

4.2.7.1 Installing oxidation catalyst converters and LPG-diesel bi-fuel fueling systems on heavy-duty diesel vehicles

As discussed in Section 2.3.2.2 of Chapter 3, a feasible as well as cost-effective policy package that will deliver significant mobile sources particulate emission reductions in the BMR is to combine I/M programs, installing oxidation catalyst converters on heavy-duty diesel vehicles (heavy trucks and buses), and installing LPG and diesel bi-fuel fueling systems on vehicles equipped with oxidation catalytic converters. Assuming the I/M programs also target a 25% overall emissions reduction, and are completely successful. Installing oxidation catalytic converters to all city buses and trucks will deliver a 35% emissions reduction from the retrofitted vehicles (city buses and trucks) on the basis of the emission level with I/M, and converting the city buses and trucks to LPG-diesel bi-fuel operation will further deliver a 50% emission reductions from these vehicles.

The following table summarizes the three control measures under this scenario and potential levels of emission reductions achieved by each specific measure.

Table 4.14 Recommended Policy Measures to Control Vehicle Particulate Matter Emissions in the BMR and Level of Emissions Reduction Compared to the Baseline

***Policy Scenario*: I/M + Installing oxidation catalyst converters + Installing LPG and diesel bi-fuel systems**	
Control Measures	Estimated Percentage of PM_{10} Emission Reduction Relative to the Baseline
I/M program targeting all diesel fueled vehicles and motorcycles (assuming completely successful)	25%
Installing oxidation catalytic converters to all city-based buses and trucks	35%
Converting all city-based buses and trucks to LPG-diesel bi-fuel operation (typically in the ratio of 80% diesel to 20% gaseous fuel)	50%

Source: Parsons, 2001

The new fleet-average vehicle emission rates and total emissions from each vehicle type are summarized in Table 4.15 (assuming the same vehicle growth rates as under the I/M scenario).

Table 4.15 Vehicle Particulate Matter Emissions in the Bangkok Metropolitan Region under the I/M-Oxidation Catalytic Converters-LPG-Diesel Bi-fuel Scenario (Year: 2008)

Vehicle Type	Number of Vehicles	Average Annual Mileage (km)	Control Measures	PM_{10} Emissions Reduction Level (%)	PM_{10} Emission Rate (g/km)	Annual PM_{10} Emissions (tons/year)	Share of Total PM_{10} Emissions (%)
City Bus	27236	97525	I/M, Converters, Bi-fuel	74 (1)	0.320	850	6.67
City Truck	89789	16000	I/M, Converters, Bi-fuel	76 (2)	0.300	431	3.38
Long Haul Truck/Bus	42481	12000	I/M	25	0.923	471	3.69
Light Duty Truck	2490636	18075	I/M	25	0.198	8915	70.00
Passenger Car	2225149	17171	N/A	0	0.003	127	1.00
Motorcycle	2788860	10000	I/M	30	0.070	1943	15.25
			Total	**38**		**12736**	**100**

Note:

(1) $1-(1-20\%)\times(1-35\%)\times(1-50\%)=74\%$

(2) $1-(1-25\%)\times(1-35\%)\times(1-50\%)=76\%$

Therefore, there is an approximate 38% PM_{10} emission reduction relative to the baseline scenario (BAU), which is estimated to be 20646 tons in 2008 (Table 4.5). Using this result, the annual health benefits under this scenario is estimated and summarized in the following table along with the estimates of costs by Parsons (2001).

Table 4.16 Health Benefits and Costs under the I/M-Oxidation Catalytic Converters-LPG-Diesel Bi-fuel Policy Scenario (Year: 2008)

***Policy Scenario*: I/M + Installing oxidation catalyst converters + Installing LPG and diesel bi-fuel systems**				
Control Measures (1)	Total % of PM_{10} Emission Reduction from Motor Vehicles	Total Annual Health Benefits	Costs of Measures	Total Costs
		(Million 2000 U.S. dollars)		
I/M programs	38%	**1392.1**	147.1	**199.7**
Installing oxidation catalytic converters			26.3 (2)	
Converting all city-based buses and trucks to LPG-diesel bi-fuel operation			26.3 (2)	

Note:

(1) Refer to Table 4.13 for more details of these control measures;

(2) Source: Parsons, 2001.

4.2.7.2 Particulate traps and gaseous-fuel fueled vehicles

Installing continuously generating particulate traps and converting to gaseous-fuel fueled vehicles are currently not feasible in the BMR but may be considered by the government in the future, given the factor that they can deliver PM emission reductions from vehicles up to 90%. In this section, three hypothetical policy scenarios are considered: (A) Installing particulate traps on all heavy-duty diesel vehicles (all buses and heavy trucks); (B) Converting heavy-duty diesel vehicles to CNG/LPG fuel operation; (C) Converting all diesel vehicles to CNG/LPG fuel operation (all buses, heavy and light trucks). Moreover, under all these policy scenarios, I/M programs are necessary to achieve the expected emission reductions by vehicle retrofitting. Evidence indicates that most retrofitted emission control systems are only durable for a short time (e.g. two years or less), and without appropriate maintenance, many of retrofitted vehicles become gross-polluters again soon (Hao et al, 2006).

The new fleet-average vehicle emission rates and total emissions from each vehicle type under the three policy scenarios are summarized in the following table (again, assuming the same vehicle growth rates as under the I/M scenario).

Table 4.17 PM Emissions Reduction from Motor Vehicles Relative to the Baseline under Three Policy Scenarios Related to Installing Particulate Traps or Converting to Gaseous-Fuel (Year: 2008)

Vehicle Type	Possible Control Measures	Policy Scenarios (1)	PM_{10} Emissions Reduction Level (%)	PM_{10} Emission Rate (g/km)	Annual PM_{10} Emissions (tons/year)	Share of Total PM_{10} Emissions (%)
City Bus	I/M, particulate traps, CNG/LPG fuel	A, B	90 (2)	0.123	327	2.83
		C				8.55
City Truck	I/M, particulate traps, CNG/LPG fuel	A, B	90 (2)	0.123	177	1.53
		C				4.62
Long Haul Truck/Bus	I/M, particulate traps, CNG/LPG fuel	A, B	90 (2)	0.123	63	0.54
		C				1.64
Light Duty Truck	I/M, CNG/LPG fuel	A, B	25	0.198	8915	77.18
		C	90	0.026	1189	31.08
Passenger Car	N/A	A, B	0	0.003	127	1.10
		C				3.31
Motorcycle	I/M	A, B	30	0.07	1943	16.82
		C				50.80
Total		A, B	**44**		**11551**	**100**
		C	**81**		**3825**	

Note:

(1) Policy scenarios:

A: Installing particulate traps on all heavy-duty diesel vehicles (all buses and heavy trucks), enforcing I/M targeting all diesel vehicles and motorcycles.
B: Converting heavy-duty diesel vehicles to CNG/LPG fuel operation, enforcing I/M targeting all diesel vehicles and motorcycles.
C: Converting all diesel vehicles (all buses, heavy and light trucks) to CNG/LPG fuel operation, enforcing I/M targeting all diesel vehicles and motorcycles.

(2) When particulate traps or CNG/LPG fuel are used in combination with I/M, their combined effect in term of emissions reduction is considered to be 90%. No separate effect by individual measures is considered here.

Table 4.18 Health Benefits and Costs under Three Policy Scenarios Related to Installing Particulate Traps or Converting to Gaseous-Fuel (Year: 2008)

Policy Scenario*	Policy Measured Included	Overall PM_{10} Emissions Reduction Level Relative to the Baseline (%)	Total Annual Health Benefits	Cost of I/M Programs	Costs of Retrofitting (Particulate Traps or CNG/LPG Fuel)	Total Costs
			(Million 2000 US$)			
A	I/M and particulate traps	44	**1618.4**	147.1	109.4	**256.5**
B	I/M and CNG/LPG fuel	44	**1618.4**		145.9	**293.0**
C	I/M and CNG/LPG fuel	81	**3057**		381.7	**528.8**

**Note*: Policy scenarios A, B and C are defined the same as in Table 4.16.

Table 4.18 indicates that installing particulate traps or converting diesel engines to be fueled by gaseous fuels (CNG/LPG) has the potential to achieve significant public health improvement due to the fact that they appreciably reduce particulate matter emissions compared to traditional diesel engines. Comparing policy scenarios A and B shows that emission reductions and consequent health benefits achieved by installing particulate traps or converting to CNG/LPG fuel are very similar, if they both target only heavy-duty diesel vehicles, whereas the total costs of the CNG/LPG scenario (Scenario B) are higher.

As discussed in Chapter 2, Section 2.3.2.2, presently the primary barrier to wide adoption of particulate traps in developing countries is the need for fuels with ultra-low sulfur content, which requires some fundamental change in the fuel industry and their products. Moreover, most recently, concerns have arise over CNG engines due to the factors that GNG combustion produces emissions that include toxic pollutants such as polycyclic aromatic hydrocarbons (PAHs) and benzene, which also pose great health risk for the public (Kado et al, 2005). As scientific research on this topic becomes more conclusive, government needs to consider the overall risk of different control measures in making pollution control decisions.

Chapter 5 Evaluating the Effectiveness of Vehicle Inspection and Maintenance Programs

Vehicle inspection and maintenance programs are currently considered as a core policy tool to control emissions from in-service vehicles. Chapter 4 focused on estimating the potential health benefits associated with the relatively new PM-oriented I/M programs. Health benefits as a function of different levels of emission reductions were analyzed and compared with the social costs of the programs (these different levels of reductions were considered due to the significant uncertainty involved in the actual emission reduction benefits of I/M programs). It was found that a minimum of about 4% reduction of the total PM_{10} emissions from motor vehicles is required in order for the total benefits to be greater than the total costs of implementing the programs. The main purpose of the current chapter is to examine how key variables affect PM_{10} emission reductions available from the I/M programs, and the desirable performance of these variables in order to achieve the 4% emission reduction objective. These analyses are conducted using an analysis approach called ‘I/M Design’ developed by Eisinger (2005) (this analysis approach is incorporated into a spreadsheet tool that relates the important program features (variables) to I/M emission reductions), and also using the best available information about I/M experience in Thailand and elsewhere as the inputs to the spreadsheet. Examples of the important variables in I/M design include the percentage of problem vehicles identified by the inspection process (identification rate), and percentage of identified vehicles that are effectively repaired and pass I/M tests afterwards, etc.

5.1 An overview of PM-related inspection and maintenance programs

Section 3.5 in Chapter 3 contained a discussion of the components of emissions reduction from an I/M program. Traditional I/M programs primarily address CO, HC and NO_x emissions from light-duty gasoline vehicles, given that these are the key air pollutants in urban areas (HC and NO_x are the precursors of ground-level ozone). Recently there has been growing interest worldwide in introducing equivalent programs for diesel vehicles targeting PM emissions from these vehicles due to the increasing concerns of the health hazards associated with PM air pollution. While the design and implementation of the two types of I/M programs is identical or similar, at present the main difference is their testing procedure. HC, CO and NO_x emissions are measured directly in inspections, whereas smoke opacity tests are usually used as the indicator of PM emissions. Since high opacity may indicate engine malfunction and increased emissions of air pollutants, primarily unburned fuel hydrocarbons (emitted as an aerosol) or soot particles, repair of high opacity emitters may result in a decrease in the contribution of diesel vehicles to the pollutant inventory (Mccormick et al, 2003). Opacity tests are commonly used in diesel vehicle inspection programs for control of PM emissions because these tests are relatively cheap and easy to conduct on a roadside, whereas a direct measure of PM emissions can be very costly. However, recently there have been rising concerns over the actual emission reduction benefits of repairs based on smoke opacity testing. For example, McCormick et al (2003) performed both opacity tests and PM emission measures on 26 heavy-duty diesel vehicles and found that smoke opacity is a poor predictor of PM emissions with an r^2 of about 0.2. They also reported that several vehicles with relatively high PM emissions exhibited low smoke opacity, indicating that smoke opacity measurements may fail to indentify all high emitters. The emission reduction benefits of I/M may be appreciably harmed if the testing procedure fails to measure emissions accurately. Some studies have suggested that PM emissions should be measured directly rather than using opacity tests (e.g. Parsons, 2001). While the research on direct PM emissions testing equipment (e.g. PM meters, Parsons, 2001) is still ongoing, the analysis in this chapter is based on the assumption that PM emissions can be measured directly and reliably in the enhanced PM-related I/M programs in the BMR. Therefore, no relationship between PM emission rates and smoke opacity is considered.

5.2 A framework to estimate the effectiveness of I/M programs

Empirical evidence on the performance of I/M programs and on the important elements affecting I/M emission reductions is fairly limited. In the U.S., only two states – California and Arizona - have systematically collected and released empirical data on traditional light-duty gasoline vehicle I/M programs in their states (Eisinger, 2005; Ando et al, 1999). Evidence on the performance of PM-oriented I/M is even less available given that these programs are still relatively new. In Thailand, the data collected by a World Bank study (World Bank, 2003) on their pilot motorcycle inspection and upgrade project in Bangkok are the most comprehensive dataset on I/M programs in the BMR. Very little information is available on the performance of I/M targeting diesel-fuel vehicles including buses and trucks. Given the limitation of data, in running the spreadsheet developed by Eisinger (2005), the values of most variables in the tool are derived based on the best available information in the U.S. and some extrapolation is performed to the BMR. An uncertainty analysis is then conducted (in Chapter 6) to characterize the overall uncertainty due to this need to extrapolate. Definitions of the variables in ‘I/M Design’ and their input values used in this study are discussed below.

Table 5.1 Variables Governing I/M Program Emission Reductions

(All variables in this table were developed and defined in Eisinger, 2005 unless noted.)

Variable Name and Definition	Variable Explanation	Values Used in Eisinger (2005)[33]	Values Used in This Study (Range, PDF)	Discussion
PartiRate: % of all vehicles required by I/M participate in the programs	Although I/M programs require all vehicles regulated by the programs to take the inspection process, there may be a certain fraction of vehicles operating illegally without participating in I/M. This variable is not in the 'I/M Design' spreadsheet[34], but is developed by this study to reflect the levels of participation in the programs in the study area.	100%	90% (80-100%, triangular[35])	It has been proposed that the new PM-related I/M programs should be linked to vehicle registrations and managed by a central database in order to significantly improve the levels of participation in the programs. It is expected that, with the government's strong will and efforts to curb severe air pollution in the BMR, the participation rate of the programs can be high. A mean estimate of 90% participation rate is assumed in this study. However, sensitivity analysis will test the role of this variable on the overall emission reductions by I/M. For simplicity, it is assumed that the fraction of problem vehicles is the same in the participation group as in the 'non-participation group', although in reality, problem vehicles are more likely to escape from the inspection process.

[33]The input values used in the paper are based on light-duty vehicle hydrocarbon (HC) inspection data from an enhanced I/M program in southern California's South Coast Air Basin.

[34]The original study examines the emission reduction benefits by an I/M program in a previous year using data on the actual number of vehicles inspected. It assumed that all vehicles subject to inspection participated in the program.

[35]Triangular distribution is selected for all the variables in Table 5.1 except for VehWaive. There is little empirical evidence to support the PDFs of the variables in Table 5.1. Therefore, the PDFs were selected based on the author's own judgment. When there is more confidence in values near the central value than in values far away on either side, the triangular distribution is selected. And in the case of the variable VehWaive, there is no reason to believe that some values between the lower and upper limits are larger than the others, and therefore, the uniform distribution is selected. Furthermore, the range was obtained based on the mean and the known theoretical limit of the variable, namely, 0% for the lower limit or 100% for the upper limit.

Table 5.1 Variables Governing I/M Program Emission Reductions (Continued)

Variable Name and Definition	Variable Explanation	Values Used in Eisinger (2005)	Values Used in This Study (Range, PDF)	Discussion
<u>IndenRate</u>: % of inspected problem vehicles that are identified by I/M	Although I/M programs aim at identifying all the problem vehicles (defined as vehicles whose emission rates exceeds I/M testing cut-points) that are inspected, the inherent limitations of I/M make a 100% identification rate unrealistic. And it is accepted that some problem vehicles, e.g. 10% of all problem vehicles, will falsely pass I/M.	Upper: 90% Lower: 71%	50% (0-100%, triangular)	This variable reflects the ability of I/M programs to identify problem vehicles. Given that at present the test protocol and program implementation for PM-related I/M programs is not as well developed as that for traditional I/M, and both may be less well developed in developing countries, the identification rates associated with these I/M programs are expected to be lower. Based on this, a 50% identification rate is assumed in this study. In uncertainty analysis, the range of this variable is set to be 0-100%, reflecting the worst case that none of the problem vehicles are identified and the ideal case that all problem vehicles are identified.
<u>ScrapFrac</u>: % of failed vehicles that are scrapped	Early scrappage of problem vehicles results in emissions reduction, if the replacement vehicles generate less emission. For simplification, I/M Design does not account for replacement vehicle deterioration, and all replacement vehicles are assumed to pass I/M two years following their purchase.	Upper: 13.3% Lower: 6.7%	5% (0-10%) for motorcycles, 2.5% (0-5%) for light duty trucks, 0.5% (0-1%) for buses and heavy trucks (Triangular distribution)	World Bank (2003) predicts that 5% of failed motorcycles will be scrapped. This result indicates that the early scrappage rates attributable to I/M programs may be lower in developing countries than in developed countries. 5% is applied to motorcycles in this study, and a 2.5% scrappage rate is assumed for light-duty trucks and 0.5% for buses and heavy trucks, given that these vehicles are generally more expensive and thus less likely to be scrapped.

Table 5.1 Variables Governing I/M Program Emission Reductions (Continued)

Variable Name and Definition	Variable Explanation	Values Used in Eisinger (2005)	Values Used in This Study (Range, PDF)	Discussion
<u>VehWaive</u>: % of identified problem vehicles waived by an I/M program	I/M design in the U.S. generally allows some fraction of problem vehicles to be waived from the programs, usually because of economic hardship. For example, the diesel I/M guideline in Salt Lake Valley states that a vehicle that continues to exceed applicable opacity standards after $750 of acceptable emissions related repairs may apply for a Certificate of Waiver (Salt Lake Valley Health Department, 2006).	Upper: 1% Lower: 4%	1% (0-2%, uniform)	In Bangkok, the government may also consider waivers in the implementation of I/M programs. In particular, for public transit such as buses, high repair costs are likely to result in significant increases in bus fares, which may prevent low income people from using them. However, given the severity of the air pollution problem in the area, a high waiver rate should be restricted. A 1% waiver rate is assumed for all vehicle types.
<u>IllegalVeh</u>: % of identified problem vehicles operate illegally	There may be some fraction of vehicles operating illegally without undergoing the requisite repairs or certifications needed to pass or be waived from the I/M inspection process.	Upper: 6.6% Lower: 13%	Motorcycles and light trucks: 20% (0-40%); buses and heavy trucks: 10% (0-20%) (Triangular distribution)	The illegal operating rates may be higher in the BMR since the I/M programs are less mature. It is assumed that for motorcycles and light trucks, the rates are both 20%, and for public transits and heavy trucks, the rates are 10% since it should be easier to identify the violations by these vehicles on road.

Table 5.1 Variables Governing I/M Program Emission Reductions (Continued)

Variable Name and Definition	Variable Explanation	Values Used in Eisinger (2005)	Values Used in This Study (Range, PDF)	Discussion
GoodRep: % of repair work initially effective	Some fraction of repairs are not effective but falsely pass re-tests. For example, random roadside tests show that a portion of the vehicle fleet fails I/M immediately after being repaired but then pass an official I/M test.	Upper and Lower: 80%	72% (44-100%, triangular)	Available information related to this variable is very limited. A study by Land Transport Department of Thailand randomly selected 21 private inspection centers in Bangkok and requested two problem motorcycles to be tested by these inspection stations. The two motorcycles were failed by 12 of the 21 stations whereas passed by the remaining 9 stations. This study indicates that only 58% $(1-9\div21)$ of testing vehicles may properly pass the I/M. However, this study was conducted on the existing I/M programs in the BMR, which is considered as ineffective (refer to Section 2.3.2.1 in Chapter 2). It is assumed that the updated I/M in the BMR considered here will improve the performance of this variable and achieve 90% of the U.S. level. Therefore, the value of this variable is: $90\% \times 80\% = 72\%$.
ExEm: % of excess emissions (emissions above allowable levels) from identified problem vehicle reduced by good repairs (repairs that properly pass an I/M test immediately)	Effective repairs motivated by I/M do not address all excess emissions. For example, I/M tests do not address emissions from cold starts, since vehicles are tested after the engine and catalyst are warm. U.S.EPA estimates that a model IM240 program identifies 92% of HC, 68% of CO, and 83% of NO_x excess emissions.	Upper: 92% Lower: 81%	81% (62-100%, triangular)	This variable is highly uncertain for I/M programs targeting PM without further research. In lack of further information, the rates of the three pollutants HC (92%), CO (68%), and NO_x (83%) are averaged (equal to 81%) and used for PM. The range is 62-100%.

Table 5.1 Variables Governing I/M Program Emission Reductions (Continued)

Variable Name and Definition	Variable Explanation	Values Used in Eisinger (2005)	Values Used in This Study (Range, PDF)	Discussion
<u>DurRep</u>: % of good repairs that remain durable	Some of the good repairs may deteriorate fast and not be durable enough to pass another I/M test after one or two years (depending on the frequency of testing required). Therefore, they will generate excess emissions in between two tests.	Upper: 94% Lower: 79%	86.5% (73-100%, triangular)	Diesel vehicles may deteriorate rapidly without proper maintenance. On the contrary, a well-maintained diesel vehicle will generally retain a good emissions performance throughout its operating life (Parsons, 2001). It is expected that the updated I/M programs in the BMR should be able to motivate vehicle owners to better maintain their vehicles in anticipation of the effective inspection process. In lack of more available information, the U.S. values are used in this study.

In addition to the input variables listed in Table 5.1, the spreadsheet also needs the following inputs related to the characteristics of the vehicle population studied: (1) Problem vehicles as percent of total vehicles: The values used in the spreadsheet are consistent with the assumptions made in Section 4.2.5 of Chapter 4 -- 10% of buses and heavy trucks, 17.5% of light trucks and 25% of motorcycles in the BMR are problem vehicles. In the uncertainty analysis (in Chapter 6), the upper and lower bounds of this parameter are assumed to be 1.5 times and half of the mean estimate, respectively. And in lack of empirical evidence to support the form of PDF of this parameter, the uniform distribution was selected based on the authors own judgment; (2) Problem vehicles as percent of total PM emissions: as discussed in Section 2.3.2.1 of Chapter 2, studies usually suggest that the gross-polluting vehicle pool is responsible for a substantial fraction -- variously estimated at 50% to 80% -- of total vehicle emissions (Harrington, 1997; Beaton et al, 1995). Based on this, this study assumes that 50% and 80% are the lower and upper limits, respectively, and the mean value of them, 65%, is the best estimate of total vehicular emissions are generated by problem vehicles. Also, the uniform distribution was selected in the uncertainty analysis based on the author's own judgment; (3) Number of vehicles (under both the baseline and the I/M scenarios), average annual VKT per vehicle and baseline fleet-average emission rates (in units of g/km-vehicle).

The testing cut-points for each type of vehicle need to be determined and input into the spreadsheet. Emission cut-points are established in I/M programs to identify the worst polluters and minimize false failures (Eisinger, 2005). In reality, vehicle emission rates usually span a wide spectrum. Conceptually, if an I/M targets a 25% reduction of the total emissions from motor vehicles, a cut-point equal to 75% of the current fleet-average emission rate will reduce the emission rates of all vehicles to 75% of the current level or lower, and thus ensure that the 25% emission reduction target is reached with confidence.

However, given that the emission rates of the large portion of "good" vehicles (e.g. 90% of total vehicles) are usually much lower than the small portion of problems vehicles (e.g. 10% of total vehicles), it is not necessary, or probably not feasible either, to cut the emission rates of all vehicles to 75% of the current average level or lower in order to achieve the 25% reduction goal[36]. More stringent cut-points may be able to fail more vehicles, in particular those with emission rates close to the failure cut-points. However, more stringent cut-points are also likely to increase the social costs of I/M programs, and to suffer from problems such as technological infeasibility and motorist acceptance of the programs. While how to select and modify testing cut-points in I/M design to optimize the program effectiveness is beyond the scope of this study, this study uses the 'ideal' cut-points discussed above, i.e. cut-points equal to 75% of the baseline fleet-average emission rates for each vehicle type in the BMR for the 'best estimate' case, followed by an examination of the impacts of alternative cut-points on overall emission reduction levels. Just for comparison, the enhanced I/M in southern California's South Coast Air Basin studied in Eisinger (2005) used a rate of 86% of the baseline the fleet-average emission rate as the failure cut-point (the baseline rate was 1.25g/mi and the cut-point was 1.08g/mi).

Eisinger (2005) considers that a small fraction (in the range of 0-7.5%) of the initial problem vehicles seek repairs in anticipation of I/M tests, and they are assumed to be 'good' vehicles in inspection and pass the I/M test. Although the emission reductions resulting from this kind of 'pre-test' repairs are taken into account in Eisinger (2005), these reductions are only responsible for a small fraction of total emission reduction

[36]A hypothetical example is provided here: Assuming that a vehicle fleet has an average emission rate of 1.0g/km. 10% of all vehicles are gross polluting and they are responsible for 50% of the total emissions. Based on this information, it can be derived that the average emission rates for good and problem vehicles are 0.56g/km and 5.0g/km, respectively. As long as the average emission rate for problem vehicles goes down to 2.46g/km, the fleet-average rate will decrease to 0.75g/km. Therefore, if all problem vehicles can be properly identified and fixed, a cut-point of 2.46g/km will ensure the 25% emission reduction goal accomplished.

benefits achieved by I/M programs, approximately ranging from 0-2%. It is expected that the fraction of problem vehicles seeking emission repairs before I/M are even smaller in a developing country than that in the U.S., given that people are generally less wealthy and less able to afford the costs of maintenance and repairs. For simplicity, this study does not consider the emission reductions resulting from pre-test repairs.

In addition, traditional gasoline I/M programs usually divide a vehicle fleet by vehicle ages in selecting testing cut-points. For example, for light-duty vehicle HC I/M programs, the U.S.EPA recommends three cut-points for three vehicle age groups -- 6.02g/mi for the 1966-1976 group; 1.16g/mi for the 1977-1992 group; and 0.80g/mi for the 1993-1999 group (Eisinger, 2005). These recommendations reflect the fact that modern vehicles are equipped with better emission control systems and thus generally generate lower emissions. For the relatively new heavy duty diesel I/M programs targeting PM emissions, the EPA recommends that states employ opacity cut-points of 40% for vehicles 1991 and newer and 55% for vehicles 1990 and older (EPA, 1999b). At present, Thailand employs only one standard for each type of in-use diesel vehicle as well as motorcycles[37]. Since it is not clear whether in the future the government may consider dividing vehicle fleets into age groupings in enforcing in-use vehicle emission standards, this study uses one emission standard (or cut-point) for each type of vehicle in running the 'I/M Design' spreadsheet for the BMR.

Equations (5-1)-(5-6) are the mathematical relationships used in 'I/M Design' (Equation (5-1) is developed by this study, and Equations (5-2)-(5-6) are developed by Eisinger, 2005). Detailed descriptions of these equations are placed in Appendix I.

$$\mathrm{ProbVeh} = \mathrm{PartiRate} \times \mathrm{IndenRate} \tag{5-1}$$

[37]Source: Pollution Control Department of Thailand, http://www.pcd.go.th/Info_serv/en_reg_std_airsnd02.html#s4 For diesel vehicles, the opacity standard is 45% (measured while parking the car at load by quick acceleration the engine to maximum rpm); and for motorcycles, the opacity standard is 30%.

$$\text{PercentRep} = \text{ProbVeh} \times [(1 - \text{ScrapFrac}) \times (1 - \text{VehWaive}) \times (1 - \text{IllegalVeh})] \quad (5\text{-}2)$$

$$\text{PercentRed} = \text{GoodRep} \times \text{ExEm} \times \text{DurRep} \times \text{EmisFrac} \quad (5\text{-}3)$$

$$\text{BenefitsRep} = \text{PercentRep} \times \text{PercentRed} \quad (5\text{-}4)$$

$$\text{BenefitsScrap} = \text{ProbVeh} \times \text{ScrapFrac} \times \text{ScrapEmis} \quad (5\text{-}5)$$

$$\text{BenefitsTotal} = \text{BenefitsGrow} + \text{BenefitsRep} + \text{BenefitsScrap} \quad (5\text{-}6)$$

Where, (variables not yet described in Table 5.1):

ProbVeh: Percent of all problem vehicles that are identified by I/M

PercentRep: Percent of all problem vehicles that are failed (identified) by I/M and subsequently repaired

PercentRed: Percent of total vehicle emission reductions achieved by repairs, for the vehicles failing I/M and getting repaired (does not include vehicles that fail I/M and are scrapped, waived, or illegally operating)

EmisFrac: Percent of total emissions represented by excess emissions

BenefitsRep: The benefits of repair work after failing I/M test

ScrapEmis: Percent of total vehicle emissions reduced, for each vehicle retired from the fleet, after accounting for replacement vehicle emissions (assuming the emission rates of replacement vehicles equal the corresponding failure cut-points)

BenefitsGrow: Percent of total emission reductions achieved due to the changes in vehicle population growth as a result of I/M enforcement (this variable is not in the original 'I/M Design' spreadsheet, but is developed by this study to reflect future I/M effectiveness as vehicle fleets change over time)

BenefitsTotal: Total program benefits, in terms of the percent emission reduction in total vehicle emissions from the I/M program

5.3 Estimating emission reduction effectiveness of the PM-related I/M programs in the BMR

Using the 'best estimate' values of the variables discussed in Table 5.1 and the projected 2008 vehicle population, emission rates and VKT, the 'I/M Design' spreadsheet was run for the year 2008. The results show that the PM-oriented I/M programs are expected to reduce total PM_{10} emissions from motor vehicles in the BMR by 10.6%. Table 5.2 summarizes the findings.

Table 5.2 Estimated PM_{10} Emission Reduction Benefits of PM-Related I/M Programs in the Bangkok Metropolitan Region (Year: 2008)

Variables \ Vehicle Type	City Bus	City Truck	Long Haul Truck/Bus	Light Duty Truck	Motorcycle	**Total**
Percent of all problem vehicles indentified by I/M (ProbVeh)	45%	45%	45%	45%	45%	
Percent of problem vehicles failed and repaired (PercentRep)	40%	40%	40%	35%	34%	
Percent reductions from all problem vehicles, achieved by post-test repairs (PercentRed)	45%	45%	45%	40%	36%	
Percent reductions due to the decrease in vehicle growth (BenefitsGrow)	0.1%	0.3%	0.3%	1.4%	0.6%	
Benefits of post-test repair wok (BenefitsRep)	17.8%	17.8%	17.8%	13.9%	12.1%	
Percent reductions from all problem vehicles due to scrap (BenefitsScrap)	0.2%	0.2%	0.2%	0.9%	1.6%	
Total reductions, as percent of total emissions from all problem vehicles (BenefitsTotal)	18.1%	18.3%	18.3%	16.2%	14.3%	
Total reductions achieved by each type of vehicles (in tonnes per year)	385	210	75	1253	258	**2180**
Percent reductions achieved within each type of vehicles	11.8%	11.9%	11.9%	10.4%	9.2%	**10.6%**

Therefore, in the ‘best estimate’ case, the proposed PM-oriented I/M programs in the BMR are expected to yield health benefits that exceed the social costs of the programs (the ‘threshold’ for achieving this goal is a 4% overall PM emission reduction achieved by the programs, as found in Chapter 4). Using the 10.6% overall PM emission reduction to run the health benefit analysis spreadsheet developed in Chapter 4 shows that the total annual benefits of the I/M programs are 382.2 million 2000 US$, and by subtracting the total annual costs (147.1 million 2000 US$) from the benefits, the net benefits of the programs are estimated to be 241.9 million 2000 US$ in 2008.

In next section, a series of sensitivity analyses are conducted to examine the sensitivity of the effectiveness of the I/M programs in terms of the percent of overall PM_{10} emission reduction to the key design elements. All the analyses focus on the estimates of emission reductions available from the PM-related I/M programs in the year 2008.

5.4 Examining the roles of key design elements on the emission reduction benefits of I/M programs

5.4.1 The effects of testing cut-points on overall emissions benefits

The 'best-estimate' in Section 5.3 is based on the assumption that failure cut-points are 75% of the baseline emission rates for each vehicle type. Since the cut-points determine the size of the initial problem vehicle pool (a more stringent testing cut-point is likely to result in more vehicles with 'excess' emissions and, hence, subject to repair or replacement), changes to the cut-points will result in changes in the other two inputs: problem vehicles as percent of total vehicles and problem vehicles as percent of total emissions. It is difficult to estimate the magnitude of changes in these two variables as a result of the changes in failure cut-points without knowing the distribution of emission rates. Here it is assumed that slight changes in cut-points do not change the values of the two variables (this may be true in the case that the majority of good vehicles have emission rates much lower than the cut-points, and the majority of problem vehicles have emission rates much higher than the cut-points), so in this case failure cut-points only affect the new emission rates of problem vehicles after repairs and retest. Based on this assumption, a cut-points sensitivity analysis is conducted. Figure 5.1 shows the results.

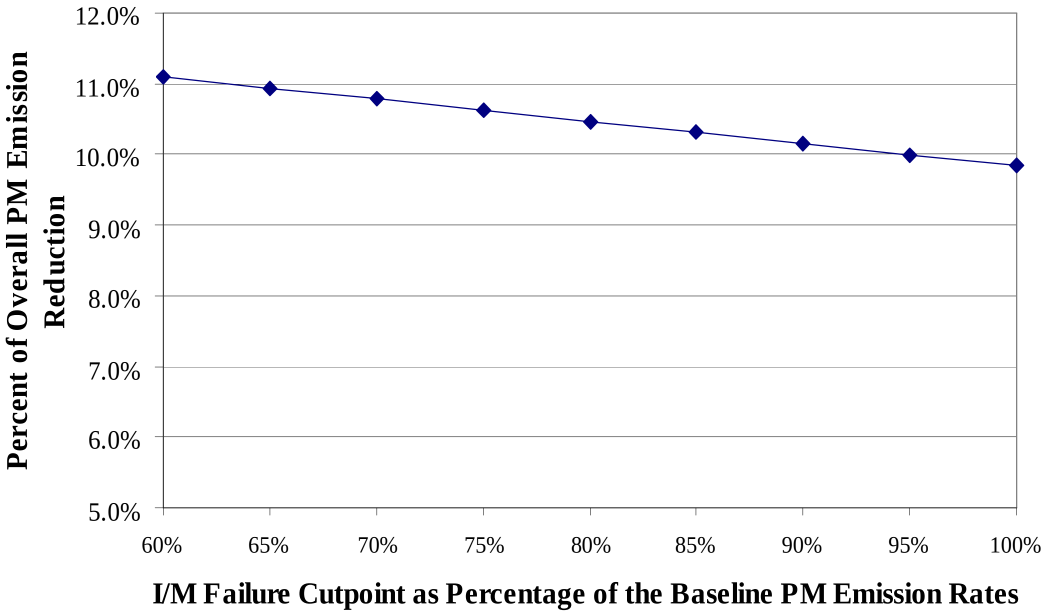

Figure 5.1 Effects of Testing Cut-points on the Percentage of Overall Emission Reduction by I/M Programs

Figure 5.1 is generated based on the assumption that all the other input variables are independent of the failure cut-points, i.e. changing the cut-points while holding all other variables constant to examine the sensitivity of overall PM emission reductions to failure cut-points. Figure 5.1 indicates that the when cut-points decrease from 100% to 60% of the baseline emission rates, the percent of overall emission reduction increases from 9.8% to 11.1%. Therefore, cut-points modifications within a certain range (e.g. from 60-100% of the baseline emission rates in this case) only have modest effects on the overall emission reduction benefits (the percent of overall PM_{10} emission reductions changes from approximately 10% to 11% as the result of changing the failure cut-points from 60-100% of the baseline emission rates), because most vehicles are considered to emit at levels well outside the range (either higher or lower the cut-points). This study considers that other I/M design elements may have more significant effects on the emission reduction benefits achieved by I/M programs. In general, important variables in

I/M design that policy makers need to address include program participation rate, problem vehicle identification rate, effective emission repair rate and problem vehicle illegal operation rate. In the following section, the impacts of these variables on emission reduction are analyzed.

5.4.2 Key variables affecting I/M effectiveness

5.4.2.1 Participation rate and problem vehicle identification rate associated with I/M programs

Participation rate (PartiRate) and problem vehicle identification rate (IdenRate) are two key elements to address in designing I/M programs. Participation rate represents the levels of program enforcement. A successful I/M program minimizes vehicle violations (vehicles required by an I/M program do not participate in the program). The 'best estimate' case in Section 5.3 assumes the majority (90%) of vehicles in the BMR required by I/M will participate in the programs, i.e. they will undertake appropriate emission inspection (reasons discussed in Table 5.1). However, if a large fraction of vehicles subject to I/M test escape from the inspection process, the emission reduction benefits of I/M are expected to decrease considerably. Problem vehicle identification rate represents the ability of I/M programs to identify vehicles that exceed the emission standards and thus need emission repairs.

Sensitivity tests of each of the two variables are conducted by changing the input value of one variable while holding all other inputs constant (presuming that all input variables are independent of each other). The goal is to examine the sensitivity of the percent of overall emission reduction to program participation rate (PartiRate) or problem vehicle identification rate (IdenRate). Figure 5.2 shows the results.

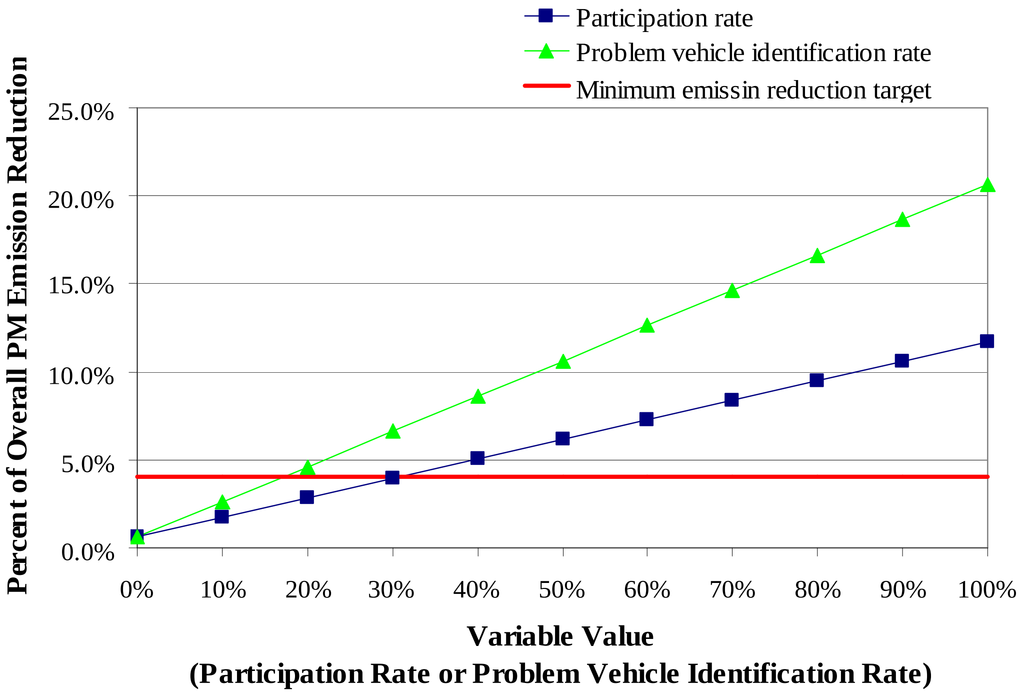

Figure 5.2 Effects of Participation Rate and Problem Vehicle Identification Rate on the Percent of Overall PM Emission Reduction by I/M Programs

The square-marked and the triangle-marked lines represent the percent of overall PM emission reductions achieved by I/M programs as a function of program participation rate (PartiRate) and problem vehicle identification rate (IdenRate), respectively (as noted in the figure). Each of the two lines was generated by incrementing the value of an individual variable (PartiRate or IdenRate) by 10% at a time (starting from 0% and ending at 100%), while setting all other inputs to their best estimates. And the red solid line represents the minimum percentage of emission reduction required in order for the benefits of the programs to outweigh the costs (the value is 4% as found in Section 4.2.6, Chapter 4).

Figure 5.2 indicates that both participation rate and problem vehicle identification rate are important determinants of overall PM emission reduction benefits achieved by I/M programs. For participation rate, when the value of this variable increases from 0% (lower bound) to 100% (upper bound), the percent of overall emission reductions from vehicles increases from 0.6% to 11.7%; for problem vehicle identification rate, the percent of overall emission reductions from vehicles increases from 0.6% to 20.6% when the variable's value changes from the lowest to the highest. Comparing the effects of the two variables in Figure 5.2 shows that problem vehicle identification rate has a greater impacts on the overall emission reduction benefits than program participate rate, since for the same increment (e.g. 10%) in the two variables, the incremental emission reduction benefits resulting from the change in the problem vehicle identification rate are greater.

In order to achieve the goal of 4% PM emission reduction from motor vehicles, the participation rate is required to be greater than 30%, if all other inputs remain the same values as in the 'best estimate' case. And the requirement for problem vehicle identification rate is 17% when setting the other variables in the spreadsheet to their 'best estimate' values.

5.4.2.2 The impacts of the effectiveness of problem vehicle repairs

Repairing problem vehicles to meet emission standards is the major source of emission reduction available from I/M programs. Three variables in the I/M Design spreadsheet are related to the effectiveness of repairs: GoodRep -- Percent of repair work initially effective; ExEm -- Percent of excess emissions (emissions above allowable levels) from identified problem vehicles reduced by repairs that properly pass an I/M test immediately; and DurRep -- Percent of good repairs that remain durable until the next I/M inspection. The following graph shows the impact of each individual variable on the levels of emission reduction achieved by I/M programs.

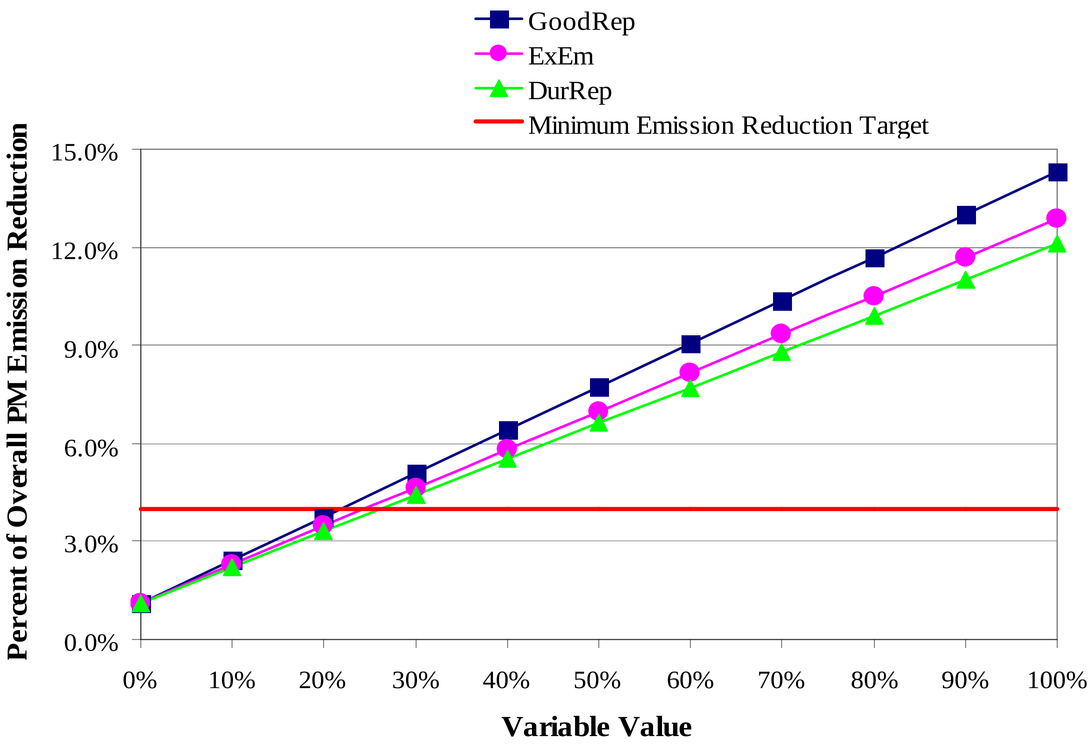

Figure 5.3 Effects of Problem Vehicle Repairs on the Percent of Overall PM Emission Reduction by I/M Programs

The three marked lines (named as GoodRep, ExEm and DurRep) were generated using the same approach as used to generate Figure 5.2: Each line is generated by incrementing the value of the individual variable it represents (GoodRep, ExEm or DurRep) by 10% at a time (starting from 0% and ending at 100%), while setting the other inputs to their 'best estimate' values. And the red solid line also represents the minimum PM emission reduction target of 4%.

Figure 5.3 illustrates that the increase in the values of any of the three variables related to problem vehicle repairs results in considerable improvement in emission reduction performance by the I/M programs, as these variables are key determinants of the effectiveness of emission repairs. Comparing the three marked lines in Figure 5.3 shows that while the same increment in any one of the three variables results in approximately similar incremental emission reduction benefits, the impact of initially effective repair rate (GoodRep) is slightly greater than the impacts of the other two variables. The 4% emission reduction target requires a minimum of 22% of repair work initially effective (GoodRep), or 25% of excess emissions from identified problem vehicles reduced by repairs (ExEm), or 26% of repairs that properly pass an I/M remain durable until the next I/M inspection (DurRep).

5.4.2.3 The impacts of illegal operation by problem vehicles

Illegal operation in this section refers specifically to failed vehicles that continue to run on roads without appropriate repairs or certificates of waiver (the variable IllegalVeh in Table 5.1). There are other types of illegal operation in I/M program implementation. For example, vehicles may run on roads without taking the inspection required by the programs. This latter type of illegal operation is considered in the program participation rate variable, so it is not taken into account here.

Illegal operation by failed vehicles may considerably damage the performance of I/M programs, since these vehicles are identified as gross emitters. Using the same sensitivity test approach as in Figure 5.2 and 5.3, Figure 5.4 is generated, which shows the effects of failed vehicle illegal operation rate on the levels of overall emission reduction achieved by I/M programs.

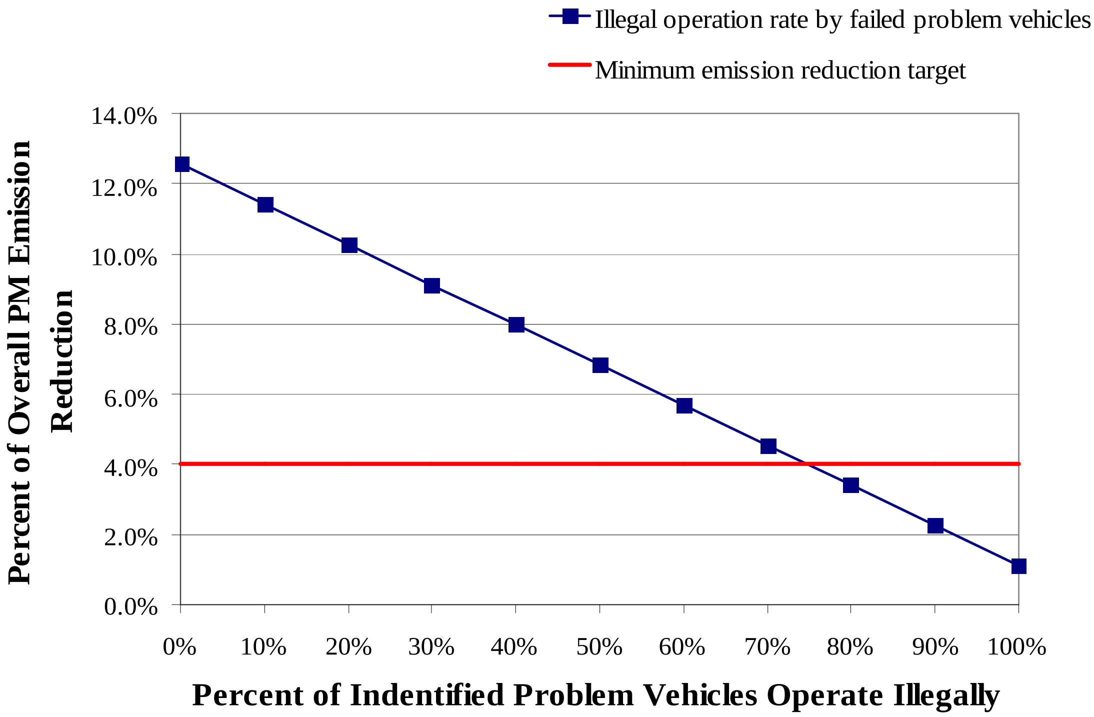

Figure 5.4 Effects of Problem Vehicle Illegal Operation on Overall Emission Reduction Benefits

Therefore, the increases in illegal operation rate by failed problem vehicles can substantially reduce the emission reduction benefits achieved by I/M programs. In order to achieve the goal of 4% PM_{10} emission reduction from motor vehicles, the rate of failed problem vehicle illegal operation should not go over 75%, presuming that the performance of the other variables is at the level of the 'best estimate'. In reality, it may be unrealistic to achieve 0% illegal operation by failed problem vehicles. However, minimizing the problem vehicle illegal operation rate is an essential I/M design element to improve the effectiveness of I/M programs.

5.4.2.4 Improving the emission reduction effectiveness of I/M programs

In defining the policy scenarios in this study (Section 2.3.2, Chapter 2), it was discussed that a 25% PM_{10} emission reduction in the BMR was proposed as an upper bound target of the I/M programs based on the past experience of similar programs in the

U.S. As also as discussed earlier, the levels of PM_{10} emission reductions actually achieved by the programs are significantly uncertain. Furthermore, the cost-benefit analysis in Chapter 4 indicates that a minimum of approximately 4% reduction of the total PM_{10} emissions from vehicles in the BMR is required to ensure the total benefits of the I/M programs considered outweigh the total costs (see Table 4.12 and Figure 4.8 in Chapter 4). The results in Figures 5.2-5.4 indicate that based on the assumptions made in Table 5.1, the improvement in the performance of any individual element is not sufficient to achieve the upper bound target of 25% overall emission reduction initially expected in proposing the programs to be adopted in the BMR.

For example, when one of the key variables discussed above reaches the upper bound, i.e. 100% (for IllegalVeh, the upper bound is 0%), while holding the other variables the same as in the 'best estimate' case, the percent of overall PM_{10} emission reduction is summarized as follows.

Table 5.3 Emission Reduction Benefits in the Case that One Key Variable Reaches the Upper Bound

Variable Reaching the Upper Bound	Percent of Overall PM Emission Reduction from Vehicles
PartiRate (100%)	11.7%
IndenRate (100%)	20.6%
GoodRep (100%)	14.3%
ExEm (100%)	12.9%
DurRep (100%)	12.1%
IllegalVeh (0%)	11.5%

The results in Table 5.3 show in the case that only one variable in the spreadsheet increases while the values of the others remain the same as assumed in Table 5.1, even if the variables achieve complete success, the maximum level of PM emission reduction benefits is 20.6% (when IndenRate reaches 100%). Therefore, the performance of more variables needs to be improved simultaneously.

As two illustrations, when the values of PartiRate, IndenRate, IllegalVeh, GoodRep, ExEm and IllegalVeh are replaced by the lower levels found in the I/M program in southern California's South Coast Air Basin (Eisinger, 2005), the percent of PM emission reduction from motor vehicles increases to 19.1%; and when the values of all the key variables discussed above (PartiRate, IndenRate, IllegalVeh, GoodRep, ExEm, and DurRep) are set to the upper values found in California's I/M, the percent of PM reduction increases to 30.9%. Table 5.3 summarizes the replaced variable values used and the new emission reduction estimates.

Table 5.4 Improving the Effectiveness of the I/M Programs in the BMR by Increasing the Values Associated with Key Design Elements

Variable	Best Estimates in This Study	I/M in Southern California's South Coast Air Basin	
		Lower Value	Upper Value
PartiRate	90%	100%	100%
IndenRate	50%	71%	90%
GoodRep	72%	80%	80%
ExEm	81%	81%	92%
DurRep	86.5%	(79%)*	94%
IllegalVeh	10% for buses and heavy trucks, 20% for light trucks and motorcycles	13%	6.6%
Percent of overall PM_{10} emission reduction by I/M	**10.6%**	**19.1%**	**30.9%**

**Note*: Data in parentheses are not used in the calculation since they are smaller than the 'best estimate' in this study.

5.4.3 Emission reduction benefits due to the change in vehicle population growth

The 'best estimate' in Table 5.2 is based on the assumption that the implementation of the new PM-related I/M programs causes 10% decrease in average annual vehicle growth rate in the BMR (discussions in Section 4.2.2, Chapter 4). This assumption is associated with the variable BenefitsGrow (the percent of emission reductions achieved due to the changes in vehicle population growth as a result of I/M enforcement). Sensitivity test is conducted by changing the percentage decrease in annual vehicle growth rate from 10% to 0% (no change in the annual growth rate), 20% or 30%, while holding all other input unchanged. The following figure summarizes the results.

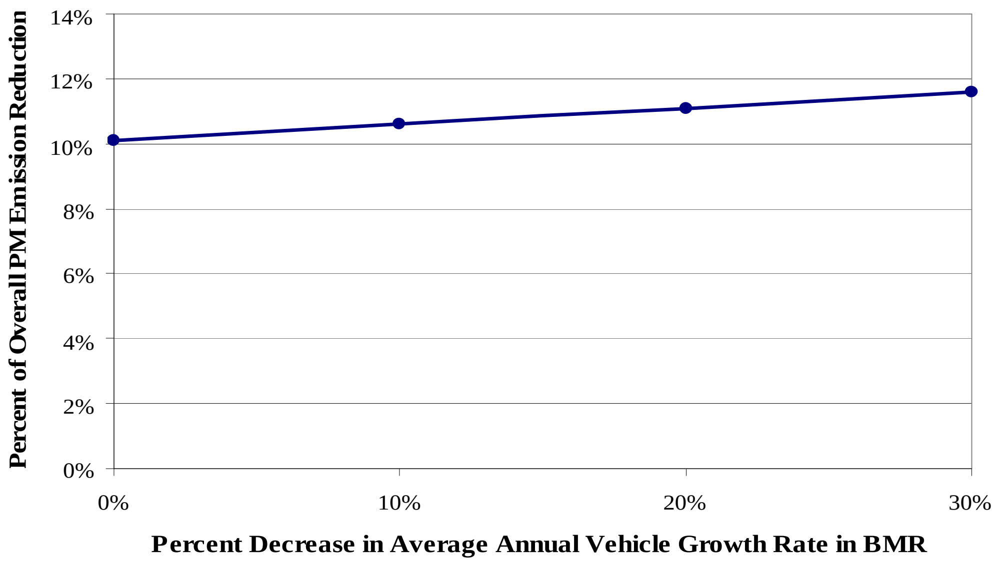

Figure 5.5 Impact of the Change in Vehicle Growth Rate on the Overall PM_{10} Emission Reductions by the I/M Programs

Past experience in rapidly developing metropolitan areas in Asia shows that the introduction of vehicle I/M programs may slightly slow down the fast growth of motor vehicles in these areas and it is expected that the percent decrease in average annual vehicle growth rate falls into the range of 0-30%. Figure 5.5 indicates that the change in the assumption about the percent decrease in annual vehicle growth rate has modest impact on the overall emission reduction benefits, when the change falls into the range of 0-30%.

5.5 Vehicle ages for testing

So far an assumption has been made in running the 'I/M Design' spreadsheet that all diesel vehicles and motorcycles (vehicle types targeted by the PM-related I/M programs) are subject to the inspection process. Conceptually, I/M aims at achieving maximum emission reductions while minimizing vehicles subject to inspection. Traditional light-duty vehicle I/M programs usually exclude some fraction of vehicles (e.g. vehicles less than 3 years old) from the inspection process, based on the assumption that these vehicles do not significantly contribute to the excess emissions. However, this assumption has been challenged by some scientific studies, which found that all vehicles, regardless of their ages, may become grossly polluting without proper maintenance (Beaton, et al, 1995).

The existing I/M programs in the BMR require all commercial trucks and buses to be inspected annually; taxis to be inspected semi-annually; passenger cars, vans, and pick-up trucks older than 7 years and motorcycles older than 5 years to be inspected annually (Parsons, 2001). The following table summarizes the vehicle age distribution information in the BMR.

Table 5.5 Vehicle Ages in the Bangkok Metropolitan Region (as of 2000)

Vehicle Type		Ages (1)
City Bus	Euro 0	Extremely old, many are about 30 years old
	Euro 1	Most are five to seven years old
	Euro 2	Introduced on 1998
City Truck		Average age: 9 years, 95% less than 20 years
Long Haul Truck/Bus		N/A
Light Duty Truck		Average age: 7 years, 95% less than 20 years
Passenger Car		Average age: 10 years, 80% less than 20 years
Motorcycle (2)		35% are between 6 and 10 years old and 4.6% are more than 10 years old

Note: Sources: (1) Parsons, 2001; (2) World Bank, 2003.

While the new PM-related I/M programs in the BMR may continue to inspect all commercial buses and trucks, earlier inspection for the other types of vehicles, particularly for diesel-fueled light trucks and motorcycles, may be considered, given the severity of the air pollution situation in this area attributable to transportation. The fleet of light-duty diesel trucks has the greatest projected growth rate among all vehicle types. As a result, the relative contribution of this vehicle type to the overall PM emissions from motor vehicles will continue to increase over time, as shown in the following graph.

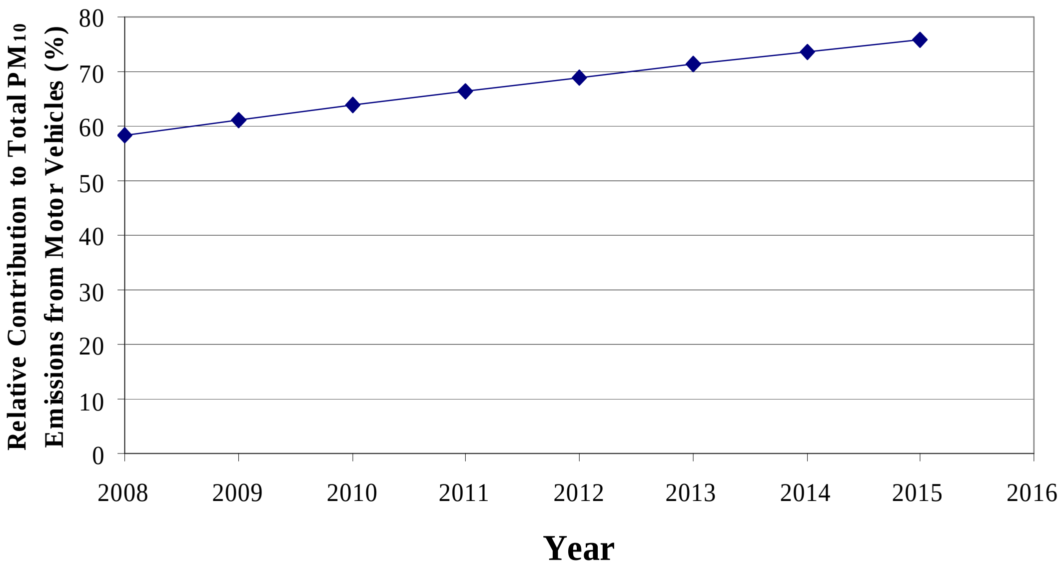

Figure 5.6 Projected Relative Contributions of Light-Duty Diesel Trucks to Total PM_{10} Emissions from Motor Vehicles: 2008-2015

Therefore, future PM-related I/M in the BMR may need to further address light diesel trucks by increasing the frequency of inspection. Furthermore, motorcycles generally turnover faster than other vehicle types. For example, the estimated average age of motorcycles in Bangkok in 1999 was 4.5 years (World Bank, 2003). Given this, they may not be well-maintained during their short life time without enforcing effective I/M programs, resulting in increasing emissions from these vehicles. The World Bank study (World Bank, 2003) found that 20% of motorcycles aged 5 years or fewer failed an initial emissions test, and 35% for older motorcycles. Therefore, if I/M programs only target motorcycles aged more than 5 years, the emission reduction benefits are likely to be much smaller than expected.

Chapter 6 Uncertainty and Sensitivity Analysis

6.1 Overview

In general, sensitivity analyses are performed to identify the parameters that contribute significantly to the uncertainty in predicting risk, given that the uncertainty in different premises do not contribute equally, and uncertainty analyses are carried out to propagate uncertainty in parameters and describe the overall uncertainty in model predictions on which conclusions are based (Crawford-Brown, 1999). In this study, the estimates of total health benefits under the policy scenarios considered are based on several key assumptions that are uncertain. Given this, an objective of sensitivity and uncertainty analyses is to determine how sensitive the best estimates of health benefits associated with control policies are to the uncertainty in each premise made in calculating the benefits. More importantly, the question to address is how the existence of uncertainty about the important parameters may affect decision-making. Section 6.2 focuses on sensitivity analysis of the health benefit analysis conducted in Chapter 4, and Section 6.3 focuses on uncertainty analysis of the total health benefits achieved by the PM-related I/M programs.

6.2 Sensitivity analysis

6.2.1 Sensitivity analysis of the health benefits of PM_{10} control policies in the transportation sector

Sensitivity analysis adjusts one premise (or parameter) within a model at a time to examine how sensitive the final estimate is to the alteration of that premise. In the following, five premises are examined for their roles in changing the estimates of health benefits achieved.

***Premise 1*: Source-specific v.s. non-source-specific PM_{10} health risk**

In the earlier analysis, in order to reflect the fact that the effects of PM_{10} from mobile sources are approximately 1.5 times larger than the overall health effects of PM_{10}, an adjustment factor of 1.5 was used to estimate the mobile source-specific health risk of PM_{10}, based on the findings from the stratified meta-analysis conducted in this study. In sensitivity analysis, the effect of the source-specific PM_{10} health risk assumption is examined by estimating the total health benefits with and without using the 1.5 adjustment factor.

***Premise 2*: Acute v.s. chronic mortality effects of particulate matter pollution**

The analysis in Chapter 5 considers acute mortality effect by using time-series studies in computing the total health benefits. As discussed in Section 2.1.5.1, Chapter 2, the sensitivity of the total health benefits to the short-term v.s. long-term mortality effects can be examined by replacing time-series studies with cohort studies. Moreover, the long-term effects of ambient particulate matter are currently considered to be solely attributable to the exposure to the fine portion of PM_{10}, i.e. $PM_{2.5}$. It has also been discussed that related cohort studies are only available in the U.S. and Europe, but not in Thailand or other developing countries. Among the existing cohort studies in the U.S, Pope III et al (2002) is selected to use in this study because the study includes the largest cohort size and area coverage (Wang & Mauzerall, 2006). This study targeted an adult cohort of ages 30 and older. However, this does not mean that particulate matter air pollution has no health impacts on those aged under 30. In reanalyzing the Harvard Six Cities Study, Krewski et al (2005) found that there was limited evidence of variation in risk with age. Given this evidence, this study assumes that the same risk estimate applies to the remaining population (aged under 30). And the study by Woodruff et al (1997) is selected for postneonatal infant (28 days – 1 year) mortality estimation. The following table summarizes the selected cohort studies and their findings.

Table 6.1 Cohort Studies Used to Estimated the Chronic Mortality of $PM_{2.5}$ in This Study

Health Endpoints	Reference	Study Area	Age Group	Percentage Increase in All-Cause Mortality per 1-μg/m^3 Increase in Annual-Average $PM_{2.5}$ Concentration	
				Mean Estimate	95% CI
Chronic Adult Mortality	Pope et al, 2002	U.S.	Adult: ≥ 30	0.58%	0.20-1.1%
Chronic Infant Mortality	Woodruff et al, 1997	U.S.	Infant: 28 days – 1 year	0.39%	0.20-0.68%

In addition, due to the lack of $PM_{2.5}$ monitoring data in the baseline year 2000 in Thailand, the ratio of $PM_{2.5}/PM_{10}$ was abstracted from the literature and used to estimate ambient $PM_{2.5}$ concentrations based on the available PM_{10} monitoring data. Note that this study focuses on diesel exhaust particulates. An emission inventory developed by California Air Resource Board indicates that approximately 98% of the particles emitted from diesel engines are less than 10 μm in diameter, 94% less than 2.5 μm in diameter, and 92% less than 1.0 μm in diameter (National Toxicology Program, 2005). Based on this information, it is estimated that the ratio of $PM_{2.5}/PM_{10}$ is about 0.959 $(94\% \div 98\% \approx 0.959)$.

Premise 3: **Exponential v.s. linear health impact function**

The analysis in Chapter 5 uses an exponential health impact function ($\Delta y = y_0 \times (e^{\beta \cdot \Delta x} - 1)$, refer to Section 3.3 in Chapter 3 for details) to estimate the number of mortality or morbidity. This sensitivity analysis further uses a linear health impact function ($\Delta y = y_0 \times \beta \times \Delta x$, details in Section 3.3) to examine the sensitivity of the total health benefits to the shape (exponential or linear) of the concentration-response functions that relate the levels of ambient PM_{10} to health endpoints.

Premise 4: **Registered v.s. real population**

In the earlier analysis, the total health benefits are estimated using the registered population to calculate the total number of population affected by air pollution. However, as a typical problem in large urban areas in developing countries, the actual population in the study area is still largely unknown, as there are many people who commute to work in Bangkok or live in the city without registration (United Nations Environment Program, 2004). A study on the un-registered population in Bangkok indicated that the ratio of true population to the registered population is approximately 1.57 (United Nations Environment Program, 2004), indicating that the total benefits attributable to air pollution control may be underestimated by relying on registered population. The sensitivity analysis is conducted by assuming that the actual population is 1.57 times of the registered population, and this ratio is applied to Bangkok and its five surrounding provinces.

Premise 5: **Exposure to the PM_{10} levels at the permanent v.s. roadside monitoring stations**

The last sensitivity analysis is conducted to examine the changes in the expected health benefits if some fraction of the population in the BMR is exposed to the roadside PM_{10} levels. The analysis in Chapter 4 uses the ambient PM_{10} concentrations from eight air quality monitoring stations in the BMR as the surrogate of exposure, presuming that the general population is exposed to the pollution levels at the monitoring sites. All of the eight are permanent stations located in residential, business or school areas, and they are considered to be representative of most residents' exposure levels. There are also temporary roadside stations in the BMR for short-term (e.g. from 15 days to 1 month) air quality monitoring, since it is are concerned that the air quality areas near road traffic can be much worse than the residential areas due to direct emissions from motor vehicles. For example, in the year 2000, 20 roadside temporary stations monitored daily 24-hour average PM_{10} for approximately 300 days (each station was only used for about half to one month with only one station used on any single day). The following graph shows the correlation between a roadside station and a permanent station on 16 days in January 2000.

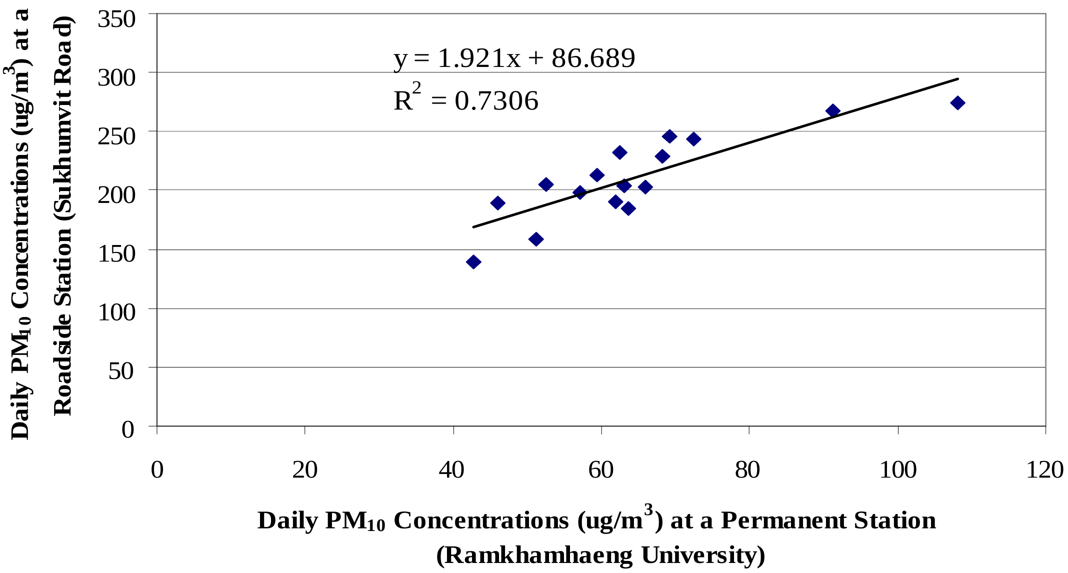

Figure 6.1 Correlation Between Daily 24-Hour Average PM_{10} Concentrations at a Permanent Station and a Roadside Station, January 6-21, 2000

Therefore, daily PM_{10} concentrations at the two stations correlated well ($R^2 = 0.73$). And by comparing the available daily PM_{10} concentrations at the roadside stations with the daily concentrations at the eight permanent stations, it is found that daily PM_{10} levels at the roadside stations were generally 2-3 times higher than those at the permanent stations (the maximum ratio of C_R / C_P was 7.4, minimum was 0.6, and the average was 2.5).

It is largely uncertain what fraction of exposure takes place at the roadside levels in the study area. In general, people who live or work close to road traffic (e.g. traffic policemen, street vendors, etc) are exposed to more traffic-related air pollution. In testing the sensitivity of total health benefits to different levels of exposure (PM_{10} levels at general residential and business areas v.s. levels at road traffic areas), it is assumed that 80% of total exposure is at the levels found in the permanent stations and the remaining 20% is at the levels found in the roadside stations. And it is assumed that the roadside PM_{10} levels equal to 2.5 times of the levels at the permanent stations, based on the average ratio. Therefore, the average PM_{10} exposure level in this sensitivity analysis can be expressed as:

$$Exposure = 0.8 \times C_P + 0.2 \times (2.5 \times C_P) \qquad (6\text{-}1)$$

As compared to the exposure in the baseline analysis in earlier chapters:

$$Exposure = C_P \qquad (6\text{-}2)$$

Where, C_P is a 24-hour average PM_{10} concentration at a permanent station.

The analyses thus far have found three important assumptions in terms of overall percent PM_{10} emission reductions from motor vehicles achieved by the proposed PM-related I/M programs in the BMR: (1) 10.6%, which is the most likely level of the emission reduction benefits of the programs as found in Chapter 5; (2) 4%, which is the lowest acceptable bound in order for the total health benefits of the programs exceed the costs as found in Chapter 4; and (3) 25%, which an upper bound target that was initially proposed as discussed in Chapter 2. Sensitivity analysis is conducted for the estimates of total health benefits delivered by the I/M programs in the year 2008 based on all the three assumptions. Figures 6.2, 6.3 and 6.4 show the results of sensitivity analysis based on these three assumptions, respectively, followed by a table that summarizes the results of the sensitivity analysis and discussions.

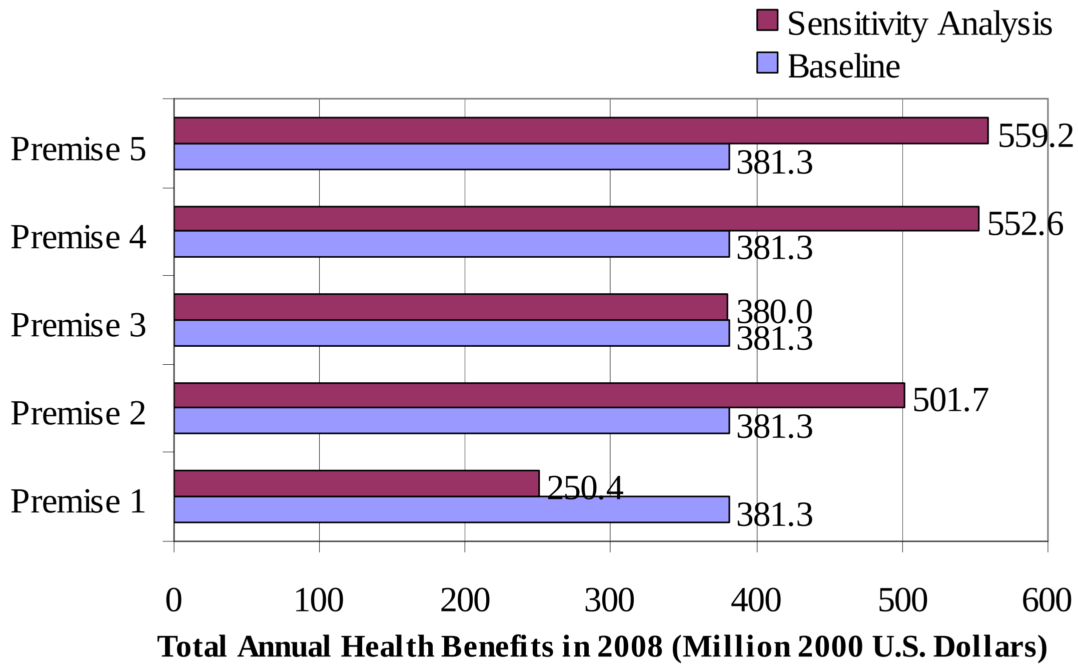

Figure 6.2 Sensitivity Analysis of the Health Benefits of the I/M Programs in 2008 (Assuming 10.6% Overall PM_{10} Emission Reductions from Motor Vehicles)

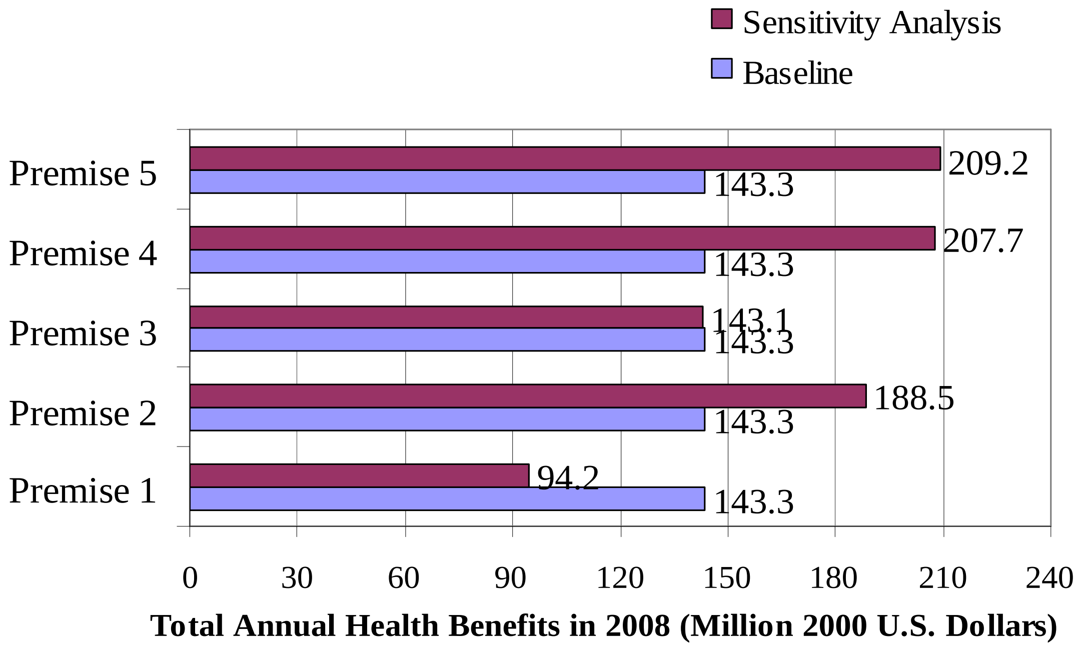

Figure 6.3 Sensitivity Analysis of the Health Benefits of the I/M Programs in 2008 (Assuming 4% Overall PM_{10} Emission Reductions from Motor Vehicles)

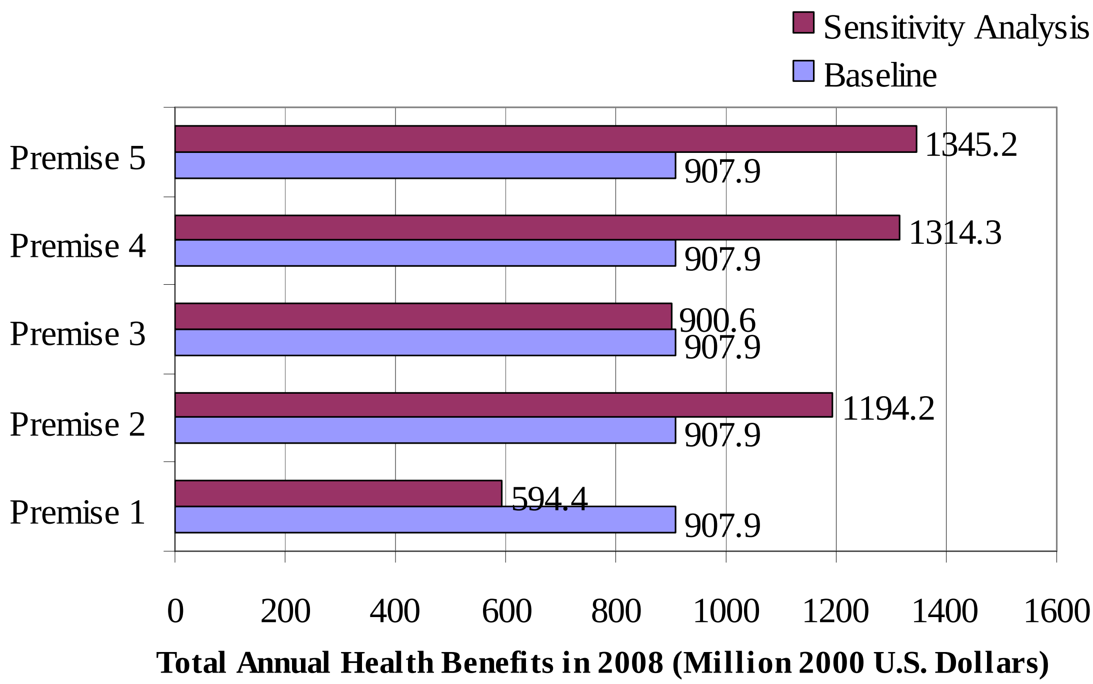

Figure 6.4 Sensitivity Analysis of the Health Benefits of the I/M Programs in 2008 (Assuming 25% Overall PM_{10} Emission Reductions from Motor Vehicles)

Table 6.2 Sensitivity Analysis of the Health Benefits of the PM-Related I/M Programs in the Bangkok Metropolitan Region – Results and Discussions (Year: 2008)

Premise			% Change Relative to the Baseline			Discussion
Number	Baseline	Sensitivity Analysis	10.6% PM_{10} Reduction	4% PM_{10} Reduction	25% PM_{10} Reduction	
1	Using a 1.5 adjustment factor to reflect the greater health risk posed by PM_{10} from mobile sources	Not using a 1.5 adjustment factor (assuming that PM_{10} from mobile sources poses the same health risk as PM_{10} from other sources)	-34.3%	-34.2%	-34.5%	The health benefits of the I/M programs are very likely to be underestimated if the source-specific PM_{10} health risk is not taken into account, since scientific studies consistently found that particles from mobile sources pose a greater risk on human health than particles from other sources.
2	Relying on time-series studies, which capture the acute effects of pollution, to estimate premature mortality attributable to PM_{10} exposure	Relying on cohort studies, which capture the chronic effects of pollution, to estimate premature mortality attributable to PM_{10} exposure	31.6%	31.6%	31.5%	Cohort studies may overestimate the relative risk attributable to long-term exposure to PM_{10} air pollution (reasons discussed in Section 2.1.5.1, Chapter 2), resulting in an overestimate of the mortality effects by relying on these studies.
3	Using an exponential health impact function to estimate the number of cases	Using a linear health impact function to estimate the number of cases	-0.3%	-0.1%	-0.8%	A linear impact function generates health outcome estimates that are very close to those generated by using an exponential function. Conceptually, exponential functions can be approximated by linear functions when exposure levels are low.

Table 6.2 Sensitivity Analysis of the Health Benefits of the PM-Related I/M Programs in the Bangkok Metropolitan Region – Results and Discussions (Year: 2008) (Continued)

Premise			% Change Relative to the Baseline			Discussion
Number	Baseline	Sensitivity Analysis	10.6% PM_{10} Reduction	4% PM_{10} Reduction	25% PM_{10} Reduction	
4	Using registered population to estimate the number of people affected by pollution	Assuming that the real population is 1.57 times the registered population in the BMR	44.9%	45.0%	44.8%	If there are a large number of un-registered residents in the study area, total health benefits attributable to pollution will be underestimated by using registered population to represent the number of people affected. However, currently the true population in the BMR is still largely unknown.
5	Assuming that all population are exposed to the pollution levels found at permanent monitoring stations	Assuming that 20% of total exposure to mobile source PM_{10} is at the roadside levels (assumed to be 2.5 times the levels found at the permanent monitoring stations), and the remaining 80% is at the baseline levels	46.7%	46.0%	48.2%	In general, most people are not expected to be exposed to the roadside level air pollution. However, in the BMR, there may be a higher fraction of people that are exposed to roadside pollution levels due to less prevalence of air-conditioning, and more people work live or work in the roadside areas. The total health benefits of the I/M programs will significantly increase if a large fraction of exposure is at the roadside levels. Nevertheless, further research is warranted to verify this assumption.

6.2.2 Sensitivity analysis of the costs of the I/M programs

The total costs of the PM-related I/M programs estimated in Section 4.2.5 (See Table 4.12) include the testing and administrative costs (estimated by Parsons, 2001) and repair costs (estimated by this study), and the estimation is based on the assumption that the I/M programs are fully successful in terms of compliance, i.e. all vehicles required by the programs participate in the programs and all falling vehicles are repaired after the inspection process. The variable cost components of an I/M program (usually including testing cost and repair cost), however, are affected by the number of vehicles inspected and repaired in the program since the aggregate costs increase as more vehicles are tested. And the number of vehicles inspected is determined by the program participation rate (the variable PartiRate in Table 5.1). The cost estimate in Table 4.12 can be considered as the upper bound. The following analysis tests the sensitivity of the total costs of the I/M programs in the BMR to the program participation rate.

Table 4.12 shows that the total annual testing and administrative costs of the I/M programs are 113.89 million 2000 US$ (estimated by Parsons, 2001) when the participation rate is 100%. Based on Parsons (2001), the fixed cost (including equipment and land use) accounts for about 57% (64.84 million 2000$) and the variable cost (testing cost, including maintenance and labor) account for about 43% (49.05 million 2000$) of the 113.39 million 2000 US$, respectively. It is assumed that the test cost is directly proportional to the participation rate and when the rate is zero, the test cost becomes zero accordingly. And it is assumed that the repair cost is also directly proportional to the participation rate and when the rate is zero, the repair cost becomes zero as well. Based on these assumptions and the cost estimations under a 100% participation rate, the costs under different levels of program participation rates were derived and summarized in the following table.

Table 6.3 Participation Rates and Associated Costs of the I/M Programs in the BMR

I/M Participation Rate	Fixed Cost	Variable Cost		Total Costs
		Testing Cost	Repair Cost	
	(Million 2000 US$)			
0%	64.84	0	0	64.84
10%		4.91	3.32	73.06
20%		9.81	6.64	81.29
30%		14.72	9.95	89.51
40%		19.62	13.27	97.73
50%		24.53	16.59	105.96
60%		29.43	19.91	114.18
70%		34.34	23.23	122.40
80%		39.24	26.54	130.62
90%		44.15	29.86	138.85
100%		49.05	33.18	147.07

Based on the information in Table 6.3, Figure 6.5 was generated to illustrate the changes in the programs' costs resulting from the changes in the participation rate.

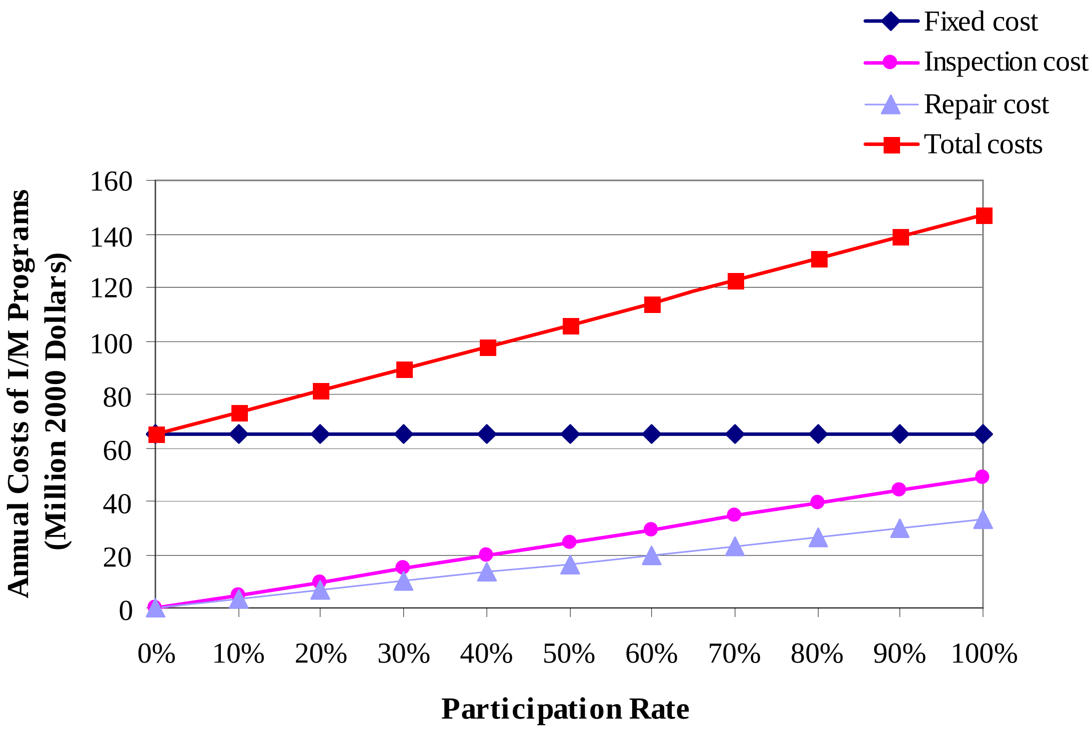

Figure 6.5 The Effects of Participation Rate on the Costs of the I/M Programs

The sensitivity analysis of the total annual costs of the I/M programs in the BMR indicates that the actual costs fall into the range of 64.84 - 147.07 million 2000 US$, depending on the program participation rate (or the total number of vehicles inspected). While the increase in the total number of vehicles inspected will increase the total annual costs of the I/M programs, the increment in costs can be offset by the increase in the total health benefits resulting from a greater participation rate. The following figure shows the total annual benefits, costs and net benefits (total benefits minus total costs) as functions of the participation rate.

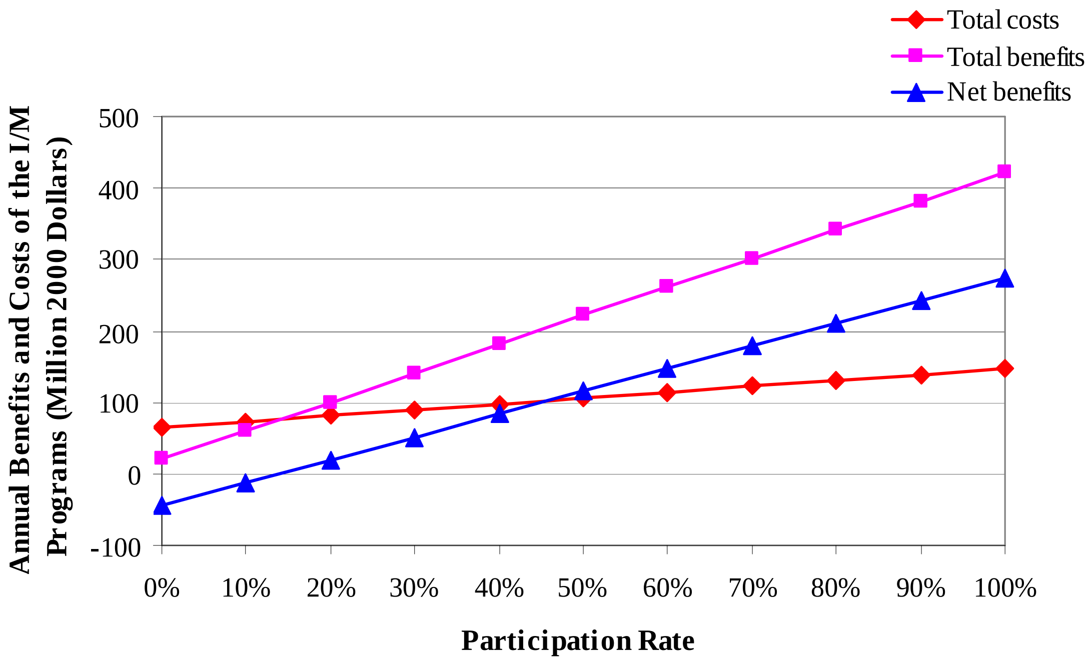

Figure 6.6 Comparing the Annual Benefits and Costs of the I/M Programs Under Different Participation Rates (Year: 2008)

Figure 6.6 indicates that when the participation rate is greater than approximately 15% (resulting in an aggregate level of PM_{10} emission reductions greater than approximately 2-3%, see Figure 5.2), the net benefits of the I/M programs become positive, and the net benefits continue to increase as the participation rate increase. In Section 4.2.6, it was found that a minimum of about 4% PM_{10} emissions reduction from vehicles is required to ensure the net benefits are positive (see Figure 4.8). This is a conservative result because the upper bound cost estimate was used to calculate the net benefits. The minimum acceptable level of emission reduction in terms of producing positive net benefits suggested here is slightly smaller than the 4%, when the change in the total costs of the I/M programs resulting from the change in the participation rate is taken into account.

6.3 Uncertainty analysis

6.3.1 Uncertain variables

In an uncertainty analysis, several parameters are modified simultaneously rather than each one at a time to estimate the overall uncertainty in the final predictions of health effects and benefits. The following table lists the major sources of uncertainty (premises) in the health benefit analysis (conducted in Chapter 4) and the I/M effectiveness evaluation (conducted in Chapter 5), as well as the PDFs assigned to each premise.

Table 6.4 Uncertain Variables and Their PDF Forms

Premise	Probability Density Function (PDF)	Mean (Range)
Health Benefit Analysis		
Percent of overall PM_{10} emission reductions achieved by the PM-related I/M programs	PDF based on the uncertainty analysis of the 'I/M Design' spreadsheet	10.6%
Regional background PM_{10} level	Triangular	3.02 $\mu g/m^3$ (0-6.04 $\mu g/m^3$), refer to Section 3.1.1
The relative contribution to total PM_{10} emissions by mobile sources	Uniform	31.62% (15.81- 47.43%), refer to Section 3.1.1
Concentration-response coefficients	Normal	Refer to Table 3.11
Economic valuation of each health endpoint attributable to PM_{10} pollution	Custom (Probability weights 0.33, 0.5 and 0.17 for the low, central and high values, respectively)	Refer to Table 3.13
I/M Program Effectiveness Evaluation		
Problem vehicles as percent total vehicles	Uniform	10% (5-15%) for buses and heavy trucks; 17.5% (9-26%) for light trucks; and 25% (13-38%) for motorcycles, refer to Section 5.2
Problem vehicles as percent of total PM_{10} emissions	Uniform	65% (50-80%), refer to Section 5.2
I/M program participation rate (PartiRate)	Triangle	Refer to Table 5.1
Problem vehicle identification rate (IdenRate)	Triangle	
Percent of failed vehicles scrapped (ScrapFrac)	Triangle	
Percent of identified problem vehicles waived by an I/M program (VehWaive)	Uniform	
Percent of illegal operation by identified problem vehicles (IllegalVeh)	Triangle	
Percent of repair work initially effective (GoodRep)	Triangle	
Percent of excess emission addressed by repairs (ExEm)	Triangle	
Percent of repairs remaining durable (DurRep)	Triangle	

In order to examine the uncertainty in the actual level of emission reduction available from the I/M programs and in the health benefits achieved by the programs, the health benefit analysis spreadsheet (developed in Chapter 5) and the I/M program effectiveness evaluation spreadsheet (developed in Chapter 6) were linked to each other (the variable that links the two spreadsheets is the overall percent of PM_{10} emission reduction from motor vehicles by the I/M programs). All the variables that involve uncertainties were assumed to be independent of each other, and thus the change in one variable does not affect the values of the other variables. The uncertainty analysis was conducted using the Monte Carlo simulation technique and it is performed in Crystal Ball Analysis Software.

6.3.2 The lowest acceptable level of PM_{10} emission reductions under uncertainty

In Chapter 4, it was found that a 4% of PM_{10} emission reduction from motor vehicles in the BMR is required to ensure that the health benefits exceed the costs of the I/M programs, and this is estimated based on the mean values of the input variables. If the uncertainty in the variables listed in Table 6.3 is taken into account, the following figure shows the cumulative distribution functions (CDF) of the total health benefits in 2008 when different levels of overall PM_{10} emission reductions from motor vehicles are assumed.

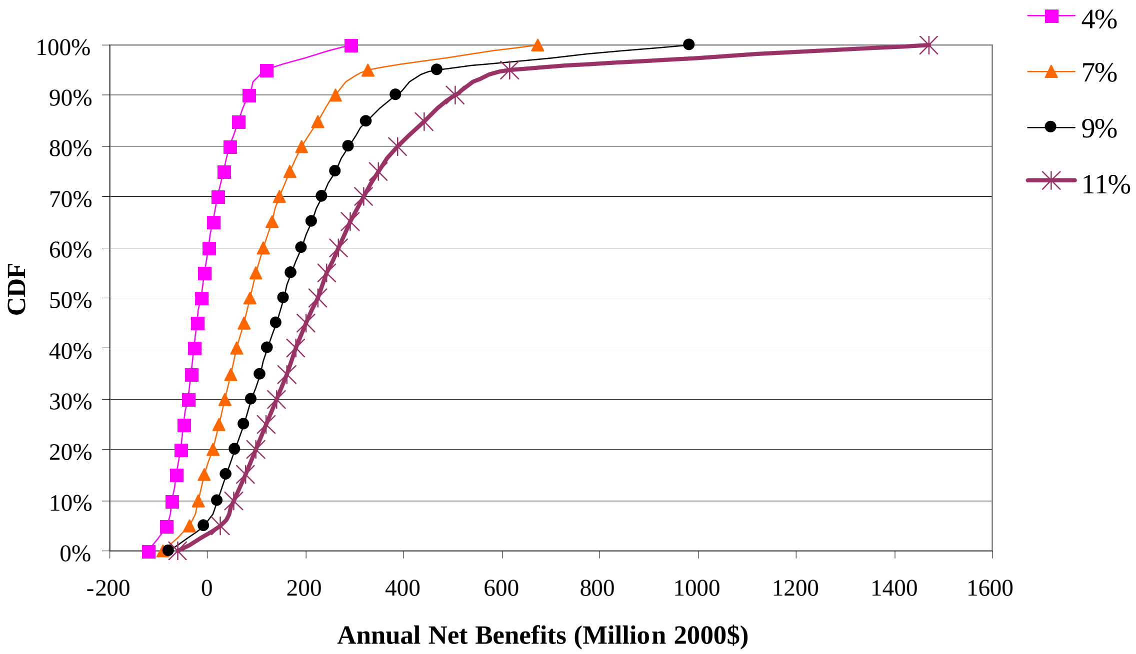

Figure 6.7 Cumulative Distribution Functions (CDF) of the Total Net Benefits at Different Levels of Overall PM_{10} Emission Reduction from Motor Vehicles in the Bangkok Metropolitan Region (Year: 2008)

Figure 6.5 indicates that if the uncertainties in the input variables are taken into account in the health benefits analysis (including uncertainties in the background PM_{10} level, the relative contribution to total PM_{10} emissions by mobile sources, concentration-response coefficients and valuation), there is only 45-50% confidence that the benefits exceed the costs (the net benefits are positive) at the 4% overall emission reduction level. A 9% overall emission reductions is required to improve the confidence level to 95% (at the 5th percentile in Figure 6.2).

6.3.3 Uncertainty in the PM_{10} emission reduction levels and its roles in predicting the total health benefits

An additional uncertainty analysis was conducted when the health benefits analysis spreadsheet and the 'I/M Design' spreadsheet are linked through the variable 'percent of PM_{10} emission reduction from motor vehicles'. In this case, a PDF for the percent of overall PM_{10} emission reductions was generated in the 'I/M Design' spreadsheet, and simulated values were used in the health benefits analysis spreadsheet to calculate the corresponding total annual health benefits. The outcomes were a probability distribution for the percent of overall PM_{10} emission reductions and a probability distribution for the total annual health benefits. Figure 6.6 and Figure 6.7 show the CDFs of these two forecast variables, respectively.

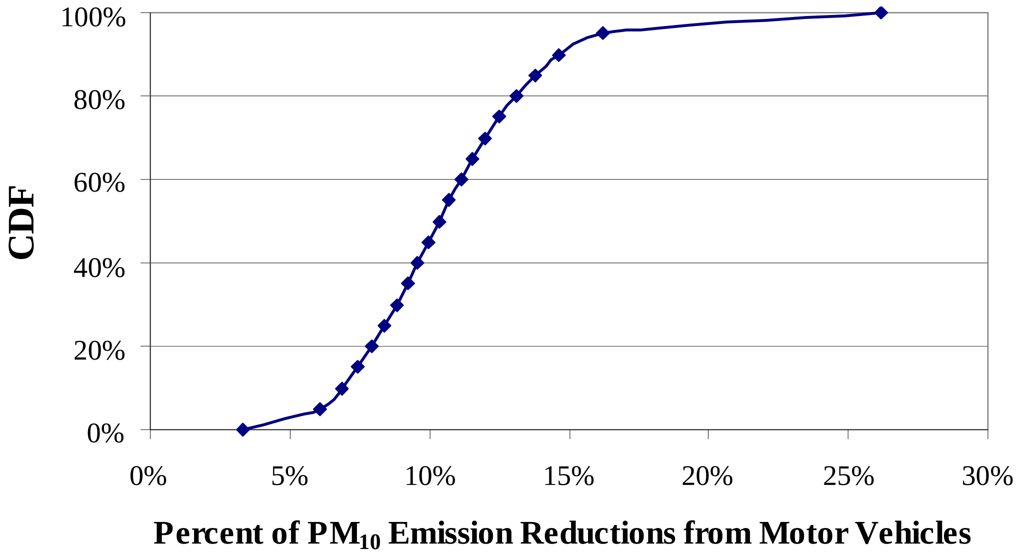

Figure 6.8 Cumulative Distribution Functions (CDF) of the Percent of PM_{10} Emission Reductions Achieved by the PM-Related I/M Programs in the Bangkok Metropolitan Region (Year: 2008)

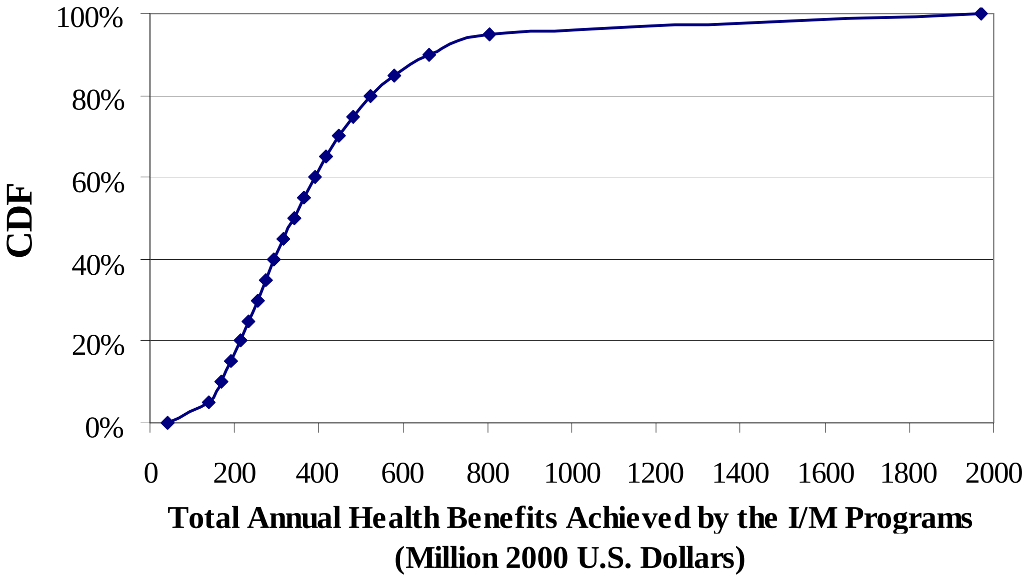

Figure 6.9 Cumulative Distribution Functions (CDF) of the Total Health Benefits Achieved by the PM-Related I/M Programs in the Bangkok Metropolitan Region (Year: 2008)

Figure 6.6 shows that there is 95% confidence (at the 5th percentile in Figure 6.5) that the percent of overall PM_{10} emissions reductions will be greater than approximately 6%. And Figure 6.7 shows that there is 95% confidence (at the 5th percentile in Figure 6.6) that the total annual health benefits by the I/M programs will be greater than 140 million 2000 U.S. dollars, which approximately equals the total annual costs of the programs. Therefore, there is 95% confidence that the goal of positive net benefits will be met.

Furthermore, Figure 6.8 compares the PDFs of the total health benefits when the uncertainty in the levels of PM_{10} emission reductions are excluded (the value is fixed at 25%) with the case when the uncertainty is included. Details of these two PDFs are described below.

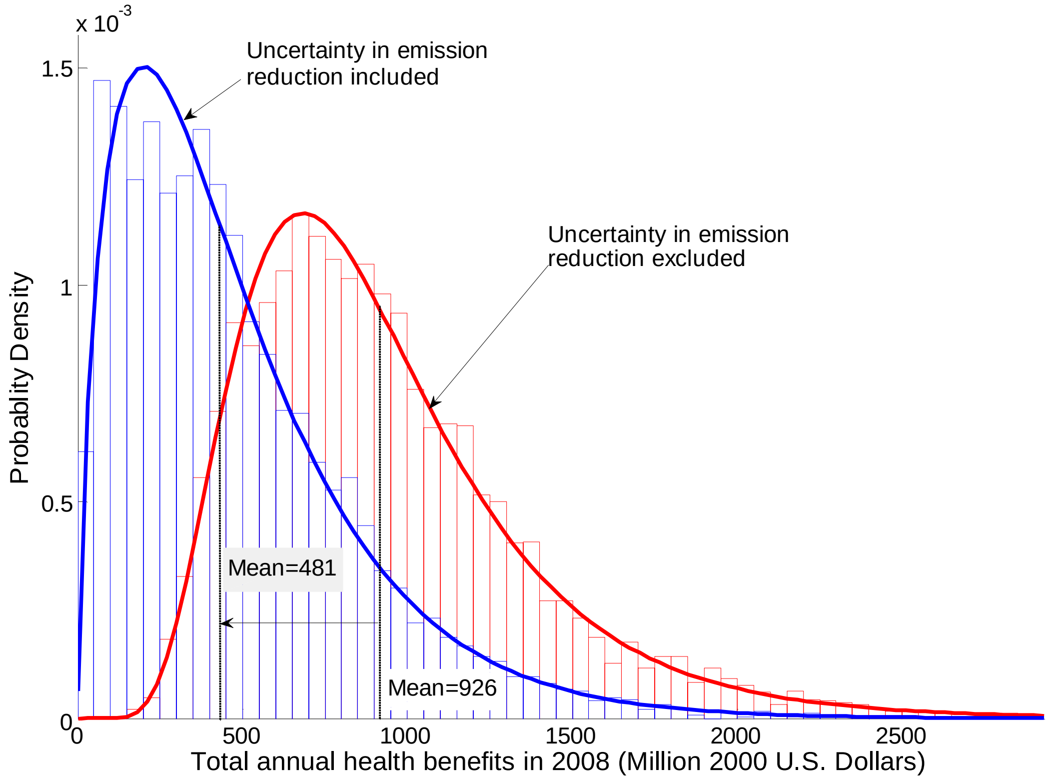

Figure 6.10 Comparing the Probability Density Functions of Total Annual Health Benefits of the I/M Programs in When the Uncertainty in Emission Reductions is Included or Excluded

Specifically, the PDF to the right (shown in the red color) was generated by including all the uncertainty sources in the health benefits analysis spreadsheet listed in Table 6.3 except for the uncertainty in the percent of overall PM_{10} emission reductions achieved by the PM-related I/M programs (therefore, the uncertainty sources included were the background PM_{10} level, the relative contribution to total PM_{10} emissions by mobile sources, the concentration-response coefficients and the economic valuation of each health endpoint attributable to PM_{10} pollution). And the value of the percent of overall PM_{10} emission reductions achieved by the I/M programs was fixed to be 25%, which is the upper bound target that has been proposed. This is a traditional approach to characterize the overall uncertainty in the benefits, namely, the uncertainty in the effect of a policy is not taken into account. And the PDF to the left (shown in the blue color) was generated by including one more source of uncertainty, namely, the uncertainty in the percent of overall PM_{10} emission reductions that has been addressed in this dissertation, in addition to all the other uncertainty sources. The PDF of this variable was generated by conducting and uncertainty analysis in the 'I/M Design' spreadsheet.

Figure 6.8 indicates that when the uncertainty in the levels of PM_{10} emission reductions achieved by the I/M programs is introduced in estimating the total annual health benefits yielded by the programs, rather than simply considering the proposed upper bound as it was done in the past, the mean estimate of health benefits shifted toward the left. A key message conveyed in this analysis is that ignoring the uncertainty in the actual effects of a control policy is likely to result in an overestimate of the potential benefits of the policy.

6.3.4 Contribution to variance

Contribution to variance is a measure of the fraction of the total uncertainty (variance) in the risk estimate that comes from the uncertainty in a particular parameter, when all parameters are allowed to vary simultaneously (Crawford-Brown, 1999). Research to reduce uncertainty then should focus limited resources on narrowing the uncertainty in the premise showing the greatest contribution to variance (Crawford-Brown, 1999). The following table summarizes the top 10 premises that contribute most significantly to the overall uncertainty in the total health benefits available from the PM-related I/M programs and the top 10 premises that contribute the most significantly to the uncertainty in the percent of overall PM_{10} emission reductions. The values of contribution to variance by all the uncertain variables are generated automatically when running an uncertainty analysis in Crystal Ball Software and can be obtained by extracting data from Crystal Ball.

Table 6.5 Premises Showing the Greatest Contribution to Variance

Rank	Premise	Contribution to Variance
Health Benefit Analysis		
1	Relative contribution to total PM_{10} emissions by motor vehicles	35.05%
2	Percent of PM_{10} emission reductions achieved by the I/M programs	34.14%
3	Valuation of mortality	14.80%
4	Concentration-response coefficient of chronic bronchitis	5.27%
5	Valuation of chronic bronchitis	4.36%
6	Concentration-response coefficient of acute postneonatal infant mortality	3.43%
7	Concentration-response coefficient of acute adult mortality	1.76%
8	Concentration-response coefficient of acute respiratory symptom days	0.49%
9	Valuation of restricted activity days	0.23%
10	Regional background PM_{10} level	0.19%
I/M program effectiveness evaluation		
1	Problem vehicle identification rate of light-duty trucks (IdenRate)	58.4%
2	Problem vehicle as percent of total PM_{10} emission of light-duty trucks	10.0%
3	Problem vehicle identification rate of buses (IdenRate)	6.4%
4	Percent of repair work initially effective of light-duty trucks (GoodRep)	5.5%
5	Problem vehicle identification rate of motorcycles (IdenRate)	3.3%
6	Percent of excess emissions reduced by repairs of light-duty trucks (ExEm)	3.1%
7	Problem vehicle illegal operation rate of light-duty trucks (IllegalVeh)	2.4%
8	Problem vehicle identification rate of city trucks (IdenRate)	1.8%
9	Program participation rate of light-duty trucks (PartiRate)	1.8%
10	Percent of durable repairs of light-duty trucks (DurRep)	1.2%

Table 6.4 indicates that the two premises that contribute most significantly to the total uncertainty in the health benefits achieved by the I/M programs are the relative contribution to total PM_{10} emissions in the study area by motor vehicles and the percent of PM_{10} emission reductions available from the I/M programs. The uncertainty in the former premise has been discussed in Section 3.1.1, Chapter 3: At present there remains significant uncertainty in the relative share of contribution to the total PM_{10} emissions by different sources. Given this parameter's high contribution to variance, further research should focus on developing a better emission inventory in the BMR. And the uncertainty in the latter premise has been addressed throughout this study. Ignoring the uncertainty in the actual level of PM_{10} emission reductions by the programs is likely to generate

misleading results and policy recommendations. In addition to the two premises, valuation and concentration-response coefficients, in particular those of the mortality health endpoint, also play relatively significant roles in the calculation of health benefits. Further research is desirable to reduce the uncertainty in these premises.

As for the I/M program effectiveness evaluation, it is found that the problem vehicle identification rate of light-duty trucks shows the greatest contribution to variance. The contribution to variance of this premise is large probably both because the overall PM_{10} emission reductions are sensitive to this premise (as found in Section 5.4.2.1, Chapter 5), and because the uncertainty in this premise is large (falls into the range of 0-100%, see Table 5.2 in Chapter 5). Moreover, the problem vehicle identification rates of several other vehicle types (buses, city trucks and motorcycles) are also among the top 10 premises that contribute the greatest to the total uncertainty. Therefore, policy design considerations need to stress this variable in order to narrow its uncertainty and improve its performance.

Furthermore, Table 6.4 indicates that the premises related to light trucks have particularly significant contribution to variance, as seven among the ten premises listed in the table are associated with light trucks. This vehicle type plays a major role in the calculation of overall emission reductions because it is projected to contribute to the largest fraction of total PM_{10} emissions from motor vehicles (see Figure 5.5 in Chapter 5). Therefore, attention probably needs to be directed to the light-duty truck fleet in introducing the PM-related I/M programs studied here.

Chapter 7 Conclusions, Limitations and Future Research

7.1 Conclusions

Urban air pollution is a major public health concern in some developing metropolitan areas in Asia, where rapid economic development and urbanization occurs, and population tends to concentrate. In particular, as polluting industrial facilities are gradually moved out from the central urban areas, the transportation sector becomes the primary source of urban air pollution. In anticipation that vehicle population will continue to grow fast in the near future as the consequence of economic development, the impacts of transportation on urban air quality need to be seriously stressed in order to ensure sustainability of development. The capital city of Thailand, Bangkok, is one of these metropolitan areas where such issues will be increasingly important for policy makers.

Among the criteria air pollutants, airborne particulate matter - the pollutant of the greatest health impacts - is the greatest concern in the Bangkok Metropolitan Region. Since the late 1990s, this area has been suffering from severe PM pollution mainly due to its wide use of diesel-fueled vehicles and motorcycles with commonly poor emission performance. As the government strives to curb pollution attributable to transportation through enforcing a series of policy measures, inspection and maintenance programs targeting PM emissions from in-service vehicles have been proposed as a key policy tool to be implemented because of the poor vehicle maintenance practices in the past. While I/M programs are now widely considered as a key motor vehicle air pollution control measure in severely polluted urban areas, the uncertainty in the actual effects of these programs has not been considered in proposing this policy measure to be adopted in Thailand.

This study produced: (1) An in-depth health benefit analysis to estimate the public health benefits achieved by the proposed PM-related I/M programs as well as the vehicle retrofitting or repowering programs in the BMR; (2) Estimates of the impacts of the actual PM_{10} emission reductions available from the I/M programs on the health benefits achieved, and in a cost-benefit analysis framework, estimates of the minimum level of emission reductions required in order for the benefits to outweigh the costs of the programs; And (3) estimates of the impacts of the key design elements on the levels of emission reductions available from the I/M programs and the minimum performance that these key elements must meet in order for the benefits to be greater than the costs.

As decision makers endeavor to set policy priority in order to better allocate limited resources to tackle the severe air pollution problem, the policy question concerns whether it is likely that the proposed vehicle PM emission control measures produce considerable public health benefits that outweigh the expenditures on policy implementation; and what levels of various program performance that must be achieved to ensure that the large expenditures on air pollution control are balanced by the benefits.

7.1.1 Examining source-specific health risk of PM_{10}

A well established causal effect of air pollution on human health is the primary justification of the expenditures on pollution control. While scientific studies have consistently observed the associations of both short-term and long-term PM_{10} exposure with premature mortality and a variety of illnesses, currently the source-specific PM_{10} health risks are not yet well established. Using a stratified meta-analysis approach, this study quantitatively estimated and compared PM_{10}-mortality coefficients associated with particles from different sources. It was found that PM_{10} from mobile sources poses the greatest health risk: the effects of PM_{10} from mobile sources are about two times larger than the effects of PM_{10} from industrial or crustal sources, and are approximately 1.5 times larger than the overall health effects of PM_{10} from all sources combined. Since earlier epidemiological studies generally did not separate the contributions of PM_{10} from different sources, using the concentration-response coefficients developed by these studies may underestimate the health benefits of controlling PM_{10} emissions from mobile sources. For example, sensitivity analysis indicates that using previous epidemiological studies conducted in Bangkok without taking into account the source-specific PM_{10} risk from mobile emissions provides an estimate of total health benefits that is 34% lower than the best estimate when source-specific risk coefficients are used.

7.1.2 Health benefits of control policies targeting PM_{10} emissions from motor vehicles

Using the health benefits analysis/health impacts assessment framework that integrates air quality modeling, exposure assessment, exposure-response assessment, economic valuation and policy analysis, it is estimated that in the year 2000, there were 1682 deaths and a large number of illnesses in the Bangkok Metropolitan Region attributable to PM_{10} emissions from motor vehicles, resulting in a total economic loss of 2.68 billion 2000 U.S. dollars, which approximately equals 2.4% of Thailand's GDP in that year. Mortality and morbidity are responsible for approximately 51% and 49% of the total health damage costs, respectively. If no further PM emission control policy targeting motor vehicles is introduced, total PM_{10} emissions from vehicles in the BMR will increase considerably over time due to the rapid growth in vehicle population, even if the fleet average emission rates are projected to decrease over time as the result of improved emission control technologies in modern vehicles. The increases in annual PM_{10} emissions from vehicles and the associated annual health damages relative to the baseline year 2000 during years of 2008-2015 are summarized in the following table.

Table 7.1 Projected Total Annual PM_{10} Emissions from Motor Vehicles and the Associated Annual Health Damages in the BMR: 2008-2015 (Business-as-Usual Scenario)

Year	Projected Total Annual PM_{10} Emissions from Motor Vehicles (Tonnes)	% Increase in Emissions Over 2000	Projected Total Annual Health Damages Attributable to PM_{10} Emissions from Vehicles (Million 2000 US$)	% Increase in Health Damages Over 2000
2008	20646	32%	3825.7	43%
2009	21718	39%	4073.8	52%
2010	22923	46%	4355.1	63%
2011	24271	55%	4673.9	75%
2012	25778	65%	5035.2	88%
2013	27458	75%	5444.7	103%
2014	29329	87%	5909.1	121%
2015	31408	101%	6436.0	140%

Control policies targeting PM_{10} emissions from motor vehicles, if successful, are expected to yield substantial public health benefits resulting from the avoided premature death and illnesses. For example in 2008, if the PM-related I/M programs targeting all diesel-fueled vehicles and motorcycles in the BMR achieves a 25% of overall PM_{10} emission reductions from motor vehicles, it is estimated that the programs will save 563 premature deaths, about thirty-three thousand cases of illnesses as well as about thirty-four million acute respiratory symptom days or restricted activity days (see Table 4.9), resulting in a total economic benefits of 908 million 2000 U.S. dollars.

The total costs of an I/M program generally include the costs of equipment, land use, testing as well as vehicle repairs. The total costs of an I/M program are expected not to vary appreciably and can be considered as independent of the levels of emission reductions available from the program. Using empirical data from earlier studies, it is estimated that the total costs of the I/M programs targeting all diesel-fueled vehicles and motorcycles in the BMR are 147 million 2000 U.S. dollars annually. A cost-benefit analysis that considered both the total health benefits and costs as functions of the percent of overall PM_{10} emission reductions achieved by the I/M programs demonstrated that a minimum of about 4% emission reductions is required to ensure the total benefits of the programs outweigh the total costs of the programs (see Figure 4.8). Furthermore, an uncertainty analysis showed that if the uncertainties involved in the inputs to the health benefits analysis were taken into account, 9% of overall PM_{10} emission reductions from vehicles are required so that there is 95% confidence that the health benefits exceed the costs. Based on these findings, it is socially desirable that the I/M programs produce larger levels of vehicular emissions reductions since the net benefits (total benefits minus total costs) of the programs increase greatly as the total emission reductions available from the programs increase.

In addition to the PM-related I/M programs, retrofitting or repowering existing heavy-duty diesel vehicle fleets is expected to deliver further emission reductions from these vehicles. Commonly used measures include installing oxidation catalyst converters, converting to gaseous fuel (compressed natural gas or liquefied petroleum gas), and installing particulate traps. All the retrofitting and repowering programs are recommended to be adopted in combination with related I/M programs in order to prevent the emission control functions of retrofitting or repowering from deteriorating quickly. It is estimated that certain combinations of the PM-related I/M programs and one or more retrofitting or repowering measures can deliver overall PM_{10} emission reductions varying from 38-81%, resulting in total annual health benefits ranging from 1.4 to 3.1 billion 2000 U.S. dollars in the year 2008. Nevertheless, since heavy diesel vehicle retrofitting or repowering may require some fundamental changes in engine systems or in fuel manufacturing, further research is needed on the technological feasibility of these programs as well as their additional impacts on the society.

As uncertainties are involved in most input variables in the health benefits analysis, uncertainty analysis using the Monte Carlo sampling technique was conducted to investigate the overall uncertainty in total health benefits and the variables that contribute the greatest to the total uncertainty. Taking the year 2008 as an example, the results show that there is approximately 95% confidence that the annual health benefits of the I/M programs considered in this study are greater than 147 million 2000 U.S. dollars, which equals to the total costs of the programs. Two premises or variables, namely, the relative contribution to total PM_{10} emissions in the study area by motor vehicles and the percent of PM_{10} emission reductions available from the I/M programs contribute the greatest to variance, followed by valuation (statistical value of life and willingness to pay) and concentration-response coefficients. An important issue addressed here is the actual effectiveness of the programs in terms of levels of emission reductions actually achieved by the programs. As this uncertainty has traditionally been neglected in the cost-benefits analysis of air pollution control policies, one objective of this study is to address this flaw. Uncertainty analysis shows that if the uncertainty in the actual levels of PM_{10} emission reductions are introduced rather than simply assuming full implementation of the I/M programs, the mean estimate of the total annual health benefits shifts to lower values by approximately 50% (see Figure 6.7). This implies that the benefits of the policy measure may be overestimated without taking into account the uncertainty in the actual effects of the policy. This issue is particularly relevant in developing countries since there may be less resource available for tackling air pollution problem.

7.1.3 Evaluating the effectiveness of the PM-related I/M programs

In order to further examine the effectiveness of the I/M programs in terms of reducing PM_{10} emissions from motor vehicles and the key design elements that determine the effectiveness of the programs, a newly developed analysis approach was used to evaluate the program effectiveness and test the sensitivity of the effectiveness to the key elements. As a result, the best estimate of the percent of overall PM_{10} emission reductions available from the proposed PM-oriented I/M programs in the BMR is 10.6%, and it will yield a total of health benefits that outweigh the social costs of the programs (4% is the minimum emission reduction level required to produce positive net benefits). The sensitivity test results are summarized in the following table.

Table 7.2 Sensitivity Test of the Percent Emission Reductions to the Key Design Variables (Year: 2008)

<table>
<tr><th colspan="2">Variable</th><th>Best Estimate</th><th>Sensitivity Test Range</th><th>Minimum Performance to Achieve the 4% PM_{10} Emission Reduction Target</th><th>Sensitivity Tests Result (Change in % Emission Reductions)</th></tr>
<tr><td colspan="2">Failure cut-points</td><td>75% of the baseline emission rates</td><td>60-100% of the baseline emission rates</td><td>N/A</td><td>11.1%-9.8%</td></tr>
<tr><td colspan="2">Program participation rate</td><td>90%</td><td rowspan="6">0-100%</td><td>>30%</td><td>0.6%-11.7%</td></tr>
<tr><td colspan="2">Problem vehicle identification rate</td><td>50%</td><td>>17%</td><td>0.6%-20.6%</td></tr>
<tr><td rowspan="3">Effectiveness of failed vehicle repairs</td><td>Percent of repair work initially effective</td><td>72%</td><td>>22%</td><td>1.1%-14.3%</td></tr>
<tr><td>Percent of excess emissions reduced by good repairs</td><td>81%</td><td>>25%</td><td>1.1%-12.9%</td></tr>
<tr><td>Percent of good repairs that remain durable</td><td>86.5%</td><td>>26%</td><td>1.1%-12.1%</td></tr>
<tr><td colspan="2">Failed vehicle illegal operation rate</td><td>10% for buses and heavy trucks; 20% for light trucks and motorcycles</td><td><75%</td><td>1.1%-12.6%</td></tr>
<tr><td colspan="2">Vehicle population growth</td><td>10% decrease in average annual growth rate under the baseline scenario</td><td>0-30% decrease in average annual growth rate</td><td>N/A</td><td>10.1%-11.6%</td></tr>
</table>

Table 7.2 indicates that the level of PM_{10} emission reductions available from the I/M programs is the most sensitive to the variable ‘problem vehicle identification rate (IdenRate)’, since when increasing a variable from its lower limit to the upper limit while holding all the other variables constant, the greatest change happened with this variable (increased from 0.6% to 20.6%). Also, the uncertainty analysis demonstrates that the same variable ‘IdenRate’ contributes the greatest to variance. Moreover, the variables

associated with light-duty trucks play a relatively major role on the effectiveness of the I/M programs due to its role as the largest contribution to total PM_{10} emissions from motor vehicles. These findings suggest that program effectiveness can be improved by narrowing the uncertainty in the problem vehicle identification rate and by identifying a greater percentage of problem vehicles. Also, attention should be directed toward the light-duty diesel vehicle fleet in introducing the programs.

The sensitivity analysis performed here indicates that, in order to increase the problem vehicle identification rate, a key point is to improve testing procedure to maximize the ability of the programs to detect vehicles that need emission repairs. The current most commonly used smoke-opacity test has been found to be a poor test procedure that fails to discover most excessive PM emissions from motor vehicles (McCormick et al, 2003). Therefore, at the very minimum, the smoke opacity emission inspection must be replaced with a better PM emission test procedure in future I/M programs (Parsons, 2001). Second, studies have suggested that using more stringent testing cut-points can increase the percent of problem vehicles that are discovered by the inspection (Eisinger, 2005). Further research on how to maximize the problem vehicle identification rates associated with I/M programs is warranted.

7.1.4 Controlling emissions from vehicles in heavily-polluted developing areas: Key findings and policy implications

Although this study focuses on the single case of Bangkok, Thailand, a typical large, rapidly developing metropolitan area of a third-world nation, where population tends to concentrate and urban air pollution is severe, the key findings of this study can be generalized and applied to other developing regions. First, rapid expansion of vehicle use is a common feature in large metropolitan areas throughout southeast Asia (indeed in the megacities developing globally), suggesting that controlling emissions from motor vehicles, in particular fine particulate matter emissions, can potentially produce substantial benefits for those societies in terms of the improvement in public health. Decision makers in all of the less developed countries, not simply in Thailand, struggle with how to allocate their limited resources to tackle enormous problems that the societies are confronted with. This study suggests that controlling emissions in the transportation field should be prioritized since there is high confidence that the net benefits of the investment on pollution control will be positive and considerable. Second, given the fact that in developing nations, vehicles are often poorly maintained and there are still numbers of old vehicles running on roads, improving the general vehicle maintenance practice through policy measures such as the in-use vehicle I/M programs studied in this project is imperative in order to advance vehicle emission performance and protect public health in all of these metropolitan areas. This study suggests that with a high level confidence, new PM-oriented I/M programs will yield substantial health benefits that exceed the expenditures spent on the programs across a wide variety of similar metropolitan areas. Compared to traditional I/M programs, PM-related I/M programs are generally likely to be more beneficial in terms of reducing health damages, given the greater health risk posed by PM exposure. Therefore, urban areas with serious PM pollution attributable to transportation may consider launching such programs. Nevertheless, they must be carefully designed and implemented to maximize the emission reduction level as it is the key factor that determines the total net benefits available from the programs. Finally, the comprehensive health benefit analysis framework developed by this study can be applied to other developing regions to evaluate the social benefits associated with emission control strategies in the transportation sector, especially since the parameter values used here were developed largely from databases on metropolitan areas throughout southeast Asia, suggesting they may be broadly applied geographically.

7.2 Limitations and future research

(1) Air quality modeling

This study replied on a simplified model to predict ambient PM_{10} concentrations under different policy scenarios at limited sites where monitoring stations are located. Future research may consider using more sophisticated air dispersion models that take into account the climate situations and/or photochemical processes involved in the transport of ambient particulate matter (e.g. the AERMOD atmospheric dispersion modeling system and the Models-3/Community Multiscale Air Quality (CMAQ) modeling system). Air quality modeling is one of the critical steps in conducting heath risk assessment for air pollutants, although it requires substantial investment of time and money. Moreover, this study only considers the change in air quality within the metropolitan Bangkok area, a relatively small modeling domain of 7761.5 square kilometers. As the long-range transport of air pollutants can affect a much larger region, it is likely that the benefits of emission control are underestimated by excluding the impacts on air quality outside the metropolitan Bangkok area. Therefore, future research may consider a larger modeling domain surrounding the BMR.

In addition, currently there is significant uncertainty in the relative contribution to total PM_{10} emissions in the BMR by motor vehicles. Since it is one of the two variables that contribute the greatest to the total uncertainty in the health benefits estimates (its contribution to variance was 35%, see Table 6.4), further research is warranted to narrow the uncertainty in this variable.

(2) Assessing human exposure

This study assumes that people in the study area are exposed to the levels of PM_{10} pollution that are found at the permanent monitoring stations generally located in residential areas. As roadside air quality monitoring indicates that ambient PM_{10} concentrations in areas near traffic can be 2-3 times greater than the concentrations found in residential areas, if a large fraction of exposure is at the roadside pollution levels, the benefits of control emissions from vehicles may be underestimated in this study. Although there is little information about what fraction of exposure in the BMR is at the roadside levels, it is anticipated that being exposed to roadside pollution level is not a negligible fact in a developing metropolitan are like the BMR, given that air-conditioning is not prevalent and residents may spend longer time near traffic. A sensitivity analysis was performed in this study by assuming that 20% of total exposure is at the roadside levels of PM_{10}. This result indicated that assessing human exposure to traffic-related PM_{10} can be improved as more information on people's activity patterns becomes available. In addition, as discussed in the sensitivity analysis, the total health benefits can be estimated with higher confidence if more accurate demography becomes available.

(3) Concentration-response functions

This study relies on time-series studies on associations between short-term exposure to PM_{10} and the acute heath effects to estimate the number of health endpoints resulting from exposure to ambient PM pollution, excluding the chronic effects. Since scientific studies consistently observed significant associations between premature deaths and both of the short-term and the long-term exposure to PM pollution despite the existing uncertainty in their relationship and relative magnitude, the total benefits may be underestimated without taking into account the long-term effects. Although a sensitivity analysis is conducted in Chapter 6 by including the long-term effects abstracting from a cohort study conducted in the U.S., related studies in Thailand or other Asian developing countries are warranted to improve the precision of results.

Furthermore, this study used a stratified meta-analysis approach to estimate the source-specific PM_{10} health risk. However, the number of available studies for the meta-analysis is rather small (only 4-6 studies were included in each group), because earlier epidemiological studies generally did not differentiate the sources of PM_{10} in examining its adverse health effects. Furthermore, even among these limited studies that were included, there is only one study (the Harvard Six U.S. City Study) that strictly differentiated the effects of PM from different source categories, whereas all the other studies only reported the major sources of PM pollution. As the concerns over source-specific health risk of PM_{10} have grown in recent years, the generalizability of observational studies on the source-specific health effects can be improved when more studies that separate the effects associated with PM from different sources can be included in the meta-analysis.

(4) Projecting future emission factors

In general, emission performance of vehicles fleets will gradually improve as new vehicles with more reliable emission control technologies are purchased and old ones are scrapped, and the benefits of the I/M programs may be reduced due to the improvement in vehicle emission performance. However, there is still large uncertainty regarding the change in emission factors over time in developing countries. A study conducted in Beijing, on the contrary, found the fleet-average PM emission factor for heavy-duty diesel vehicles increased by approximately 2% from 1995 to 1998 (Wu et al, 2002). Due to the lack of sufficient empirical evidence in Asian countries, this study roughly assumes an identical 5% annual PM_{10} emission factor decrease rate for all the vehicle types (including City Bus, City Truck, Long Haul Truck/Bus, Light Duty Truck, Passenger Car and Motorcycle, see Figure 4.1) in the BMR. The limitation of making this assumption is that the emission factor temporal decrease rate can be different for different types of vehicles, mainly due to the distinction in the technologies used to improve emission performance. For example, for motorcycles, the Thai government is pushing the replacement of two-stroke motorcycles by four-stroke engines in order to improve the emission performance of motorcycles, whereas this technology change does not apply to the other vehicle categories. This suggests, at least on the surface, that the temporal decline in emissions from motorcycles (the surrogate used in the current analysis) may be larger than for the general vehicle fleet. Future emission factors can be projected more precisely when more empirical data, particularly the data of a single vehicle type, become available.

(5) Evaluating the costs of I/M programs

As discussed in Section 4.2.6, the total annual cost of the PM-oriented I/M programs in the BMR was estimated based on the assumption that the programs will achieve 25% emission reduction from motor vehicles. And since this estimate is considered to fall into the upper range of the possible social costs that the programs bear on the society, the net benefits calculated based on it are considered to be conservative estimations. Given that in reality, the variable cost component (inspection and repair costs) of an I/M program may affect the aggregate level of emission reductions delivered by the program, further research may focus on establishing a relationship between the cost and emission reduction, and incorporate the findings into the estimation of net benefits in order to better understand the net benefits of an I/M program as a function of the aggregate level of emission reduction available from the program.

(6) Evaluating the effectiveness of I/M programs

The main limitation of using the new analysis approach to evaluate the effectiveness of the I/M programs is that presently there is very limited information on the performance of the important program design elements in Thailand. The findings from running the 'I/M Design' spreadsheet could be improved when more empirical data in Thailand for the input variables are collected.

One source of emission reductions achieved by the I/M programs is improved maintenance of vehicles in anticipation of the required inspection process, but this kind of emission reduction relative to the baseline is not considered in estimating the emission reductions delivered by the programs. This portion of emission reductions can be large for diesel-fueled vehicles since pollution levels from these vehicles are heavily dependent on maintenance, perhaps resulting in an underestimate of the emission reduction benefits of the programs. Given the information gap, further research is warranted to examine the potential emission reductions due to improved maintenance by vehicle owners in anticipation of required I/M testing.

At present, the testing procedure for PM-related I/M programs is less mature as the in-use opacity test has been challenged for its ability to indicate the actual PM emission levels. For simplicity, this study assumed that PM emissions can be measured directly in future I/M programs. If technologies are not available for direct measurement of PM emissions in near future, estimates of the levels of emission reductions can be improved by further examining the correlations between the 'measured' emission levels and the actual levels.

Another issue is, similar as the problem of un-registered population, that there are a large number of 'un-registered' vehicles in the study area. For example, in 2005, 25% of the total number of registered vehicles in Thailand was registered in Bangkok, whereas it was estimated that 60% were actually used in Bangkok (Asian Development Bank, 2006). It would be difficult to predict the emission reduction benefits of I/M programs relative to the 'no I/M case' if a large fraction of vehicles are not regulated appropriately in terms of registrations. The government needs to address the un-registered vehicle issue while introducing an enhanced I/M system in the BMA, and the programs can be designed in a way that facilitates the identification of illegally-running vehicles. For instance, using windshield stickers as a visual indicator of having passed an I/M test is a method adopted by the I/M programs in many States in the U.S.

Lastly, further research needs to concentrate on the question that how to design and implement a PM-oriented I/M program in order to improve the performance of the key variables found in this study, e.g. the problem vehicle identification rate associated with an I/M program, so that more concrete and practical advice is provided to decision makers based on the theoretical conclusions of the present study.

(7) Uncertainty analysis

For some of the variables in both the health benefit analysis and the 'I/M Design' model, there is very little evidence in the literature for deriving a functional form for their PDFs. Given this fact, a PDF was selected based on the author's own judgment. Future uncertainty analyses can be improved if more evidence such as empirical data or expert judgment to justify the PDFs of the uncertainty variables becomes available. Or, in the case that such evidence is absent, sensitivity tests of overall uncertainty to the selected PDF forms can be conducted.

(8) Other additional research

On the basis of this study, further research could investigate the roles of controlling PM emissions from motor vehicles on reducing the levels of fine particulate matter ($PM_{2.5}$) in the BMR, as the government has started to collect ambient $PM_{2.5}$ data since 2002, and the country may consider introducing a national ambient $PM_{2.5}$ standard.

Further research could focus on the benefits, costs and uncertainty issues of introducing diesel retrofitting programs in combination with the PM-related I/M programs. In particular, an in-depth analysis of the effectiveness of retrofitting programs.

Finally, the single category of benefits of traffic-related PM control polices included in this study is health benefits. Future research may include other benefits resulting simultaneously from policy measures targeting PM emissions from motor vehicles, such as possible ancillary emission reductions in other pollutants and improved fuel economy, so that a more comprehensive evaluation of the benefits associated with motor vehicle emission control can be presented.

Appendix: Mathematical Equations in I/M Design Spreadsheet

(Equation (1) is developed by this study, Equation (2)-(6) are developed by Eisinger, 2005, and used in this study with minor modifications)

Equation (1) describes the percentage of all problem vehicles that are indentified by an I/M program. This variable is a function of the program participation rate and problem vehicle identification rate:

$$\text{ProbVeh} = \text{PartiRate} \times \text{IndenRate} \quad (1)$$

Where:

ProbVeh: Percent of all problem vehicles that are identified by an I/M program

PartiRate: Percent of total required vehicles that participate in an I/M program

IndenRate: Percent of problem vehicles inspected that fail the test

Equation (2) describes the percentage of problem vehicles that are both identified by I/M and subsequently undergo repair work:

$$\text{PercentRep} = \text{ProbVeh} \times [(1 - \text{ScrapFrac}) \times (1 - \text{VehWaive}) \times (1 - \text{IllegalVeh})] \quad (2)$$

Where:

PercentRep: Percent of all problem vehicles that are failed (identified) by I/M and subsequently repaired

ProbVeh: Fraction of problem vehicles (vehicles emitting above certification standards) identified by the I/M program test (this is a function of false pass rates)

ScrapFrac: Percent of failed vehicles that are retired from the fleet within one year of failing their I/M test

VehWaive: Fraction of problem vehicles (vehicles emitting above certification standards) identified by the I/M program test but allowed a waiver from needed repair work (a function of money spent on repairs)

IllegalVeh: Fraction of problem vehicles (vehicles emitting above certification standards) operating without obtaining the requisite repairs or certifications needed to pass or waive out of the I/M inspection process

Equation (3) describes the percent of a vehicle's total emissions that are reduced through repair work:

$$\text{PercentRed} = \text{GoodRep} \times \text{ExEm} \times \text{DurRep} \times \text{EmisFrac} \qquad (3)$$

Where:

PercentRed: Percent of total vehicle emission reductions achieved by repairs, for the vehicles failing I/M and getting repaired (does not include vehicles that fail I/M and are scrapped, waived, or illegally operating)

GoodRep: Fraction of repairs that are "good" (effective), as measured by percent of repaired vehicles that immediately pass a retest

ExEm: Fraction of excess emissions (where "excess" means emissions above allowable levels, usually referred to as the "cutpoint") reduced from identified problem vehicles that receive good repairs (I/M does not address all excess emissions, for example cold start emission problems)

DurRep: Durability of good repairs, as measured by percent of vehicles with good repairs that pass retests at 12 or 24 months

EmisFrac: Fraction of total vehicle emissions represented by pre-repair excess emissions (this is a function of the "cutpoint" used to define the point at which a vehicle is allowed to pass I/M; emissions above the passing cutpoint are considered excess). In other words, emissions below I/M cutpoints are essentially acceptable, emissions above cutpoints are excess; this variable represents the percent of total vehicle emissions considered excess.

Equation (4) describes the benefits of repair work after I/M test:

$$\text{BenefitsRep} = \text{PercentRep} \times \text{PercentRed} \quad (4)$$

Where:

BenefitsRep: The percentage of vehicles repaired (PercentRep), multiplied by the percentage reduction achieved per repair (PercentRed); units are in percent of total vehicle emissions reduced.

Equation (5) describes emission reduction benefits from vehicle retirements due to I/M test failures:

$$\text{BenefitsScrap} = \text{ProbVeh} \times \text{ScrapFrac} \times \text{ScrapEmis} \quad (5)$$

Where:

BenefitsScrap: Percent reductions from all problem vehicles, due to vehicles that are scrapped (considering emissions from the replacement vehicles)

ScrapEmis: Percent of total vehicle emissions reduced, for each vehicle retired from the fleet, after accounting for replacement vehicle emissions

Equation (6) describes total program benefits, in terms of the percent emission reduction in total vehicle emissions from the I/M program:

$$\text{BenefitsTotal} = \text{BenefitsGrow} + \text{BenefitsRep} + \text{BenefitsScrap} \quad (6)$$

Where:

BenefitsGrow: Percent emission reductions achieved due to the changes in vehicle population growth as a result of I/M enforcement. It is calculated by using the projected total number of vehicles under the baseline and the I/M scenarios, as well as the fleet-average emission rates. This variable is not in the original ‘I/M Design’ spreadsheet, but is developed by this study to reflect future I/M effectiveness as vehicle fleets change over time.

The units for BenefitsTotal are percent of total vehicle emissions reduced, for the entire problem vehicle fleet.

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