AIR QUALITY IMPROVEMENTS AND CARBON DIOXIDE REDUCTION POLICIES: A METHOD FOR ASSESSING SECONDARY BENEFITS

by

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ABSTRACT

A method is developed to assess the potential air quality improvements that result from changes in air pollutant emissions (SO_2, NO_x, CO, and VOC) associated with policies intended to reduce atmospheric carbon dioxide – so-called secondary benefits. A global emissions prediction and policy analysis model is integrated with an urban-scale atmospheric chemistry model to determine ambient pollutant concentrations. *Concentrations*, as opposed to emission rates used in previous research, are utilized as an indication of air quality changes. This method is applied to urban agglomerations or "megacities" in the United States. Results suggest that current claims for the offsetting effect of secondary benefits on initial implementation costs of CO_2 reduction policies for industrialized countries may not only be overstated, but overlook potentially adverse effects. Under certain climate change mitigation policies, secondary pollutants such as ozone and PAN may *increase* rather than decrease, damaging human and environmental health in addition to raising the compliance challenges faced by urban regions. The policy implications are that, given the potential for limited or inconsistent secondary benefits, flexibility in policy design is needed to address this uncertainty.

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CHAPTER 1

INTRODUCTION

Global climate change policy analysis focuses on the potential benefits and costs of limiting carbon dioxide (CO_2) emissions by reducing fossil fuel consumption. Combustion of fossil fuels, in addition to contributing 80 percent of global anthropogenic CO_2 emissions, also emits local and regional air pollutants such as sulfur dioxide (SO_2), nitrogen oxides (NO_x), hydrocarbons (VOC), carbon monoxide (CO), and particulate matter (PM). Climate change policy may thus have the additional benefit of improving public health and reducing ecosystem and material damage by improving regional air quality. Preliminary research (Ekins, 1996; Fankhauser, 1995) suggests that these secondary effects may contribute benefits of the same order of magnitude as CO_2 emission reductions.

Existing analyses addressing secondary benefits apply a top-down approach and determine the value of benefits from a country's aggregate emission reductions. However, potential air quality benefits vary considerably within a country between areas of high pollutant concentrations and those of low pollutant concentrations. In order to account for these variations, this thesis develops a methodology which focuses on "megacities", defined as areas of high population density, high emission rates per area, and high pollutant concentrations.

Figure 1.1 summarizes this methodology and outlines the inputs and tools applied to accomplish the stated objectives. Given a climate change policy, an economic model forecasts annual emissions for 12 geopolitical regions of the world. These regional emissions are then distributed to megacities within the region by assuming a static megacity-to-region emission ratio, which is established from a base year inventory. From annual emissions, daily emissions are characterized, and together with meteorological fields then are input into a reduced-form atmospheric chemistry model. For the given megac-

ity, this model approximates daily pollutant concentrations which provide an indication of the air quality. The differential in concentrations between a "policy" and "no policy" case allows the impacts on air pollution to be assessed. In addition, these concentration changes are input into epidemiological models to quantify the damages avoided to human health and ecosystems.

Figure 1.1   Methodology for determining the secondary effects of a CO_2 reduction policy.

Input	Tool	Objective
Climate Change Policy	Economic Model	Forecast Regional Emissions
Regional & Megacity Base Year Emissions	Maintain Region to Megacity Ratio	Distribute Emissions to Megacities
Daily Megacity Emissions & Meteorology	Atmospheric Chemistry Model	Determine Ambient Pollutant Concentrations
Change in Ambient Concentrations	Epidemiological Literature	Quantify Human Health Impacts

1.1   Climate Change

The benefits of CO_2 reductions are highly uncertain as are the costs of mitigation and adaptation. However, summaries of scientific evidence linking anthropogenic emissions and global warming do exist (IPCC,1995). This warming trend, if human activities continue unaltered, could translate into a possible rise in the global mean temperature of 1 to 5 degrees Celsius by the year 2100. Such a temperature change may be accompanied by changes in precipitation patterns, sea level rise, and storm frequency and intensity.

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The threat posed by these potential changes has led virtually every nation to sign the United Nations Framework Convention on Climate Change (FCCC) which has a stated goal of stabilizing concentrations of greenhouse gases "at a level that would prevent dangerous anthropogenic interference with the climate system."[1]

However, economic and political challenges resulting from the complex scientific characteristics of global warming have complicated the implementation of global or national policies. The first of these difficulties is the long time horizon of climate change. Action to curb climate change must begin decades before results will be achieved, because CO_2 molecules in the atmosphere have a lifetime of between 50-200 years. The impact of climate change will not affect today's generation, but the initial mitigation costs will. This trade-off between limiting resource use today and the potential of preserving the environment for the future is prioritized differently, depending on the perspective from which individuals or countries view the problem. For example, a number of developing countries perceive climate change as a problem of the industrialized world, as these countries have emitted the highest proportion of CO_2 over the years (Porter and Brown, 1991). In addition, devoting resources to the possibility of benefits for future generations when current populations are suffering from immediate social and environmental problems is not practicable for a number of countries (Jung, 1996; Parikh, 1992).

The second challenge inherent in the science of climate change is the necessity for global commitments. Countries cannot effectively act alone to reduce the risk of climate change because CO_2 molecules are evenly distributed throughout the atmosphere regardless of their geographic source. If one country implements a CO_2 control policy, its initiatives may be offset by increases in CO_2 emissions in another part of the world. Scenario analyses have shown that industrialized countries, even if they eliminate CO_2 emissions, could not mitigate the impact of climate change without the participation of developing countries (Jacoby et al., 1996).

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[1] United Nations Framework Convention on Climate Change, 1992.

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1.2 Secondary Benefits in Policy Decision Making

In the face of the substantial uncertainty, the long-term benefits and short-term costs, and the need for global commitments, secondary benefits can play an important role in climate change policy decisions. If climate change is mitigated by actions to curb energy use, it has been suggested that near-term benefits would accrue largely in the country where the constraints were instituted and offset a portion of the implementation costs (Ekins,1996; Burtraw, 1996; IPCC, 1995). Benefits of this character may provide additional justification for individual action. Potentially, these secondary benefits may solicit commitments by developing countries in global negotiations.

These secondary benefits, such as air and water quality improvement, may provide a buffer against the initial cost of climate change measures. Some propose (Barker, 1993; Ekins, 1996) that the prospect of these benefits outweighing the costs offers a potential "no regrets" policy option. Regardless of the actual threat of global warming, it is possible that the resources will not have been applied in vain. However, this is not to suggest that climate change policy is an economically efficient means to achieve desired levels of air quality.

The prospect of improving air and water quality in the short-term may provide an incentive for a commitment from industrialized and developing countries. Limited resources and a host of local environmental problems are two reasons why developing countries have chosen not to commit to climate change policy (Jung, 1996). However, if climate change policy can offer benefits of improved local air quality, which currently poses an increasing threat to the health of urban inhabitants, developing countries may have reason to reconsider. For example, the access to improved resources and technologies available through joint implementation, or technology transfer under international climate change agreements, may assist developing countries in addressing short-term environmental risks.

While the existence of secondary benefits may provide incentive for countries to act on climate change, identifying the nature of these benefits will also inform the details of na-

tional policy decisions. To identify the appropriate policy instrument to implement on a national level, consideration of all the costs and benefits is necessary. For example, the secondary benefits of a carbon tax will differ from those of a energy (BTU) tax. A carbon tax effects the use of fuels with high carbon content, such as stationary source coal combustion; whereas, an energy tax, effects the use of fuels with high energy content, such as petrol-fueled mobile sources. Given that, in urban areas, a main source of pollution is petrol-fueled vehicles, an energy tax may potentially have a greater effect on air quality than a carbon tax (Sheraga and Leary, 1993). These secondary air pollution effects should be incorporated into global climate change policy analysis.

1.3 Existing Approaches to Quantify Secondary Benefits

Approaches to climate change mitigation can be divided into two categories: sequestering carbon and reducing emissions. Carbon sequestration includes options such as increasing forest land and CO_2 removal and disposal technologies[2]. These technologies are currently economically unfeasible, and attention has focused on reducing emissions by changing to less carbon-intensive fuels or reducing energy consumption. Conservation, end-use efficiency improvement, changing consumption patterns and reducing economic and population growth are ways to limit energy consumption.

Limiting CO_2 by reducing fossil fuel combustion may also lead to decreases in other pollutant emissions. This activity accounts for 80% of CO_2, 83% of SO_2, 67% of NO_x, and 31% of CO emissions on a global scale (Liu, 1994). However, it is difficult to quantify the effect that these reductions will have on the environment. Simply assigning a monetary value to the emission reductions is not adequate because the pollutants emitted undergo numerous processes before they impact human health or ecosystems. Figure 1.2 presents the phases and factors which lead from human activity to pollutant emissions and formation to exposure and finally to adverse effects. These phases are related in complicated ways, and numerous factors shift these relationships and create non-linearities.

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[2] Carbon dioxide removal processes will not be discussed in this thesis. For a discussion of the options see Eckaus *et al.* (1996)

Figure 1.2 The phases and factors of pollution formation which lead to adverse effects.
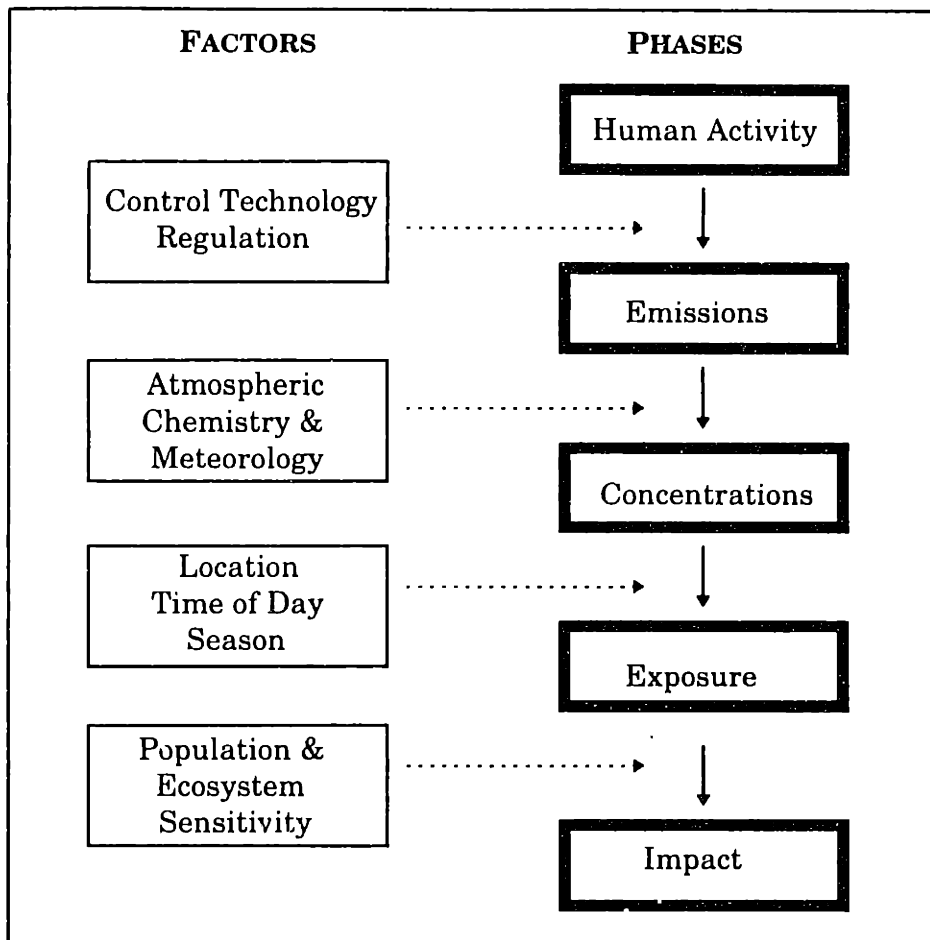
For example, the amount of emissions produced from human activity depends on the process, the technology used, the level of regulation enforced, etc.

Investigations have attempted to quantify the level of these secondary benefits by proceeding directly from emission level to impact assessment and neglecting these intermediary steps. Table 1.1 provides a non-exhaustive list of these assessments, the region of analysis and the method used to value the damages avoided. These preliminary studies have focused on industrialized nations, namely the U.S., the U.K., Norway, and Germany. With the exception of the ICF (1995) model, regional resolution is not provided. Therefore, these studies must quantify the benefits based on aggregate emission reductions. In doing so, they typically apply a damage cost to the emissions or fuel. For

Table 1.1 Secondary Benefit Assessments

MODEL	GEOGRAPHIC RESOLUTION	VALUATION METHOD
Alfsen (1992)	Norway	$/ton of emission
Ayres and Walter (1991)	U.S., Germany	Elasticity factor
Barker (1993)	U.S., U.K., Norway	$/ton of emission
Battelle (1993)	U.S.	Not modeled
Boyd et al. (1995)	U.S.	$/ton of fuel [1]
Carnegie Mellon Univ. (1993)	U.S.	Not modeled
Goulder (1993)	U.S.	$/ton emission [2]
ICF (1995)	10 Regions in U.S.	Transport Model [1,3]
Jorgenson et al. (1995)	U.S.	$/ton of fuel [1]
Pearce (1992)	U.K., Norway	$/ton of emission

[1] Valuation includes effects of secondary pollutants.
[2] Modeled by Sheraga and Leary (1993)
[3] Modeled by Burtraw et al. (1996) for NOx only.

example, based on the cost to reduce emissions or the cost of possible damages avoided, the emissions of each ton of pollutant are assigned a dollar value. Because regional resolution is provided in the ICF model, a more detailed analysis of the damage avoided (a chemistry transport model) is used.

Geographic Resolution

A primary limitation of these models is their lack of geographic resolution which in turn constrains the method of damage valuation and disallows an explicit account of the adverse effects considered. The geographic resolution provided by these models, with the exception of the ICF model, is restricted to a national level. This aggregation prohibits the modeling of local conditions including meteorology, emission profiles, and exposed populations. These local characteristics impact the level of achievable benefits.

Depending on the existing local conditions, geographic regions will benefit to different degrees from a reduction of air pollutants. For example, Los Angeles experiences high

ozone concentrations but has relatively low concentrations of SO_2. In contrast, Beijing experiences high concentrations of SO_2 but has low concentrations of ozone. Therefore, reducing ozone concentrations would provide benefits to Los Angeles and not to Beijing, whereas SO_2 reductions would provide benefits to Beijing and not to Los Angeles. In general, the potential benefits depend on the level of existing concentrations and are specific to the pollutant emissions reduced.

In a preliminary study, Burtraw *et al.* (1996) extends the ICF model to analyze the importance of regional disaggregation. The ICF projections for NO_x reductions were used as inputs into a reduced form atmospheric transport model linked to health valuation. The model valued health benefits from changes in NO_x concentrations and secondary nitrate concentrations, but it did not incorporate the contribution of NO_x to ozone formation, visibility impairment, and other environmental impacts of nitrogen deposition. Depending on the initial characteristics of a region, the level of benefits varies from emission reductions. This preliminary investigation demonstrates the disconnect between emission reductions and benefits.

Valuation of Benefits

This lack of geographic resolution has resulted in highly simplified valuation schemes. The method used by these studies to evaluate damages may misrepresent the actual level of achievable benefits. By assigning a dollar value to emissions or fuel use rates, these studies essentially attempt to represent the baseline air quality, atmospheric chemistry and physics, resulting concentration levels, population exposed, human health and ecosystem responses with a single metric. The underlying assumption in this valuation model is that fuel use and/or emissions are linearly related to the level of environmental impact. As discussed previously, factors such as atmospheric reactions counter this assumption. Therefore, the impact of this assumption on the results must be examined before the conclusions based on these studies can be used.

The valuation method employed by these studies yields estimates based on either a per-emission dollar value, a per-fuel-dollar value, or an elasticity factor. An elasticity factor assumes that a percentage change in one variable will result in a given percentage change in another variable.

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- Scheraga and Leary (1993), using the Goulder (1993) model, quantify the damages using a range of estimates of dollar value of benefits per ton of emission reduction. The dollar per ton benefit estimates were taken from a variety of sources incorporating health, crop and material damage and visibility. It is only in this indirect way that secondary pollutants have been incorporated in any of the existing studies. The estimates range from $360 to $2,400 per metric ton of VOCs, $300 to $1,800 per metric ton of SO_x, $430 to $10,900 for particulates, and $10 to $100 for NO_x.

- Jorgenson *et al.*(1995) and Boyd *et al.* (1995) quantify the damages using a dollar value of benefits per ton of fuel reduced. This value incorporates the externalities of both primary and secondary pollutants.

- A third method of valuation is employed by Ayres and Walter (1991). They assign relative weighting factors to each pollutant to account for difference in toxicity. These calibrated emissions are then aggregated and used to determine benefits by multiplying by a -0.05 elasticity of mortality to pollution reduction.

These preliminary valuations result in claims of high potential air quality benefits from CO_2 reduction policies. For example, using the Goulder (1993) model, Sheraga and Leary (1993) impose a tax to achieve 1990 CO_2 concentration in 2000. In the case of a carbon tax, the secondary benefits range from $300 million to $3 billion. In the case of a BTU tax, these benefits range from $500 million to over $4 billion.

While recognizing the limitations, these preliminary studies have proposed that these assessments provide a relative order of magnitude of the potential achievable secondary benefits. However, if these arguments are to be upheld, rigorous attempts to verify the governing assumptions are necessary. This thesis argues that geographic resolution and the accompanying emissions profiles, atmospheric conditions, and populations are necessary to understand the impact of climate policies on air quality, which is by definition a local condition. A method to incorporate these considerations is developed.

1.4   Thesis Outline

This thesis addresses the limitations of the existing secondary benefit studies by developing a method which examines air quality impacts from climate change policy based on pollutant concentrations. Figure 1.3 illustrates the structure of the modeling system applied to this task. As described previously, driving damage evaluation by emission rates alone is inadequate and may misrepresent the air quality effect of such policies.

Pollutant emission rates, atmospheric chemistry and meteorology determine the pollutant concentrations in ambient air for a given region at a given time. In addition to forming air pollutants, these relationships impact the production and transport of greenhouse gases to the global atmosphere. These relationships are discussed in Chapter 2 along with a presentation of the atmospheric chemistry model that is used in this study.

Figure 1.3   Development of an Improved Method
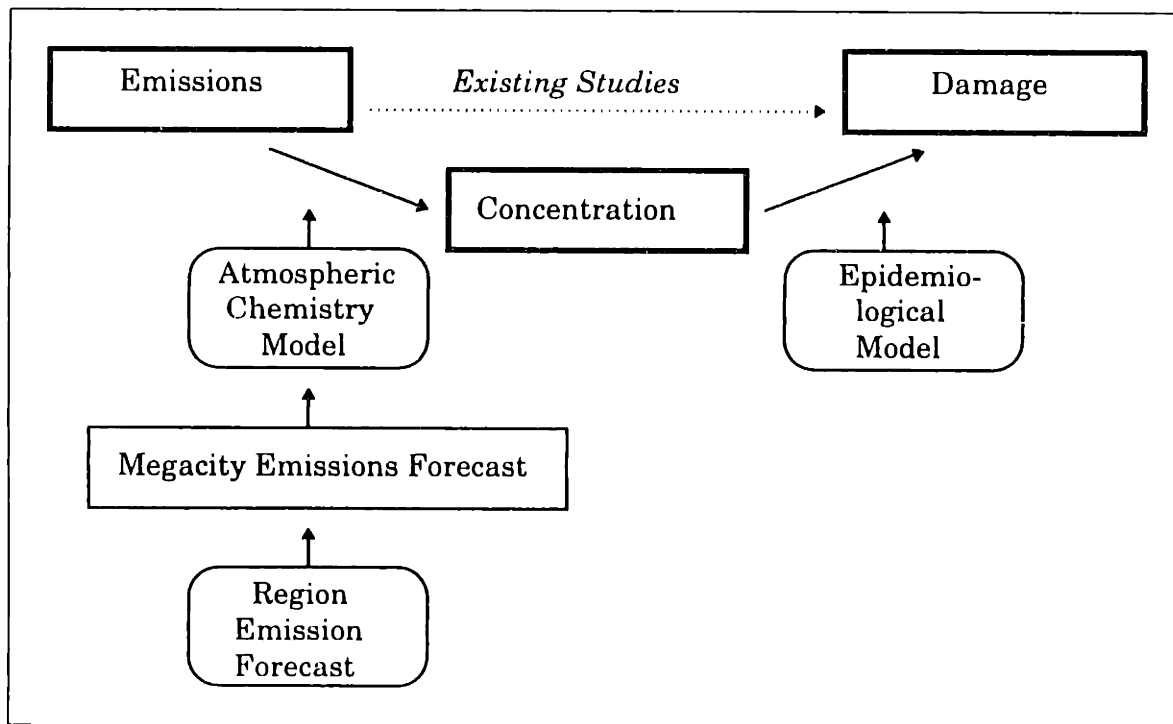
This atmospheric chemistry model operates on an urban scale and requires megacities and their emissions to be defined. As the purpose of this research is to determine the impact of climate policy, it is not enough to establish a current inventory of emissions for different megacities. Rather these emissions must be forecast into the future under different policy scenarios. In order to accomplish this task, an economic and policy analysis model is applied to predict energy use, sector activity levels, and CO_2 emissions. However, the these regional emission forecasts are on a national and multi-national level. Therefore, a method to disaggregate these national emissions into megacities is needed. The methods and models used for forecasting and distributing emissions are discussed in Chapter 3.

Once these emission forecasts are transformed into concentrations using the chemistry model, damages can be assessed. This study demonstrates an evaluation of human health impacts of pollution and can be extended to include ecosystem damages. The existing epidemiological techniques and models for linking concentrations and impacts are discussed in Chapter 4.

This method is applied to the United States, and findings are presented in Chapter 5. The air pollutants considered are SO_2, NO_x, VOC, CO, photochemical oxidants and acid deposition. Particulate matter, while a very serious health concern, is not modeled. Over 90% of particulate matter emissions in the United States are from miscellaneous sources such as agriculture and forestry, wildfires, wind erosion, unpaved and paved roads[3]. In addition, a portion of particulate matter is sulfate aerosols, a result of SO_2 emissions reacting in the atmosphere. However, the percentages of particulate matter attributed to these emissions is variable between regions and highly uncertain.[4]

In conclusion, this thesis discusses the policy implications of the findings. Next steps in the research of secondary benefits and possible approaches to improve upon this meth-

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[3] The source mix varies between countries. For example, a large percentage of particulate matter emissions in China come from coal combustion. Therefore, fossil fuel reduction may have considerable health benefits related to PM reductions.

[4] As discussed previously, existing literature accounts for PM effects in a single unit valuation thereby avoiding this problem. Studies focusing on air pollution and its costs have attempted a more detailed analysis of PM emissions (Small and Kazimi, 1995).

odology are discussed. This necessarily brings forth the current limitations of the models. The method was developed with the intent to examine megacities around the world. However, a number of assumptions apply specifically to industrialized countries. Using this analysis framework, further research can develop assumptions specific to the megacities of developing countries.

CHAPTER 2

LINKING EMISSIONS TO URBAN AIR QUALITY

The chemistry of urban regions and of global scale climate are linked in three important ways: through the chemistry of the atmosphere, through the policy of CO_2 emission control, and through air pollution control measures. This thesis addresses the second of these links, the effects climate change policy has on urban air quality. Due to high pollutant concentrations and species variety in urban areas, an aging process, or the production of secondary pollutants from primary emissions, occurs within an urban plume. These reactions create a non-linear relationship between emissions and concentrations. For this reason, in the investigation of the secondary benefits, pollutant emissions are not an adequate indicator of air quality levels. Pollutant concentrations, both primary and secondary, must be investigated to accurately portray the resulting ambient conditions.

This chapter introduces the reduced-form chemistry model that is used in this study to model pollutant concentrations from emissions in an urban plume. Before examining the details of this model, the science governing the relationship between air chemistry and the global climate and the formation of air pollutants and greenhouse gases is presented.

2.1 The Greenhouse Gases and Global Climate

The greenhouse effect is the process by which a portion of the sun's radiation is trapped in the atmosphere by naturally occurring gases, referred to as greenhouse gases, warming the atmosphere. These greenhouse gases include water vapor, carbon dioxide, meth-

ane, nitrous oxide, CFCs[5], and ozone. The sun emits short-wave radiation, approximately 70 percent of which is absorbed by the earth's atmosphere with the remaining 30 percent reflected into space. In turn, the earth emits long-wave or infrared radiation into the atmosphere. A portion of this out-going terrestrial radiation is emitted into space and a portion is absorbed and re-radiated to the earth by the greenhouse gases.

The balance between the solar energy absorbed by the earth and the amount of energy radiated back into space is determined by the amount of greenhouse gases in the atmosphere. Since pre-industrial times, the sources of greenhouse gases, for example fossil fuel combustion, have increased while the sinks of the greenhouse gases, for example, forests, have decreased. This has led to an increase in carbon dioxide, methane, and nitrous oxide concentrations of 30 per cent, 145 per cent, and 15 per cent respectively (IPCC, 1995). This increase has heightened concerts about a possible anthropogenic increase in the greenhouse effect, with global warming as its consequence.

The capacity of a greenhouse gas to warm the atmosphere depends on three characteristics:

- its atmospheric concentrations, the number of molecules in the atmosphere
- its instantaneous radiative forcing power, the ability of the gas to absorb out-going infrared radiation and reradiate
- its atmospheric residence time, the lifetime of the molecule which is a function of its chemical properties, its sources, and its sinks.

Table 2.1 provides a summary of these characteristics for anthropogenic greenhouse gases, and for gases which indirectly contribute to the formation of greenhouse gases. When assessing the potential of a gas to warm the atmosphere, these characteristics must be considered concurrently.

The presence of these greenhouse gases in the atmosphere, with the exception of aerosols, contributes to the warming of the atmosphere by increasing radiative forcing. In

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[5] Unlike the other greenhouse gases, there are no biogenic sources of CFC emissions.

contrast, aerosol[6] particles in the atmosphere and in clouds scatter solar radiation back into space causing a cooling effect.

However, concluding that the negative radiative forcing counteracts the positive radiative forcing of the other gases is not entirely accurate. Due to high regional variability in aerosol concentrations within the atmosphere[7], the negative radiative forcing of these particles occurs in some regions and not in others. Therefore, globally averaged radiative forcing as an indicator of potential climate change is limited and must be used with caution (IPCC, 1995).

Table 2.1 Characteristics of Greenhouse Gases

Trace Gas	Average 1994 Concentration	Approximate Lifetime	Change in Direct Radiative Forcing from 1950-1992
CO_2	358	50-200 years	1.56
CH_4	1.72	12 years	0.47
N_2O	0.312	120 years	.14
CFC-11	268	50 years	0.25
O_3	10^{-2}-10^{-1}	variable	0.4
SO_2	10^{-5}-10^{-4}	40 days	-0.4 (Aerosols)
NO_x	10^{-6}-10^{-2}	1 day	N/A
CO	0.05-0.2	65 days	N/A

Sources: IPCC, 1995, Seinfeld, 1986

Viewing all the properties of a gas concurrently, one can avoid potentially misleading interpretations. For example, the lifetime, or atmospheric residence time of the pollutant, is indicative of its regional variability or how well-distributed the pollutant is in the atmosphere. The terms "flow" and "stock" are often used to distinguish between pollut-

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[6] Aerosols impact the climate in an indirect way also by changing the optical properties of clouds and influencing the cloud formation process.

ants that are not well mixed and those that are well mixed respectively. Hence, the distributional qualities of aerosols and ozone can be accounted for by the atmospheric residence time. In addition, the duration of the pollutant in the atmosphere is related to the time necessary to decrease atmospheric concentrations. On a elementary level, controlling emissions of flow pollutants today can result in a decrease in concentrations today[8], whereas limiting stock pollutants today will not effect concentration levels for years or decades.

While the distinction between greenhouse gases and air pollutants exists, Table 2.1 highlights the overlap between the gases. Throughout this thesis, air pollutants refer to gases which have consequences for the local and regional communities, specifically, NO_x, CO, SO_2, VOCs, and ozone. Greenhouse gases refer to the gases which directly contribute to increased changes in the radiative forcing of the atmosphere, specifically, CO_2, NO_2, CH_4, ozone and aerosols.

Air pollutants are of concern in the troposphere, whereas climate change variables are of concern in both the troposphere and the stratosphere. The troposphere is the layer of the atmosphere extending from the ground up to 15 km at the equator and 10 km at the poies. The stratosphere extends from the tropopause, the top of the troposphere, to 50 km. Temperature decreases with height in the troposphere keeping the air relatively well mixed, while the temperature is relatively constant in the lower stratosphere. In the upper stratosphere, temperatures increases with altitude due to the absorption of solar radiation from ozone.

A distinction must be made between stratospheric and tropospheric ozone. Stratospheric ozone limits the amount of solar radiation and protects the earth. Tropospheric ozone, however, is a greenhouse gas and a major component of photochemical air pollution, an increasing problem in urban environments. Throughout this paper, ozone refers to tropospheric ozone.

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[8] While this may be the case for a number of pollutants, it is not without exception. As was discussed previously, a linear relationship does not exist between emissions and concentration. In particular, this non-linearity can be seen in the production of ozone. A decrease in its precursor emissions does not necessarily yield a decrease in its concentration.

2.2 The Chemistry and Physics of the Troposphere

The atmosphere acts as the medium for chemical and physical reactions of primary pollutant emissions which produce secondary pollutants. For example, Figure 2.1 shows the reactions which occur from nitrogen emissions entering the atmosphere. Secondary pollutants formed through reactions with water vapor or the hydroxyl radical (OH) include photochemical oxidants (O_3) and acid aerosols (HNO_3, NO_3). These reactions decrease the primary pollutants transported from the urban plum into the global atmosphere while increasing the flux of secondary pollutants.

Figure 2.1 The Fate of Nitrogen Emissions in the Troposphere
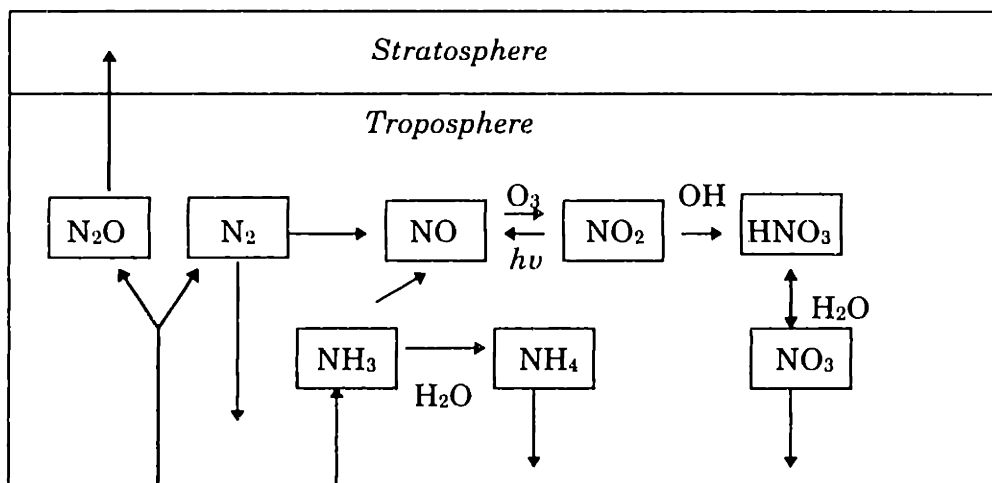
Source: Seinfeld, 1986

High initial concentrations of pollutants and high pollutant emission rates increase the number of chemical reactions in the troposphere, and promote high pollutant concentrations. Hence, it is important to understand the chemistry and physics which drive chemical reactions in the atmosphere. The following section provides a simplified summary of the chemistry that occurs on a local and regional scale[9]. The hydroxyl radical (OH) is a critical element in determining the chemical composition and oxidizing capacity of the troposphere. The primary source of OH in the atmosphere is the reaction between a singlet oxygen atom ($O(^1D)$), which is produced by the photolysis of ozone, with water vapor:

$$O_3 + h\upsilon \rightarrow O_2 + O(^1D)$$
$$O(^1D) + H_2O \rightarrow OH + OH$$
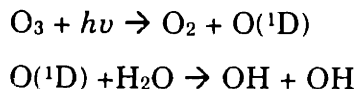
The levels of OH are impacted by the level of other gases in the atmosphere and OH acts as a critical element in determining the level of air pollutants and greenhouse gases (Bouwman, 1990; Guenther $et.$ al, 1994). For example, OH decomposes CH_4, CO, NO_x, and SO_x through reactions:

$$CH_4 + OH \rightarrow CH_3 + H_2O \rightarrow \text{multiple steps} \rightarrow CO$$
$$CO + OH \rightarrow CO_2 + H$$
$$NO_2 + OH \rightarrow HNO_3$$
$$SO_2 + OH \rightarrow HSO_3$$
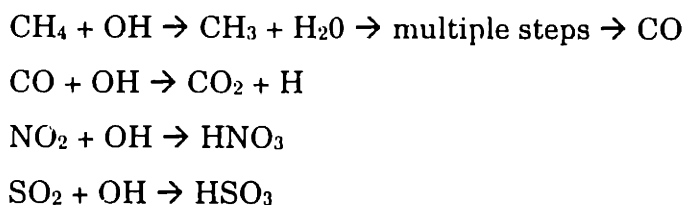
Increasing concentrations of OH decrease the lifetime of CH_4, produce secondary CO_2, and contribute to the formation of acid aerosols. These reactions also supply other radicals to the atmosphere. For example, the hydrogen atom (H) formed from the reaction between CO and OH reacts quickly with the oxygen molecule (O_2) to form a hydroperoxy radical (HO_2). This radical is critical in the NO_x air system as it allows the regeneration of OH without consuming NO.

$$H + O_2 \rightarrow HO_2$$
$$HO_2 + NO \rightarrow OH + NO_2$$
$$NO_2 + h\upsilon \rightarrow NO + O$$
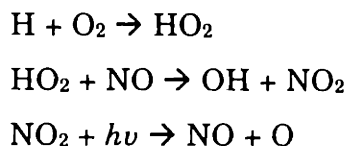
These reactions demonstrate the multitude of dependencies and interactions of chemical species within the atmosphere. The connection between the troposphere, air pollutants, and greenhouse gases becomes apparent.

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[9] Local scale is typically defined as a 10^2 km area with reaction rates of one day; regional scale is 10^3 km and a few days; global scale is 10^4 km and 30 years)

2.2.1 Air Pollution Formation

The formation of acid aerosols, methane, and ozone result from chemical reactions in the atmosphere as a function of pollutant emissions. Because these gases effect the level of radiative forcing and hence the global climate, air pollution can no longer be considered only a local or regional problem. The difference in the concentrations of clean and polluted troposphere are outlined in Table 2.2.

Table 2.2 Air Pollutant Concentrations in the Troposphere

POLLUTANT	CLEAN TROPOSPHERE (PPB)	POLLUTED AIR (PPB)
SO_2	1 - 10	20 - 200
CO	120	1000 - 10000
NO	0.01 - 0.05	50 - 750
NO_2	0.1 - 0.5	50 - 250
O_3	20 - 80	100 - 500
VOC	varies	500-1200

Adapted from Seinfeld (1986), p. 37

Secondary pollutants are pollutants which are not directly emitted into the atmosphere. Rather they are created in the atmosphere by the reactions of other emissions. The primary emissions that contribute to the formation of the these secondary species are referred to as precursor emissions. Two such secondary pollutants that have local and regional consequences are photochemical smog and acid rain. The formation of these pollutants in the troposphere are discussed in the following section.

Formation of Photochemical Oxidants

Ozone is a main component of photochemical smog, a significant contributor to urban air pollution. The two primary precursors to ozone formation are NO_x[10] and VOC emissions.

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[10] NO_x emissions are typically 95% NO and 5% NO_2, but are often reported in NO_2.

VOCs are highly reactive organic compounds, also referred to as non-methane hydrocarbons (NMHC). Ozone is formed as a result of the NO_2 reacting with sunlight:

$$NO_2 + h\upsilon \rightarrow NO + O$$
$$O + O_2 + M \rightarrow O_3 + M$$
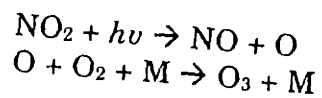
where M represents N_2 or O_2 or another third molecule that absorbs the excess energy and stabilizes the O_3 formation. Considering only this reaction, ozone concentrations are stabilized at low levels because the ozone generated reacts rapidly with the NO in the atmosphere regenerating NO_2:

$$O_3 + NO \rightarrow NO_2 + O_2$$
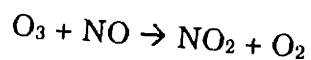
In the absence of other chemical species, this process reaches a steady state. This steady state depends on the rate, k, in which NO is converted to NO_2:

$$[O_3] = k[NO]/[NO_2]$$

where the brackets indicate concentration levels.

However, other chemicals are present in the atmosphere, specifically VOCs. The VOCs react with the hydroxyl radical (OH) to form free radicals, some of which generate additional free radicals by reacting with NO and NO_2. During the lifetimes of these free radicals, many molecules of NO can be converted to NO_2. Essentially, one radical formed as a result of the reaction between OH and VOC will participate in several NO to NO_2 conversions before extinction. This process converts NO to NO_2 without the consumption of ozone, and ozone is accumulated[11] (Seinfeld, 1986).

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[11] In the following reaction of VOCs and the hydroxyl radical, there are three NO to NO_2 conversions without the consumption of ozone:

$$RH + OH \rightarrow RO_2 + H_2O$$
$$RCHO + OH \rightarrow RC(O)O_2 + H_2O$$
$$RCHO + h\upsilon \rightarrow RO_2 + HO_2 + CO$$
$$HO_2 + NO \rightarrow NO_2 + OH \quad (1)$$
$$RO_2 + NO \rightarrow NO_2 + RCHO + HO_2 \quad (2)$$
$$RC(O))_2 + NO \rightarrow NO_2 + RO_2 + CO_2 \quad (3)$$
$$OH + NO_2 \rightarrow HNO_3$$
$$RC(O)O_2 + NO_2 \rightarrow RC(O)O_2NO_2$$
$$RC(O)O_2NO_2 \rightarrow RC(O)O_2 + NO_2$$
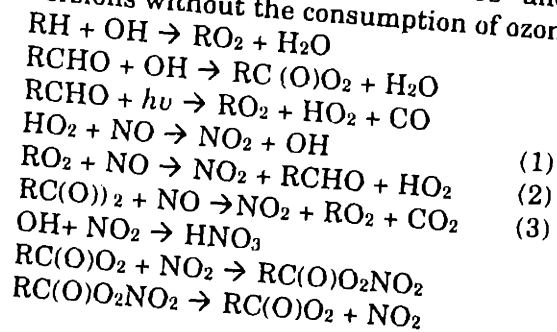
Due to the chemistry described above, ozone formation is limited by the ability of the troposphere to convert NO to NO_2. Two situations in which ozone is not accumulated is either at low VOC/NO_x ratios or high VOC/NO_x ratios. At low VOC/NO_x ratios, on the order of 1-2, ozone formation is limited by VOCs because enough free radicals are not generated to convert NO to NO_2 efficiently. At very high VOC/NO_x ratios, on the order of 20 or more, the excess of VOCs either react with and consume ozone or they react with NO_2 removing it and its ability to produce ozone (Milford et al., 1989).

These chemical features of ozone formation can be represented by viewing isopleths of maximum ozone concentrations for given combinations of NO_x and VOC concentrations (Figure 2.3). The general features of this plot are duplicated in nearly all photochemical smog systems, but the concentration levels of the ozone isopleths vary with the specific meteorological, topographical and emission conditions (Milford et al., 1989). Depending upon the combination of NO_x and VOCs in the atmosphere, reducing NO_x alone or VOCs alone may either decrease or increase the levels of ozone.

Figure 2.3 Ozone Isopleths
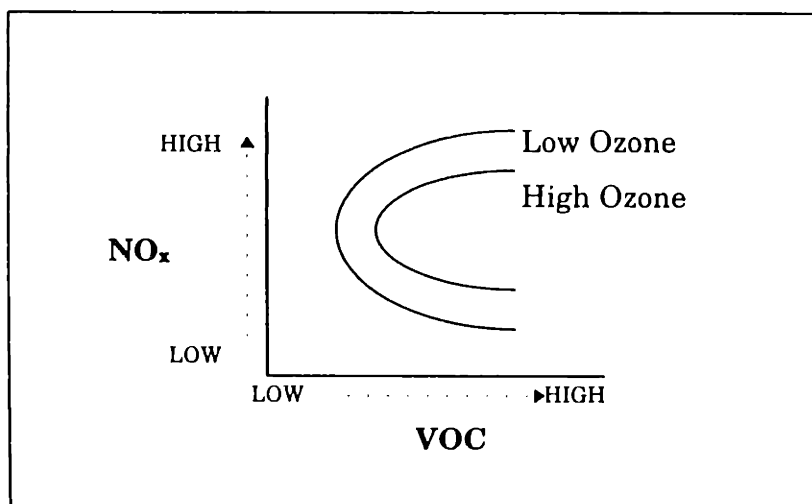
In addition to its role in ozone formation, NO_x acts as a catalyst for a number of other reactions which produce pollutants such as peroxyacetyl nitrate (PAN), and nitrate aerosols (HNO_3 ,NO_3, and N_2O_5). PAN is the product of the decomposition of VOCs through a series of reactions involving NO and NO_2. PAN functions as a transport and storage mechanism for NO_x. Due to its thermal sensitivity, PAN decomposes into NO at high

temperatures; therefore, typically during the evening, PAN is transported from one region to the next. In the morning, under the sun's radiation, PAN heats up and decomposes, unloading NO into the atmosphere. These high initial concentrations of NO make the production of ozone possible regardless of emission rates. This process contributes to high ozone levels in areas downwind of urban areas.

Formation of Acid Aerosols

The formation of sulfuric or nitric acid aerosols have local, regional, and global health and environmental consequences. Deposited dry or wet, these particles contribute to human health problems, damage lakes and vegetation, and corrode building material. Aerosols in the atmosphere and clouds have a negative effect on the radiative balance, cooling the atmosphere.

Acidic compounds are formed when the emissions of sulfur dioxide and nitrogen oxides react with the hydroxyl radical (OH). OH reacts with sulfur dioxide emitted in the atmosphere to form sulfuric acid (H_2SO_4) in the following manner:

$$SO_2 + OH \rightarrow H\,SO_3$$
$$H\,SO_3 + O_3 \rightarrow HO_2 + SO_3$$
$$SO_3 + H_2\,O \rightarrow H_2\,SO_4$$
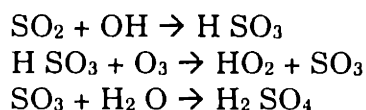
Similarly, OH reacts with NO_2 to produce nitric acid (HNO_3):

$$NO_2 + OH \rightarrow HNO_3$$
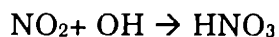
Because these reactions depend on the OH levels which are affected by ozone levels, the concentrations are linked in a complex manner to the level of other pollutants in the atmosphere.

2.2.2 Meteorology

In addition to the chemistry, atmospheric physics or meteorology plays a significant role in pollution formation, and its influence is expected to change with climate change. The production of secondary pollutants, which may have local, regional, and global implications, may be aggravated by climate change variables such as changes in wind speeds

and patterns, circulation, and increases in regional temperatures (Smith and Tirpak, 1991). These factors may change the nature and duration of air pollution episodes.

- **Temperature**: Air temperature impacts the rates of the photochemical reactions. These rates increase[12] at higher temperatures. Depending on the ratio of VOC to NO_x, ozone accumulation may increase with increasing temperatures (Smith and Tirpak, 1990).

- **Mixing layer height**: This parameter refers to the distance between the ground and the first temperature change, or upper layer inversion. This temperature inversion decreases vertical mixing and in doing so traps pollutants. Mixing layer height and pollution concentration levels are inversely related, the lower the mixing layer height the higher the pollution concentrations.

- **Wind speed**: The speed of the wind determines the rate of transport of the pollutants out of an air shed. The faster the wind, the more diluted the pollutants are over the region and the lower the concentrations are.

The mixing layer height and wind speed are a function of high and low pressure systems which change with the global circulation. High pressure systems characteristically have a low mixing layer height, low wind speed, and less cloud cover, producing conditions ideal for high pollutant concentrations. In contrast, low pressure systems provide good ventilation with high mixing layer height, high wind speed, and high precipitation levels and hence, lower pollutant concentrations (Smith and Tirpak, 1990).

2.3 An Atmospheric Chemistry Model for the Analysis

Atmospheric pollutant concentrations indicate the level of air quality within a region. Due to the complex chemistry within an urban plume, atmospheric concentrations are not linearly related to pollutant emissions. Therefore, emissions are not an adequate

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[12] Increasing temperature has also been shown to affect cloud cover, mixing layer thickness, reactant concentrations, evaporative emissions, biogenic VOC emissions, among other factors contributing to air pollution and climate change. (Smith and Tirpak, 1990)

measure of air quality. However, measuring emission rates and changing emission rates resulting from the implementation of a specific climate change policy is a workable task. On the other hand, determining and predicting pollutant concentrations from policy demands the extra labor of converting emission rates into concentrations. Accomplishing this objective requires the use of atmospheric chemistry models.

2.3.1 Atmospheric Chemistry Model of Urban Regions

One such model is the California Institute of Technology-Carnegie Institute of Technology (CIT) Urban Air Shed Model (McRae, *et al.* 1982). This model simulates the physics and chemistry of the atmosphere that age the pollutants in an urban plume. This model predicts spatial and temporal distribution of both inert and chemically active pollutants in the urban scale. For practical considerations, this model is not appropriate to simulate the air quality conditions over an entire year. It was designed to investigate air pollution episodes, which typically last up to three days.

The CIT model requires hourly, grid-resolved emission data, boundary and initial chemical conditions, and hourly fields of several meteorological variables. In addition, the model is computationally expensive to simulate one day. For the purposes of an investigation of secondary benefits, the desired model is capable of simulating conditions for an entire year for a number of different megacities. Therefore, for this application, a reduced-form of the CIT Urban Air-Shed Model, which is less data and time intensive, is necessary.

2.3.2 Reduced-Form Atmospheric Chemistry Model

The reduced-form atmospheric chemistry model employed in this research, the so-called Metamodel described in detail in Calbo *et al.* (1997), is a parameterization of the CIT Urban Air Shed Model. The Metamodel defines a set of polynomial equations to approximate the predictions of the CIT Urban Air Shed Model. The Metamodel allows an efficient estimation of the concentrations of several pollutants over an entire year and accommodates the limitations of data availability and computational time. In this way,

emissions from pollutant sources and local meteorology are taken into account to simulate the aging processes that occur in an urban plume.

The Metamodel utilizes the probabilistic collocation approach which uses random variables to represent the uncertain parameters (Tatang, 1994). The model becomes increasingly expensive, computationally, with the addition of more input parameters. The Metamodel provides a good fit to the CIT model using fourteen uncertain input parameters. These inputs of emission data and meteorological fields are used to determine net fluxes of pollutants from the urban plume to the global atmosphere, and also to predict pollutant concentration levels over the megacity. The model assumes a 300 square kilometer domain which captures the urban-scale chemistry while limiting the effect of the boundary conditions. This domain does not account for topography, and the area is modeled as a two-dimensional surface.

The meteorological conditions aid in defining characteristics specific to a given megacity, as well as in determining the potential level of pollution. Solar radiation is the catalyst for a number of photochemical reactions and a key factor in determining the rate of the chemical reactions. The amount of solar radiation entering the domain is modeled by three parameters: the seasonal effect or the day of the year, the position of the city or the latitude, and the amount of cloud cover. In addition to solar radiation, reaction rates are also proportional to the air temperature.

Essential to the simulation of pollutant formation within an urban plume is wind velocity and mixing layer height. The mixing layer height, which acts to trap pollutants, varies over the course of the day. The wind velocity parameter is modeled by the residence time, or how long the pollutant remains in the domain, which incorporates both wind velocity and domain size. The wind velocity is assumed unidirectional and constant over space and time.

In addition to meteorological data, data on emission rates is necessary to define the city and potential pollution. The pollutant emissions, SO_2, CO, NO_x, and VOC, are distrib

Table 2.3   Metamodel Inputs

Input	Parameter
Seasonal Effect	Date
Position of City	Latitude
Air Temperature	Difference in actual temperature and climatic mean temperature for day and latitude
Cloud Cover	*Assumptions* Uniform in space Constant in time *Range* 0-1
Mixing Layer	Maximum mixing layer height per day *Assumptions* Daily cycle

Residence Time	Ratio between the length of domain and wind velocity
	 Assumptions Flat domain
	      No vertical variation
	      Constant wind speed
	      Unidirectional wind

SO$_2$ Emissions	Mass CO emitted daily per area *Assumptions* Evenly distributed as area sources No daily cycle *Range* 0-50 kg/km^2/day *Maximum probability* 10kg/km^2/day
CO Emissions	Mass CO emitted daily per area *Assumptions* Evenly distributed as area sources Follows daily traffic cycle. *Range* 0-500 kg/km^2/day *Maximum probability* 100 kg/km^2/day

VOC Emissions	Mass VOC emitted daily per area
	 Assumptions Correlated to CO emissions
	          $E_{voc/co} = .32 E_{co} + .98$
	          $E_{voc} = (1 + DVOC)E_{voc/co}$
	      Follows daily traffic cycle
	 Range DVOC = +/- 70%

NO$_x$ Emissions	Mass NO$_x$ emitted daily per area
	 Assumptions 95% NO, 5% NO$_2$
	      Correlated to CO emissions
	          $E_{nox/co} = 0.23 E_{co}$
	          $E_{nox} = (1 + DNOX)E_{nox/co}$
	      Follows daily traffic cycle
	 Range DNO_x = +/- 70%

Initial and Boundary Conditions	Air Quality Indexes (AQI)
	 NO$_x$, VOC, SO$_2$, O$_3$:
	 Assumptions I.C.'s vary over domain
	      B.C.'s constant over domain
	      Surrounding area not very polluted
	 Range 0-1

uted as area sources across the domain, and they are assumed to enter the megacity in a circular pattern. In other words, 25% of the emissions are input into the center of the domain with emission levels decreasing with distance from the urban core. The emissions are distributed into these six rings with the ratios of 1, 0.31, 0.125, 0.075, 0.05, and 0.125 (Figure 2.4).

Figure 2.4 Spatial Distribution Pattern of Emission within the Megacity.
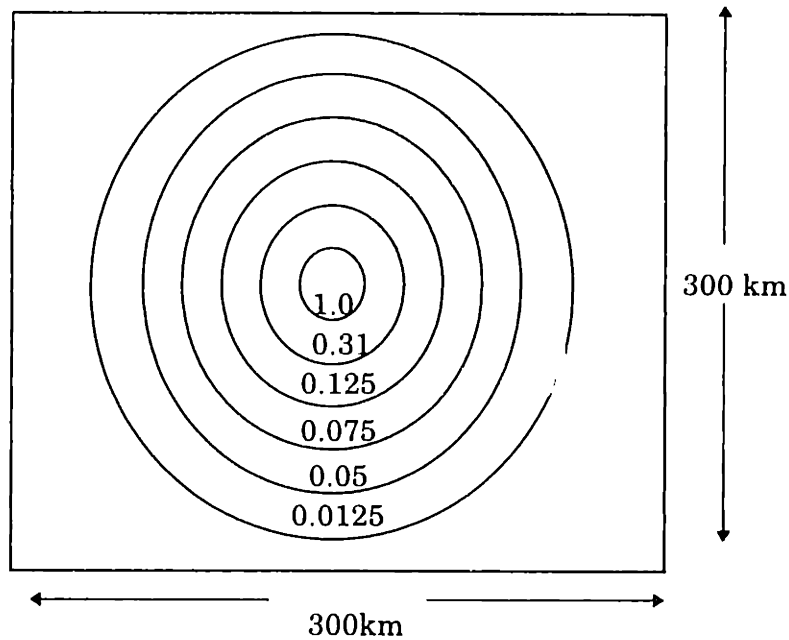
The CO, VOC, and NO_x emissions follow a daily cycle based on the traffic cycle (Figure 2.5). SO_2 is uniform throughout the day. The initial and boundary conditions are set with air quality indexes which range from zero to conditions of a polluted atmosphere. Regarding boundary conditions, it is assumed that the pollutant emissions in areas surrounding the megacity meet the National Ambient Air Quality Standards. The initial AQI value is the concentration in the megacity center and directly downwind. Lower concentrations are assumed for other regions in the domain.

Given these fourteen input parameters, the Metamodel is capable of simulating the performance of the CIT model. While inputs are expensive using the probabilistic collocation method, the outputs are essentially free. The Metamodel outputs include average concentrations of ozone, SO_2, NO_x, and CO at 8 hour intervals (Table 2.4). Maximum ozone concentration is output in the form of the average ozone concentration over the

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Figure 2.5   Daily Cycle of CO, NOx, and VOC Emissions
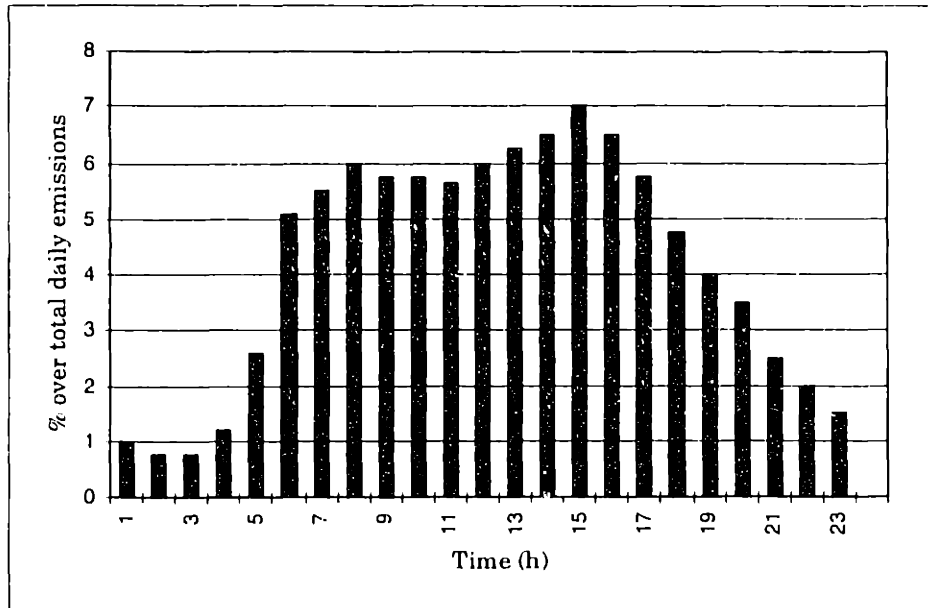
Source: Calbo et al.(1997)

domain at the time of maximum ozone. The deposition of SO_2 and the mass and flux of HNO_3 and PAN are determined every 6 hours.   These concentration outputs represent the eight-hour average concentration over the domain.

Pollutant concentrations provide an indication of air quality over the megacity and the Metamodel is used to determine the effect of a CO_2 reduction policy on air quality. Implementing this model requires the characterization of megacities over a region, aggregating meteorological fields and forecasting pollutant emission rates over time.

Table 2.4 Metamodel Outputs

Output Parameter	Time	Species
Total Mass inside the domain	0, 6, 12, 18, 24 Hour	All species
Net fluxes out of the domain	0, 6, 12, 18, 24 Hour	All species
8-hour average concentrations	8, 16, 24 Hour	SO_2, NOx, CO, O_3, SO_2 deposition, Peak O_3

CHAPTER 3

FORECASTING MEGACITY EMISSIONS

Air pollution is the product of primary emissions and complex chemistry generated largely in urban plumes. The analysis of air quality must take place on an urban-scale as opposed to a national scale. In this way, the non-linear relationships between pollutant emissions and its subsequent adverse impacts can be taken into account. The Metamodel presented in the previous chapter (Section 2.3.2) is applied to this task and the process to determine the input parameters are developed in this chapter. This process includes the characterization of the megacities and a method to forecast SO_2, NO_x, CO and VOC emissions.

An existing global scale economic model is used to forecast emission scenarios under different climate change policies. Because the forecasts are resolved for a nation or a group of nations, these regional emissions must then be distributed to the defined megacities within the region. Each step in this process requires assumptions. These assumptions are explicitly stated in Table 3.1, addressed in the following text, and discussed with regards to improvements in Chapter 6.

Table 3.1   The governing assumptions of this methodology.

Task	Assumption
Characterize Megacity	No new megacities develop, they grow where they are.
Forecast Regional Emissions of Trace Gases	The disaggregation of energy use per fuel type within a sector remains constant over time.
Distribute Megacity Emissions	Constant megacity-to-region emission ratio.
Impose Policy	Regional trace gas emission reductions from policy are applied uniformly to emissions in megacities.

3.1 Characterizing Megacities

The first step in analyzing the secondary effects of CO_2 reduction on an urban scale is to define the characteristics to identify a megacity within a region. This definition must meet a number of design criteria including domain size, per area emission rates, and population density. The megacity must be of a size to simulate urban-scale atmospheric chemistry. The emission rates per area must be relatively high because these levels act as a significant determinant in pollution formation. Finally, high population density, correlated to high emission rates, translates to greater exposure to high pollutant concentrations which can lead to potentially high damages. Practical criteria must be considered as well. These include the availability of data, the requirements and constraints of the models employed, and the desire to develop a megacity definition which is applicable to any world region.

To locate areas around the world that meet this criteria, population and emission inventories are essential. The Global Emissions Inventory Activity (GEIA) databases (Graedel, et. al, 1996) include global inventories of nitrogen oxide (NO_x) emissions, sulfur dioxide (SO_2) emissions, and natural VOC (NVOC) emissions for the year 1985, and population data for the year 1990. These databases are global and have a one degree by one degree resolution, or 110 km^2 at the equator for each grid cell. The GEIA databases are well documented and represent a compilation of the best available emissions data[13].

The geographic resolution and coverage provided by the GEIA database facilitate the identification of the megacities around the globe. Choosing a domain size consistent with both the Metamodel and the resolution of the database, the megacity is defined as an area of 300 square kilometers. A domain of these dimensions adequately accounts for the tropospheric chemical reactions occurring on the urban scale, while limiting possible distortions from boundary condition assumptions. Therefore, in terms of the GEIA database, a megacity covers nine 1° x 1° grid cells. Naturally, the dimensions of the regions in square kilometers varies with latitude.

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[13] More detailed emission inventories exist for the United States, the OECD countries and Japan, and the GEIA inventory for these areas is more accurate than for developing countries.

The following method is used to identify the relative importance of the megacity within a region or country:

1. The NO_x, SO_2, and population data are assigned to each grid cell designated by latitude and longitude using the GEIA databases.

2. Because each megacity incorporates nine grid cells, it is necessary to rank the cells based on not only their attributes, but on the levels in the surrounding grid cells in order to determine the high emission regions. The emissions and population are totaled for each grid cell and its eight adjacent cells:

$$\sum_{i=i-1}^{i+1} \sum_{j=j-1}^{j+1} P_{ij}$$

where P_{ij} is the pollutant emissions or population at latitude i and longitude j. Therefore, each grid cell designated by latitude and longitude is assigned a value which represents the aggregation of itself and its eight surrounding cells.

Each cell initial contains
data of one GEIA cell

1	1	1	1	1
1				1
1				1
1				1
1	1	1	1	1

⟶

Each cell then contains
data of nine GEIA cells

	13	15	13	
	15		15	
	13	15	13	

3. The values of these grid cells are ranked for each pollutant and population[14], with cell #1 representing the maximum emissions or population.

4. The grid cells incorporated in the highest ranking grid cells (i.e. eight adjacent grid cells) are eliminated from the ranking to avoid double counting. Then the

[14] The latitude effects on area have not been taken into account when determining the ranking of NO_x, SO_2, and population. Each grid cell is given equal weight.

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second highest rank is identified, the ranking of its adjacent grid cells are eliminated, and so on.

Defining a megacity based on 1985 emissions and population distribution does not consider the fact that new megacities may evolve due to economic and population growth and migration. The number of cities with population more than four million rose from 13 to 35 over the thirty year period from 1950-1980. Current projections anticipate 66 such cities by 2000 and 135 by 2025, with the majority appearing in developing countries (UN, 1989). Therefore, while this assumption may be adequate for industrialized countries over the next 15 years[15], development and growth should be accounted for in the designation of future megacities, specifically in developing countries. One possible approach to this issue involves identifying areas of potential growth using urbanization forecasts. Given the latitude and longitude of these potential megacities, these new areas can be represented. The difficulties in modeling the development of megacities also present challenges to forecasting megacity emissions. This topic is revisited in Section 3.2.2: *Distributing Regional Emissions to Megacities*.

3.2   Forecasting Megacity Emissions

Once megacities are defined for a nation or region, the path of emissions must be forecast into the twenty-first century under different climate change policy scenarios. Existing integrated economic and climate models, examining the costs and benefits of different climate change policies, predict energy use, activities and CO_2 emissions. However, the geographic regions analyzed are nations or a group of nations. While this approach provides adequate resolution for climate change analysis, it does not provide enough detail for the analysis at hand. Therefore, application of these existing models requires a method to distribute these regional emissions to the megacities as defined above.

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[15] The time horizon on which to evaluate secondary benefits.

3.2.1 Forecasting Regional Emissions

This study uses the MIT Emission Prediction and Policy Analysis Model (EPPA) to predict regional energy use and CO_2 emission forecasts under different climate change scenarios (Yang, *et al.* 1996). EPPA is a multi-sector, multi-region, global, computable general equilibrium model which can be used to project economic activity, energy use, and greenhouse gas emissions. These projections are driven by various activity levels within the model (Table 3.2) and are generated for 12 economic regions (Table 3.3) from 1985 through the year 2100. The resulting projections of greenhouse gas emissions act as inputs to a coupled chemistry-climate model (Yang, *et al.* 1996).

Table 3.2: Production Sectors in EPPA

Production Sectors	
Non Energy	Agriculture
	Energy-intensive industries
	Other industries and services
Energy	Crude Oil
	Natural Gas
	Refined Oil
	Coal
	Electricity, gas, and water
Future Supply Technology	Carbon liquids backstop
	Carbon-free electric backstop

The EPPA model predicts emissions of CO_2 and trace gases including methane, nitrous oxide, sulfur dioxide, nitrogen oxides, and carbon monoxide. CO_2 emissions are calculated directly from the period-to-period levels of energy sector activities using emission factors for coal, oil, refined oil, and natural gas. Emission factors quantify the mass of pollutant emitted for every unit of energy use. For CO_2, emission factors depend only on the fuel used.

Table 3.3 The 12 Economic Regions in EPPA
For energy sector activities and greenhouse gas emissions

Regions	
United States	Energy-exporting LDC's [16]
Japan	China
EC	India
Other OECD	Dynamic Asian Economies
The former Soviet Union	Brazil
Central and Eastern	Rest of World

The EPPA model predicts trace gas emissions by first establishing a base-level emission inventory. Emissions are then expected to grow as a function of the changing levels of energy and non-energy activities, as documented in Liu (1994). While this method of predicting trace gas emissions is adequate for the purpose of the global model, limitations exist for this application on two fronts. First, air pollution policies and/or the implementation of control technologies are not accounted for. Second, this method does not distinguish between fossil fuel combustion from electricity generation, industry, or transportation. This is important because unlike CO_2 emissions, NO_x, SO_2, CO, and VOC emissions vary depending on the type of fuel used, the technology used, and the sector it is used in. For example, NO_x emissions per energy unit from oil industrial boilers and NO_x emissions from light duty gas vehicle use differ by a factor of two. Therefore, it is necessary to disaggregate energy used per fuel type on a sectoral basis.

In this thesis, the component of the EPPA model which predicts NO_x, SO_2, CO emissions is replaced by a new forecasting technique. The new approach is based on the use of aggregate emission factors as illustrated in Complainville and Martin (1994). For this application, aggregate emission factors specific to the fuel, the sector activity and the region are developed for not only NO_x, SO_2, CO, but also include VOC emissions. In order to develop aggregate emission factors, emissions and energy use per fuel type by sector are required. To illustrate this method, the aggregate emission factors for the United States for a matrix of fuels (coal, crude oil, natural gas, and refined oil) and sector activi-

ties (oil refineries, electricity, transportation and industry) are derived using existing inventories and databases as described below. In further research, emission factors will need to be developed for other global regions.

Energy Emissions

These U.S. aggregate emission factors are based on 1985 emissions data from the U.S. EPA and 1985 fuel use per sector data from the International Energy Agency (IEA). The procedure is:

1. The energy quantities of coal, crude oil, natural gas and refined oil used are divided into the following sectoral uses: oil refineries, electricity generation, transportation, and industry using IEA energy data for the U.S. for 1985.

2. SO_2, NO_x, CO, and VOC emissions are aggregated into a consistent fuel and sector matrix using EPA data for the U.S. for 1985.

3. Dividing the EPA emissions by the IEA energy quantities yields emission factors for the various pollutants by fuel and sector for 1985 as in Table 3.4 (Emission Factor = Emission/ Energy).

Table 3.4 Emission Factors for each pollutant by fuel and sector activity for the United States in 1985 (g/GJ)

(a) SO_2

	Oil Refineries	Electricity	Transportation	Industry
Coal	-	957	-	723
Crude Oil	1134	-	-	-
Natural Gas	-	-	-	48
Refined Oil	-	529	36	190

(b) NO_2

	Oil Refineries	Electricity	Transportation	Industry
Coal	-	371	-	236
Crude Oil	208	-	-	-
Natural Gas	-	195	-	199
Refined Oil	-	153	536	72

(c) CO

	Oil Refineries	Electricity	Transportation	Industry
Coal	-	13	-	37
Crude Oil	1697	-	-	-
Natural Gas	-	17	-	36
Refined Oil	-	16	4507	11

(d) VOC

	Oil Refineries	Electricity	Transportation	Industry
Coal	-	1	-	3
Crude Oil	2237	-	-	-
Natural Gas	-	1	-	8
Refined Oil	-	4	563	4

4. In order to determine emissions, the emission factor is multiplied by the energy use. EPPA predicts energy use for fuels and for sector activities, but it does not accurately apportion the energy use per fuel type among different sector activities. This level of resolution is not necessary for the prediction of aggregate energy use and CO_2 emission projections[17]. Therefore, the 1985 IEA energy data is used to determine the sectoral percentages of energy use per fuel type for the U.S. (Table 3.2).

Table 3.2   Energy Activity Percentage[18]

	Oil Refineries	Electricity	Transportation	Industry
Coal	-	83%	-	14%
Crude Oil	1%	-	-	-
Natural Gas	3%	17%	-	44%
Refined Oil	-	5%	68%	18%

5. In order to verify these emission factors, the 1985 emissions for each pollutant are recreated in the following manner:

$$P_{1985} = \sum_{i=1}^{4} \sum_{j=1}^{4} X_i \times S_i \times EF_{ij}$$

---

[17] Work is currently underway to calibrate these disaggregations in the EPPA model.

[18] Not included are 3% coal, 35% gas, and 8% refined oil to category other. Also note that percentages for crude oil and refined oil are calculated from the total of crude and refined oil.

where P is the pollutant emissions, X_i is the EPPA output for fuel i, S_i is the sectoral percentage for fuel i, EF_i is the emission factor for fuel i and sector j. The sectoral percentages are calibrated in order to predict emissions consistent with the EPA 1985 inventory. Therefore, these emission factors can not be applied to other regions.

6. In order to predict emissions, the approach used to recreate the 1985 baseline is employed for the year of interest. As mentioned previously, the emission factors are assumed to change with time due to improving technologies and increasing pollution control regulations. In this model, an emission factor scaling or a control factor accommodates the dynamic nature of technology. The control factor ranges from 0 to 1, with zero representing a regulation or control technology which limits a sectors emissions completely and 1 representing no change.

$$P_t = \sum_{i=1}^{4} \sum_{j=1}^{4} X_{it} \times S_i \times EF_{ij} \times C_{ijt}$$

where C_{ijt} is a control factor adjustment at time t for fuel i and sector j.

Non-Energy Emissions

The discussion thus far has focused on emissions from energy activities. However, non-energy activities such as metals processing and solvent utilization also produce NO_x, SO_2, CO, and VOC emissions and must be accounted for when developing forecasts. The contribution of these emissions varies among regions depending on the level of industrialization. This study uses elasticity factors to forecast trace gas emissions from these activities.

Non-energy activities accounted for 3.66% of NO_x, 9.33% of SO_2,, 5.9% of CO, and 45.3% of VOC emissions in 1985 (EPA, 1995). The necessary detail to forecast these emissions is limited in terms of EPPA outputs. However, the output of the energy intensive sector does provide a first-cut approximation. In this study, the non-energy activity emissions are forecast in the following manner:

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$$P_t = P_{t-1} \times \left(1 + \alpha \left(\frac{(EI_t - EI_{t-1})}{EI_t} \right) \right)$$

where P is the pollutant emissions at time t, α is an elasticity factor, and EI is the EPPA Energy Intensive sectoral output. The elasticity factor is the variable that represents changes in technology and regulation over time.

In general, the approximations of the non-energy emissions are not as robust as those of the energy emissions. However, given the small percentages of non-energy emissions, with the exception of VOC, this does not impact the forecasts to a large degree. Because almost half of the VOC emissions are from non-energy activities, the sensitivity of the VOC emission forecast to the air quality concentrations is analyzed to determine its effect on the results (*Section 5.6 Sensitivity Analysis*).

This method, using aggregate emission factors for energy emissions and elasticity factors for non-energy emissions, is applied to generate regional emission paths for NO_x, SO_2, CO, and VOC for different climate change scenarios and replaces the existing EPPA trace gas forecasting model. These emissions forecasts are generated for EPPA regions, however, and must be distributed to the megacities within the region in order to explore air quality.

3.2.2 Distributing Regional Emissions to Megacities

Once the regional emissions are forecast and megacities are defined, the next step in the process is to distribute these regional emissions into the megacities. Due to the dynamic nature of urbanization and industrialization, distributing these emissions to the megacity is a difficult task. There are various ways in which these emissions can be distributed, each of which requires various amounts of information and assumptions. Three such methods are addressed below.

1. *Constant region-to-megacity ratio over time:* The existing emissions and population databases are resolved on a 1° by 1° grid. The distribution of emissions in this inventory is constructed from the number of point sources and area sources within the region as well as the control technologies employed. Assuming no new industrial or urban centers develop, maintaining a constant region-to-megacity ratio consistent with the base year database provides a first-cut approximation at the emission distribution over time. This assumption, however, does not account for the fact that new cities develop over time, more so in developing countries than industrialized countries. Therefore this assumption, while valid for industrialized countries, may not be adequate for developing countries. Developing countries experience rapid growth rates, rapid industrialization, and increasing levels of motorization which will effect the distribution of emissions over time.

2. *Constant per capita emissions:* To improve upon this scenario, emissions could be distributed on a per capita emission basis. This method would require a dynamic population distribution forecast. Urban and rural population forecasts which account for population migration, growth, and industrialization could be used. This approach would account for the fact that new megacities develop and acquire a greater portion of the regional emissions over time, and the portion of emissions attributed to the originally defined megacities would decrease.

3. *Segregation of mobile and stationary emissions:* A combination of the two approaches can be used to manage the discontinuities between the control and growth of stationary-source and mobile-source emissions. In urban areas of industrialized countries, over 50% of the pollution is from transportation (Faiz, 1988). However, transport emissions are not only a problem for developed countries. The problems of motor vehicles are often magnified for developing countries because of older vehicle fleets, poorly maintained vehicles, and lack of transportation infrastructure.

To use such an approach, megacity emissions must be disaggregated between the stationary and mobile sources. Likewise, this disaggregation must be forecast through time. The emissions from mobile sources can be forecast as a function of

population or motor vehicle registration[19]. The remaining emissions for stationary sources can be based on the distribution of point-source emitters. For example, large point sources and their emission rates are geographically resolved for the United States, and given current environmental legislation, a geographic emission distribution can be predicted into the future.

For this analysis, the simplest and the least data-intensive method is used: the megacity-to-region emissions ratio is assumed to remain constant over time. Therefore, for the various pollutants the relationship is as follows:

$$P_{u,t} = P_{u,1985} \times {P_{r,t}} \big/ {P_{r,1985}}$$

where P represents the pollutant emissions or population, u refers to the megacity, r refers to the region or nation the megacity is in, and t refers to the year for which the forecast is desired.

For industrialized countries, the urban population tends to grow at the same rate as the national growth ratio (Chen *et al.*, 1994). Hence, for industrialized countries, this method of forecasting emissions does not differ dramatically from a per captia emission distribution scenario. Because changes in land use, urbanization, and plant siting should not change dramatically over the duration of this analysis, which is the next 15 years, this assumptions is adequate. While this assumption may be plausible for industrialized countries, this method must be refined when applied to developing countries.

This method provides an estimate of the annual emissions in the megacity. However, as discussed in Chapter 2 (Section 2.3.2), the Metamodel requires daily emission rates. Therefore, these annual emission figures must be disaggregated over the year. For this study, the annual emissions are simply divided by 365 days per year to yield daily emis-

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[19] Growth in motor vehicle use is projected to outpace the growth of total and urban population (Faiz, 1988).

sions. However, seasonal emission patterns, which are regionally dependent, do exist should be accounted for on a pollutant by pollutant basis in future iterations.

Establishing Base Year Emissions

In order to use this approach, defining the megacity-to-region emission ratio requires regional and megacity emission inventories for the base year. The 1985 emissions inventories for the designated megacity are established from existing inventories which include the pollutants essential to characterizing air pollution: SO_2, NO_x, VOC, and CO.

SO_2, NO_x, and NVOC: As discussed previously, emissions of SO_2, NO_x, (Benkovitz, 1995) and NVOC (Guenther, et. al, 1996) have been inventoried by GEIA on a global grid with a one degree by one degree resolution for 1985. The GEIA inventory is a compilation of the most recent and reliable inventories from each region.

CO and VOC: Currently, high resolution inventories do not yet exist for CO and VOC for the world. However, for the U.S. the Ozone Technical Assessment Group (OTAG) has aggregated state inventories for CO and VOC for point and area sources.[20] This data is used for regions in the United States with the exception of California. For California, information on air shed emissions from the California Air Resources Board (CARB) is used. When analyzing megacities without such inventories, statistical relationships must be developed. For example, in urban areas the CO and VOC emissions are largely from the transport sector. Therefore, they can be approximated as a function of population or NO_x transport emissions.

Population: As stated previously, the GEIA database provides an inventory of 1990 population with a one degree by one degree resolution (Li, 1996).

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[20] The EPA has developed an emissions inventory of CO and VOC for the United States for 1985 with ¼ by 1/6 degree resolution (NAPAP, 1985).

3.3 Implementation of Climate Policies

Imposing climate change policies in EPPA determines changes in energy use, sector activity and CO2 emissions. Using aggregate emission and elasticity factors in conjunction with these changed variables, the changes in trace gas emissions are forecasts. These emission reductions on a pollutant by pollutant basis apply to the aggregate reductions across the nation. These reductions must be distributed to the megacities and this requires an assumption. In this iteration of the model, the megacity emissions are reduced by this regional amount. Therefore, each megacity experiences the same percent of emission reductions. For example, if a carbon tax impacts fossil-fueled power plants harder than it does the transport sector, the NO_x emission reductions may be overestimated in the urban areas. An underestimation may result if the transport sector activity is affected to a greater degree.

CHAPTER 4

EVALUATING BENEFITS OF REDUCED AIR POLLUTION

Urban and industrial environments are efficient for the production of air pollution, and so the current rate of urbanization and industrialization worldwide presents challenges from a public health and environmental perspective. Because urban agglomerations continue to experience rapid growth, especially in developing countries, populations are reaching figures in the millions. For example, it is projected that 66 cities will possess over 4 million inhabitants by the year 2000, and by 2025 there will be 135 such cities (UN, 1989). The combination of high pollution and high population in urban areas leads to increased exposure and increased damage costs associated with high pollutant concentrations.

Of particular concern is the rapid rate of urbanization and industrialization occurring in developing countries (Figure 4.1). While the world's urban population is projected to reach 47% by the year 2000, that of Latin American countries will near 80%. The level of a country's development influences the level of ambient pollutant concentration, because development plays a significant role in the levels and types of pollutants emitted, the level and diffusion of control technologies, and the environmental regulation and enforcement. Several studies (Selden and Song, 1994; Grossman and Krueger, 1993; Shafik and Bandyopadhyay, 1992) have identified an inverted-U relationship between pollution and economic development. As nations begin to develop and industrialize pollutant concentrations increase. Once they reach a certain level of development, characterized by per capita gross domestic product (GDP), these increasing concentrations begin to decrease and level off. This pattern is attributed to the diffusion of control technologies, the demand for environmental quality, and the nature and siting of industry that accompany increasing wealth.

Given these relationships, it is anticipated that the urban agglomerations of developing countries have the most potential to gain from secondary air quality benefits of CO_2 reduction policies. In these cities, the baseline air quality is poor, control technology is not diffused, and environmental regulation, if existing, may not be enforced (Wijtilleke and Karunarantne, 1995).

Figure 4.1 Percent Urban Population of Total Population for World Regions
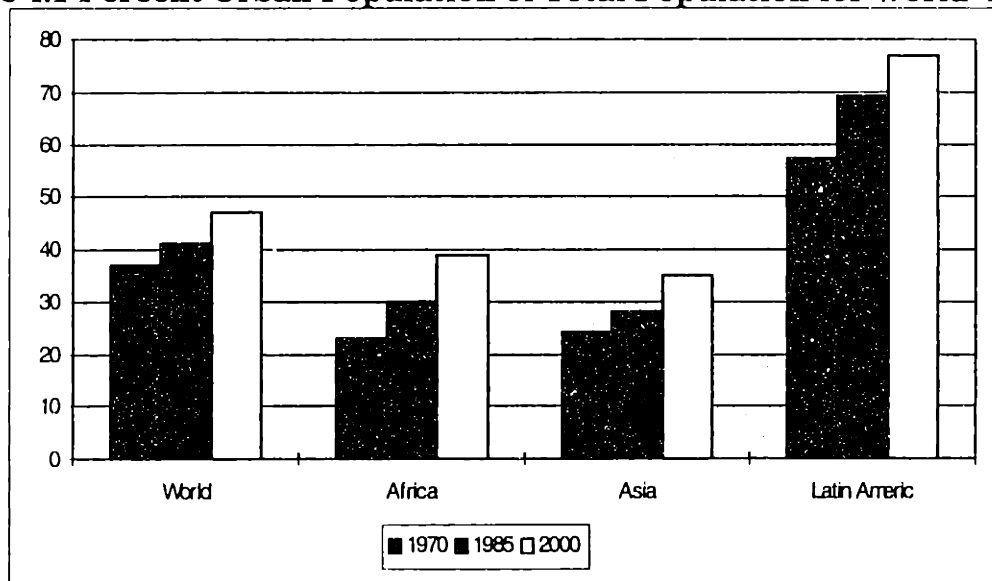
Source: UN (1991)

4.1 Qualitative Discussion of the Adverse Effects of Air Pollution

The rise in air pollutant emissions that are expected to accompany the rise in energy use related to urbanization will expose roughly 300-400 million urban inhabitants to increased health risks. Most studies of the direct damages from air pollution concentrations identify human health effects as the dominant component of the air pollution costs (Small and Kazimi, 1995). Therefore, urbanized areas with high population density and concentrated pollutants experience the bulk of the adverse effects[21] because the number of people exposed and susceptible to damages is high.

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[21] Non-urban areas may also suffer from high pollutant concentrations due to transported or biogenic emissions.

The process leading from human activity to adverse impact, discussed in Chapter 1 (Figure 1.2), is reviewed with focus on the link between concentrations and impacts. Pollutants are emitted as either products or by-products of human activities, such as electricity generation or motor vehicle use. These emissions are diluted, transported, and transformed in the atmosphere and a certain level or concentration of the pollutant remains in the ambient air. People outdoors are exposed to air pollutant concentrations and receive a "dose" specific to their breathing rate among other characteristics. The person, having inhaled or sensed the pollutant, may exhibit acute and chronic adverse health effects which depend on age, health status, etc.

The pollutants and their suspected human health impacts are shown in Table 4.1. While clinical and epidemiological studies support the linkage between pollutant exposure and human health impacts, the effects remain highly uncertain. The first six pollutants in this table, from sulfur dioxide to lead, are regulated in the U.S. by the National Ambient Air Quality Standards under the Clean Air Act Amendments of 1990 (Table 4.2). All of these so-called criteria air pollutants are primary pollutants with the exception of ozone. Based on health studies, particulate matter, ozone, and lead pose the greatest risks to human health. In the U.S., environmental legislation has limited lead emissions. However, due to the complexity of ozone formation (See Chapter 2, Section 2.3) and the disperse sources of PM, these pollutants remain a considerable threat to exposed populations. In developing countries, the situation is more serious. Due to the lack of control technologies and the use of lead in gasoline, all of these pollutants present serious risks to the exposed populations.

Other air pollutants in Table 4.1 include VOCs and toxic substances. VOCs or hydrocarbons are present in many forms in the atmosphere and have different levels of toxicity. VOCs are both emitted directly from sources and created by chemical reactions in the atmosphere. Toxic substances are released into the air largely as a result of industrial activity and motor vehicle use in industrialized countries. In addition to the risks presented on a pollutant by pollutant basis, there is growing evidence that the synergistic effects of these pollutants in combination may be far more serious than the adverse effects of individual pollutants (Faiz, 1988).

Table 4.1 Health Effects of Air Pollutants

Pollutant	Health Effect
Sulfur Dioxide	Aggravates existing lung diseases, especially bronchitis. Constricts the breathing passages, especially in asthmatic people and people doing moderate to heavy exercise. Causes wheezing, shortness of breath, and coughing. High levels of particulates appear to worsen the effect of sulfur dioxide, and long-term exposures to both pollutants leads to higher rates of respiratory illness.
Particulate Matter	Aggravates existing heart and lung diseases. Changes the body's defenses against inhaled materials. Damages lung tissue. The elderly, children and those with chronic lung or heart disease are most sensitive. Lung impairment can persist for 2-3 weeks after exposure to high levels. Chemicals in and on particles can also be toxic.
Nitrogen Oxides	Irritate the nose and throat, especially in people with asthma. Appears to increase susceptibility to respiratory infections.
Carbon Monoxide	Weakens the heart's contractions and lowers the amount of oxygen carried by the blood. Reduces your ability to exercise and is dangerous for people with chronic heart disease. Can cause nausea, dizziness, headaches, and when it's very concentrated, death.
Ozone	Irritates the lungs and breathing passages, causing coughing and pain in the chest and throat. Increases susceptibility to respiratory infections and reduces the ability to exercise. Effects are more severe in people with asthma. Long-term exposure may lead to scarring of lung tissue and lowered lung efficiency.
Lead	Affects circulatory, reproductive, nervous, and kidney systems. Suspected to cause hyperactivity and lowered learning ability in children.
VOCs	Low-molecular weight compounds cause eye irritation, coughing and sneezing; High-molecular weight compounds may have carcinogenic or mutagenic effects.
Toxic Substances	Suspected of causing cancer, reproductive problems and birth defects. Benzene and asbestos are known carcinogens. Aldehydes and ketones irritate the eyes, cause short-term respiratory and skin irritation and may be carcinogenic.

Source: Faiz(1988), NJEPA (1997)

Table 4.2 National Ambient Air Quality Standards

Pollutant	Averaging Time	Primary	Secondary
Sulfur Dioxide	24 Hour Annual Average 3 Hour	0.030 ppm 0.14 ppm	0.5 ppm
Particulate Matter	24 Hour Annual Mean	150 μg/m 50 μg/m^3	
Nitrogen Oxides	Annual Average	0.053 ppm	
Carbon Monoxide	8 Hour 1 Hour	9 ppm 35 ppm	
Ozone	1 Hour	0.12 ppm	
Lead	Calendar Quarter	1.5 μg/m^3	

[1] Set to protect human health.

[2] Set to protect environmental health.

These adverse impacts to human health coexist with damage to ecosystems and materials. Acid deposition increases the acidity of lakes and streams presenting a hazard for fish and other aquatic life, can lead to changes in the nature of soil leading to deterioration in forest growth, and can erode the surface of many materials such as stone, metal, and stained glass (ApSimon and Cowell, 1996). Ozone also has adverse effects on crops, vegetation, paints, and rubber products.

4.2 Quantifying the Effects from Air Pollution

Clinical, epidemiological, and ecological studies have attempted to link the exposure of pollutant concentrations to an adverse effect. This research develops critical load models

or dose-response functions for environmental damages and health damages respectively. Each of these is reviewed below.

Critical Load Models

The critical load function assumes that the pollutant receptor, such as the soil, a building, etc., has some capacity to absorb the pollutant without experiencing adverse effects. The highest level at which no damage occurs is referred to as the critical load. Damage functions are then created such that no damage occurs below the critical load value and increases toward 100% loss as the impact increases, with more moderate damage occurring at impact levels in between (Gregory, *et al.* 1996).

In this thesis, while sulfur dioxide deposition, HNO_3 and PAN masses in the region and fluxes out of the region are presented, there is no quantification of these damages. This damage assessment is data-intensive and a number of the effects occur outside the megacity. Significant research has been devoted to these environmental damages since the discovery of transboundary pollutants in Europe in the early 1980's and critical load models are available[22].

Dose-Response Models

The adverse effects to human health are typically modeled not by critical load functions, but with dose-response functions. Using epidemiological data, dose-response functions attempt to relate exposure and biological impact and thereby indicate how an organism's response to a substance changes as its exposure increases. The dose-response relationships assume a continuous variation of impacts with changes in exposure including impacts at low doses (Krupnick *et al.*, 1993). These models assume a critical load, or threshold level, at which zero damage occurs does not exist. This assumption, while upheld by epidemiological and clinical data, is highly debated among scientists and policy makers[23].

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[22] For further discussion, see ApSimon, H. *et al.* (1996); Chestnut (1996).
[23] This debate has taken center stage as the U.S. EPA is proposing to revise the current NAAQS for ozone and particulate matter in which the legislation is a health based standards and does not allow a cost-benefit analysis.

Dose-response functions tend to be estimated in two ways: a time-series or cross-sectional approach. The time-series approach relates a time series of pollution concentration data to a given cohort of people whose health status has been recorded over time. The cross-sectional approach correlates health data in different areas within cities, or even across cities, with factors that are likely to explain variations in that health status (Pearce, 1996). The dose-response relationships vary between regions due to lifestyle and other behavioral differences such as diet, smoking, and time spent outdoors. Most dose-response studies have been conducted in industrialized countries (Krupnick *et al.*, 1993).

4.3   Method to Assess the Health Impacts

This study examines the health effects of air pollution using existing epidemiological studies and a methodology similar to that used by the U.S. EPA in their Regulatory Impact Analysis for setting NAAQS. Health damage studies proceed by establishing average levels of the ambient concentration of each pollutant. These concentrations are then related to health effects using the dose-response functions and are related to the population at risk. The change in health is then modeled as follows:

$$dH = m_{DRF} \times POP \times dC$$

where *dH* represents a change in health, m_{DRF} represents the slope of the dose-response function, *POP* represents the population exposed, and *dC* represents the change in pollutant concentration.

The change in pollutant concentrations (dC):
In this study, the ambient air concentrations for a given megacity are forecast for a policy and no policy case. The differential in air pollutant concentrations on a daily basis provides the incremental change required to determine the resulting change in health. The pollutant concentrations output from the Metamodel represent average concentrations across the 3° x 3° domain. However, pollutant concentrations vary throughout the day and among different areas across this domain.

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Population exposed (POP):.

In order to determine the impact of the effects it is necessary to determine the population exposed. Ideally, human exposure estimates account for the population's typical outdoor, indoor and in-vehicle exposures by time of day, activity level, and demographic group (Hall *et al.*, 1992). However, this detail of human exposure modeling is not available for this study. The population exposed to the pollutant is taken from the 1990 GEIA database for the 3° x 3° megacity. The number of children, adults, and asthmatics in the megacity are estimated from the U.S. Census database.

Slope of dose response function (m_{DRF}):

The slope of the dose-response function represents the adverse effect associated with the dose or concentration. This study utilizes the dose-response functions for criteria air pollutants aggregated by Oslo (1994). Oslo (1994) reviews the current epidemiological and clinical literature which link air pollutants to adverse health effects and extracts dose-response functions where available from peer-reviewed literature. The health effects for the ozone, SO_2, NO_x, CO, and PM which are characterized in Oslo (1994) are shown in Table 3.2. A range of estimates are provided in order to quantify the uncertainty in these models. The central estimate represents the highest probability case and the low estimate represents a lower bound on effects and the high estimate represents an upper bound on the effects. These models have been annualized by Oslo(1994).

However, the data used in determining the dose-response relationship represent a daily change and a daily effect. Given the outputs of the Metamodel used here, the slope of the dose-response functions are reconverted to accommodate daily figures. In order to determine the annual benefits, it is necessary to aggregate the daily figures. For each pollutant and each symptom of each pollutant the daily effects can be summed across the year. However, in the process of this aggregation important findings may be lost. For example, if ozone increases on a number of polluted days, but overall the trend is a decrease, the annual aggregate of the effects may be lower under the policy scenario and knowledge of the possible increased concentrations may be overlooked.

This approach of analyzing the health benefits is limited in three general respects: linear concentration-response functions taken from literature, spatially-limited air quality

data, and aggregate population data. First, these concentration-functions are linear and may not distinguish between the severity of effects at different concentration levels. In addition, the models incorporate assumptions about exposure times and intake doses. Second, because air quality differs across the defined megacity, using domain-averaged concentrations does not allow the examination of this variation. Third, the population and their exposure varies across the region. This approach, while not meeting the rigor of health effect assessments, can produce an order of magnitude range of effects (Hall, 1996), and thus provide an understanding of the extent and severity of air pollution impacts resulting from climate change policies.

Table 4.1 Dose-Response Model for Health Impacts of Air Pollutants

a) Ozone (per year in $\mu g/m^3$)

Health Effect	Upper	Central	Lower
Change in Respiratory hospital admissions	0.012 *dO_3 max	0.0077 *dO_3 max	0.0038 *dO_3 max
Occurrence of minor restrictions in activity	51 * dO_3 max	34 * dO_3 max	17 * dO_3 max
Change in respiratory symptom days/ person	96.6 * dO_3 max	54.75 * dO_3 max	28.11 * dO_3 max
Change in eye irritations	29.9 *dO_3 max	26.6 * dO_3 max	23.4 * dO_3 max
Change in asthma attacks per year	189.8 * dO_3 max	68.44 * dO_3 max	38.69 * dO_3 max

b) Sulfur Dioxide (per year in $\mu g/m^3$)

Health Effect	Upper	Central	Lower
Percent change in mortality	0.121 * dSO_2	0.048 * dSO_2	0.020* dSO_2
Change in cough per 1,000 children	0.0262 * dSO_2	0.0181* dSO_2	0.010* dSO_2
Change in chest discomfort per adult	0.015 * dSO_2	0.010 * dSO_2	0.005 * dSO_2

c) Nitrogen Oxides (per year in $\mu g/m^3$)

Health Effect	Upper	Central	Lower
Change in respiratory symptoms per year	$14.42 * dNO_2$	$10.22 * dNO_2$	$6.02 * dNO_2$

d) Carbon Monoxide

There is little quantitative dose-response information linking CO exposure to a meaningful health endpoint.

e) Particulate Matter (per year in $\mu g/m^3$)

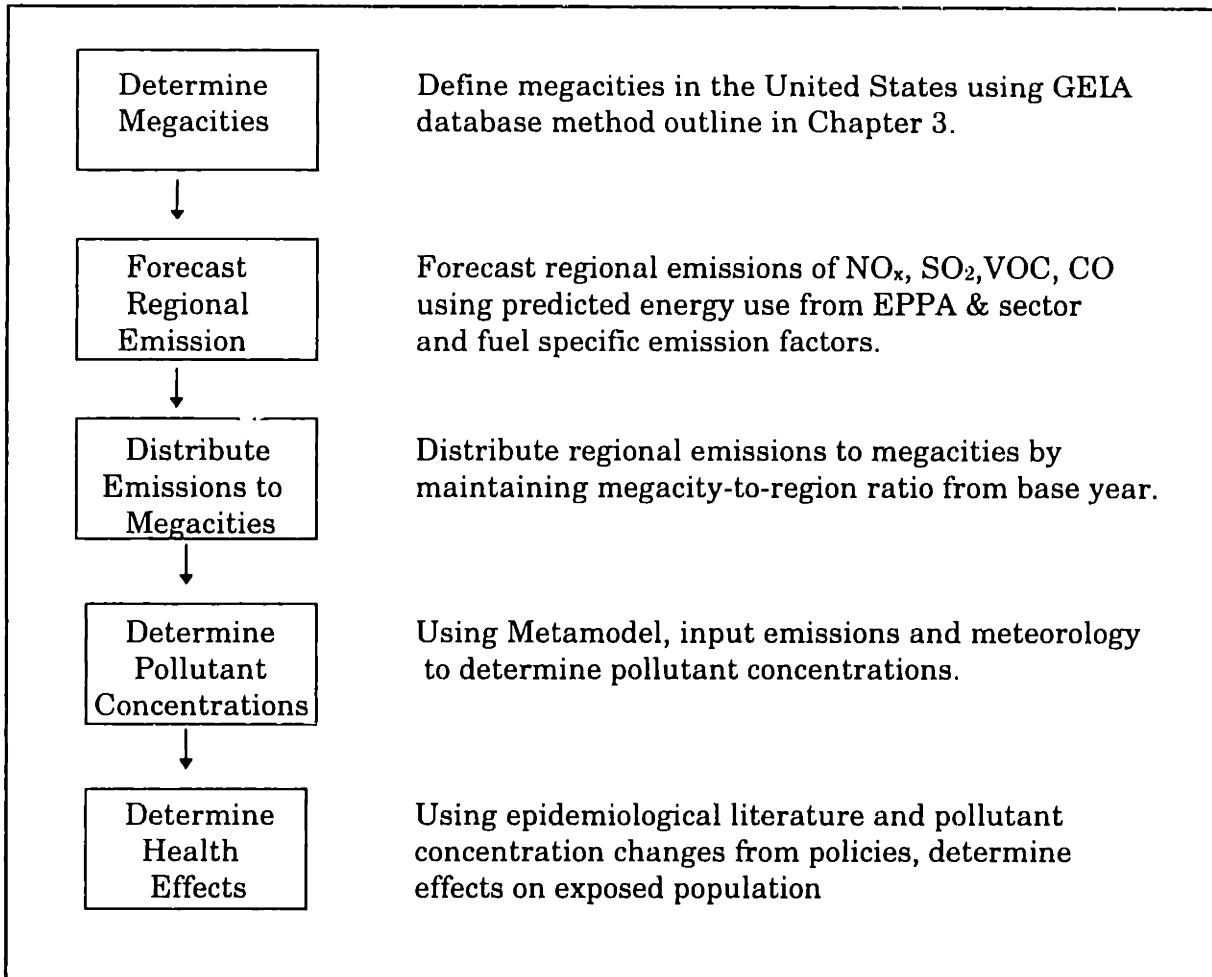| Health Effect	Upper	Central	Lower
Change in mortality	$9.1*10^{-6} *dPM_{10}$	$6.72 * 10^{-6} *dPM_{10}$	$4.47 * 10^{-6} *dPM_{10}$
Change in RHA per 100,000 Adults	$1.56 * dPM_{10}$	$1.20 *dPM_{10}$	$0.657* dPM_{10}$
Change in Emergency Room Visits	$34.25 * dPM_{10}$	$23.54 *dPM_{10}$	$12.83 *dPM_{10}$
Change in Restricted Activity Days	$0.0903* dPM_{10}$	$0.0575 * dPM_{10}$	$0.0404* dPM_{10}$
Change in Respiratory Illness in Children	$0.00238* dPM_{10}$	$0.00169 * dPM_{10}$	$0.0008* dPM_{10}$
Change in asthma attacks	$0.237 * dPM_{10}$	$0.0326* dPM_{10}$	$0.0163* dPM_{10}$
Change in respiratory symptoms per person	$0.274 * dPM_{10}$	$0.183 * dPM_{10}$	$0.091*dPM_{10}$

Source: Oslo (1994)

CHAPTER 5

APPLICATION OF METHOD TO THE UNITED STATES

Under three different climate change policy scenarios, local air quality is investigated for ten U.S. megacities in the year 2000. Air quality is quantified by 8-hour pollutant concentrations of CO, SO_2, O_3, NO_x, H_2SO_4, PAN, and HNO_3 over the megacity. These scenarios give insight into three main relationships: first, the relationship between pollutant emissions and pollutant concentrations; second, the effect of regional characteristics on pollutant levels; and third, the impact of different CO_2 reduction policies on pollutant levels. The main findings of this research are as follows:

- Primary pollutant emission reductions do not correspond linearly to primary or secondary pollutant concentrations; therefore, approximating a damage function based on decreases in precursor emissions is not adequate.

- The rate and combination of pollutant emissions and meteorology specific to a region determine the level of secondary pollutant production and the magnitude of decreases in primary pollutant concentrations. The changes in secondary pollutants are considerably smaller relative to changes in their precursor emissions. It is necessary to account for regional variability in determining the level of secondary benefits.

- The level of secondary air quality effects depends on the climate policy implemented. Under certain combinations of pollutant emission reductions and meteorological characteristics, the production of certain secondary pollutants may increase rather than decrease and an aspect of the air quality may degrade. However, the complex chemistry and the numerous chemical species in the atmosphere make emission reductions valuable.

- An integrated approach to climate change policy and air pollution policy is necessary given the possibility for negative impacts on air quality. Coordina-

tion between the two efforts could lead to determination of cost-effective solutions which meet multiple policy objectives.

The steps in this process are as follows:

Determine Megacities	Define megacities in the United States using GEIA database method outline in Chapter 3.
Forecast Regional Emission	Forecast regional emissions of NO_x, SO_2,VOC, CO using predicted energy use from EPPA & sector and fuel specific emission factors.
Distribute Emissions to Megacities	Distribute regional emissions to megacities by maintaining megacity-to-region ratio from base year.
Determine Pollutant Concentrations	Using Metamodel, input emissions and meteorology to determine pollutant concentrations.
Determine Health Effects	Using epidemiological literature and pollutant concentration changes from policies, determine effects on exposed population

5.1 Determining the Megacities of the United States

The United States is used to test the application of this methodology, and present the types of information such a study could supply. The United States is chosen because of the accuracy and availability of emissions data and sectoral energy use, and the ability to determine emission factors. In addition, the difficulties in addressing the rapid rates of urbanization, population migration, industrialization expected in developing countries can be averted. However, it is hypothesized that the dramatic effects on air quality from CO_2 reduction policies will occur not in industrialized countries but in developing coun-

tries where local emission controls are not diffused. It is in these regions that the baseline air quality will be considerably low and the potential for accruing benefits will be higher. With success in the U.S., the issues that face developing countries can be addressed in future research.

The megacities for the United States are determined using the GEIA databases and the method outlined in Chapter 3 (Section 3.1). The distribution of SO_2 emissions (Figure 5.1), NO_x emissions (Figure 5.2), and population (Figure 5.3) are plotted for the United States. The value in each 1° x 1° cell represents the emissions or population in this cell as well as the data in the surrounding eight grid cells. In this respect, each cell defines a potential megacity.

Inspection of these graphs highlights the difference in emissions and population distribution across the nation. These differences are attributed to the siting of industrial centers, fossil-fueled power plants, and urban centers. This also highlights the necessity for a regional analysis, such as the megacity approach discussed here. Urban and industrial areas contribute a majority of the secondary pollutants to the national total. The SO_2 emissions characterize the power plant locations, whereas the population figures characterize an urban area. The NO_x emissions, due to power plants and motor vehicles, is correlated well with both population and SO_2 emissions.

The SO_2 emission distributions are heavily concentrated in the Midwestern United States. This area is densely populated with coal-fired power plants. Because 975 grams of SO_2 are emitted for every gigajoule of energy for electricity generation from coal which accounts for 83% of the coal use, high concentrations of SO_2 are expected in this region. For industrialized countries, SO_2 generally is not a problem in urban areas due to the siting of power plants outside urban areas and strict emission standards and National Ambient Air Quality Standards. SO_2 emissions are not an appropriate indicator of urban agglomerations.

Figure 5.1   SO₂ Emissions for the United States
Each grid cell represent the emissions of a total of nine GEIA 1° x 1° grid cells.
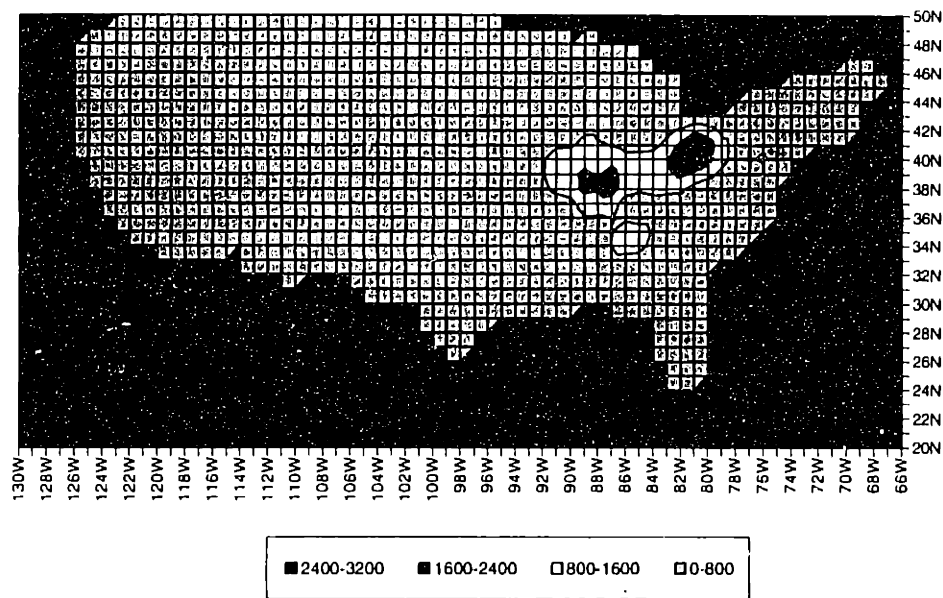

■2400-3200   ■1600-2400   ☐800-1600   ☐0-800

Figure 5.2   NOₓ Emissions for the United States
Each grid cell represents the emissions of a total of nine GEIA 1° x 1° grid cells.

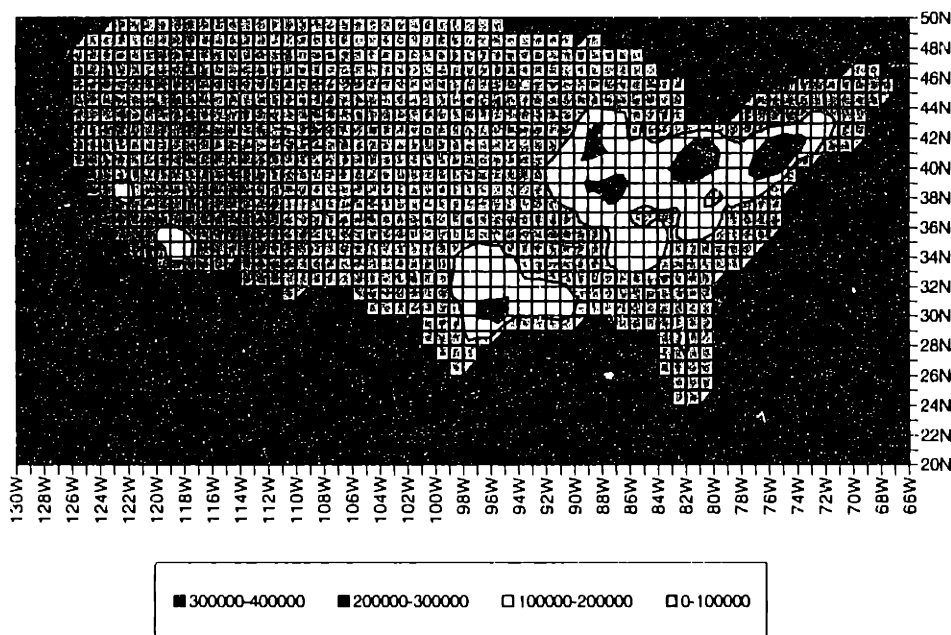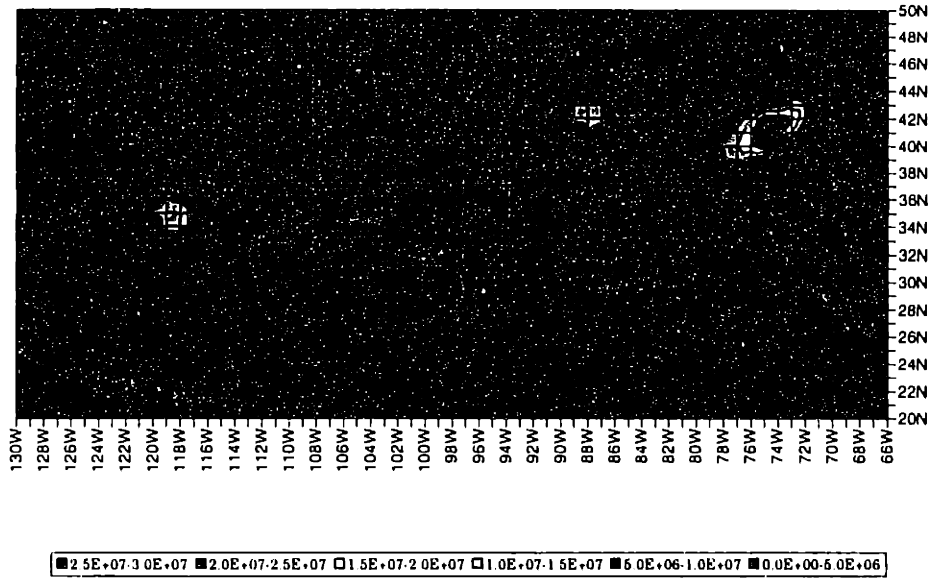■300000-400000   ■200000-300000   ☐100000-200000   ☐0-100000

Figure 5.3 Population Distribution for the United States
Each grid cell represents a total of nine GEIA 1° x 1° grid cells.

2 5E+07-3 0E+07 2.0E+07-2 5E+07 1 5E+07-2 0E+07 1.0E+07-1 5E+07 5 0E+06-1.0E+07 0.0E+00-5 0E+06

Population density provides and indication of urban agglomerations. In the United States, highly populated areas cover the Eastern seaboard, Southern California, and the Chicago area. The major source of air pollution, CO, NO_x, VOC, in urban areas is transportation (Figure 5.4). It is precisely high concentrations of these emissions that create ozone and other photochemical pollutants.

Reviewing the NO_x emission distribution, the high concentration areas are well-correlated with population and also cover the Eastern seaboard, Southern California and the Southeast coast of Texas. On average for the United States, NO_x emissions are roughly 50% from transportation sources and 50% from stationary combustion sources. Therefore, both the industrial areas and the urban agglomerations are highly concentrated with NO_x emissions.

Figure 5.4 Transport Emissions for the United States by Regional Scale
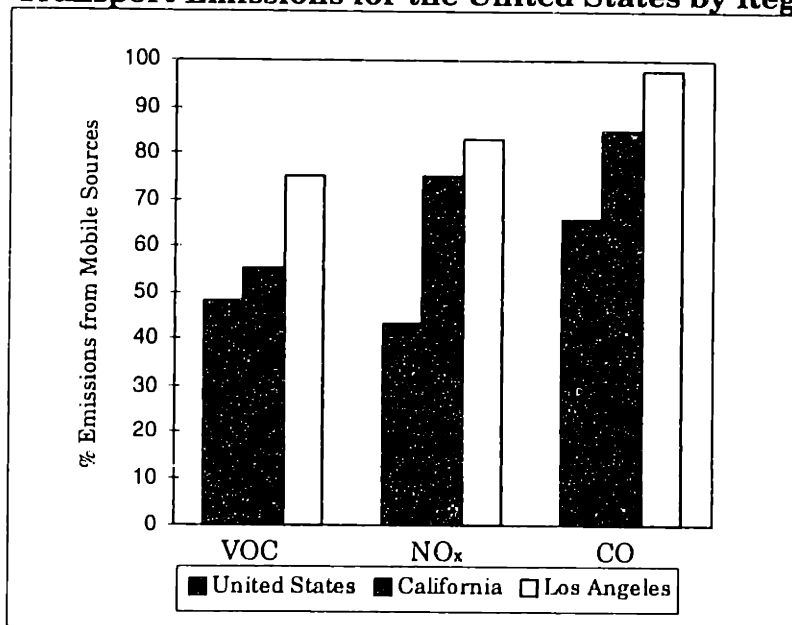
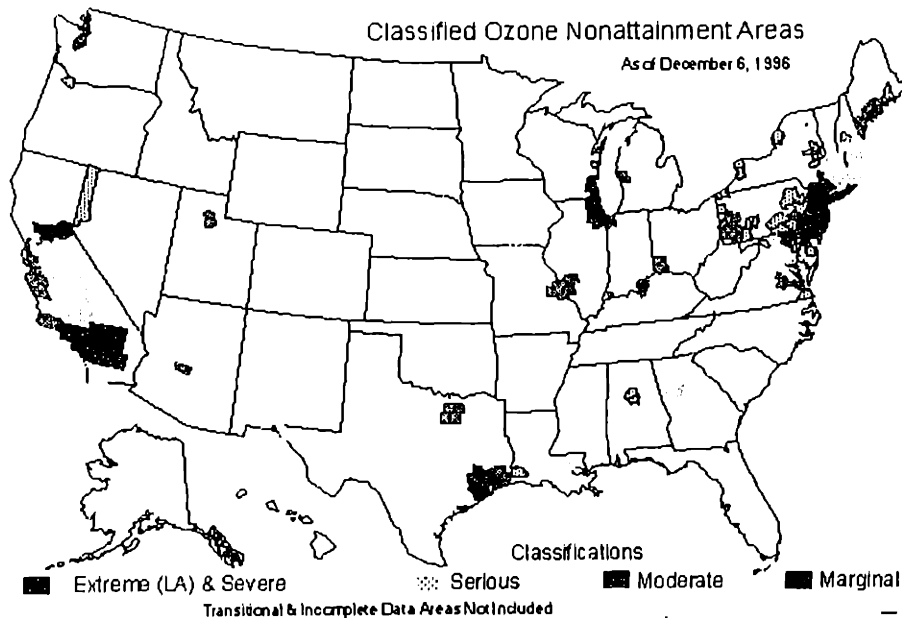Source: CARB (1993), EPA(1995)

For this analysis, population is used as an indicator of megacities (Table 5.1). To facilitate discussion, the megacities are referred to by the largest urban agglomeration enveloped; however, the megacity is not limited to any one metropolitan area because it encompasses a region 300 km². The megacities defined under this scenario overlap with the megacities defined using NO_x emissions as an example.

While the primary emissions may not be the highest of the nation, it is in these areas that the highly concentrated emissions react in the atmosphere producing secondary pollutants, such as ozone. The ozone non-attainment areas in the United States, those air quality districts that do not meet the National Ambient Air Quality Standard of 0.12 ppm ozone hourly maximum, are highly correlated to the population distribution (Figure 5.5).

Table 5.1 U.S. Megacities determined using population as indicator.

New York City	San Francisco
Los Angeles	Jacksonville
Chicago	Indianapolis
Boston	Dallas
Philadelphia	Atlanta

Figure 5.5   Ozone Non-attainment Areas in United States as of 12/96

(EPA,1997)

Classified Ozone Nonattainment Areas

As of December 6, 1996

Classifications

■ Extreme (LA) & Severe   ::: Serious   ■ Moderate   ■ Marginal

Transitional & Incomplete Data Areas Not Included

However, exceptions do exist. For example, the southeastern coast of Texas, the Houston area, is in ozone non-attainment, and this region is not visible as one of the top ten areas when using population as an indicator. The high ozone concentrations in this area are due in a large degree to its high natural VOC emissions (Figure 5.6). Natural VOCs are emitted from biogenic sources such as plants, and increase the reactivity of the atmosphere.

5.2 Scenario Analysis

Once the megacities are chosen, the U.S. emissions of NO_x, CO, VOC and SO_2 are forecast until the year 2050 under three different policy scenarios: a reference case, a CO_2 stabilization policy, and an energy tax (Table 5.1, Figure 5.7). These forecasts are

Figure 5.6   Natural VOC Emissions for the United States
Each grid cell represents a total of nine GEIA 1° x 1° grid cells.
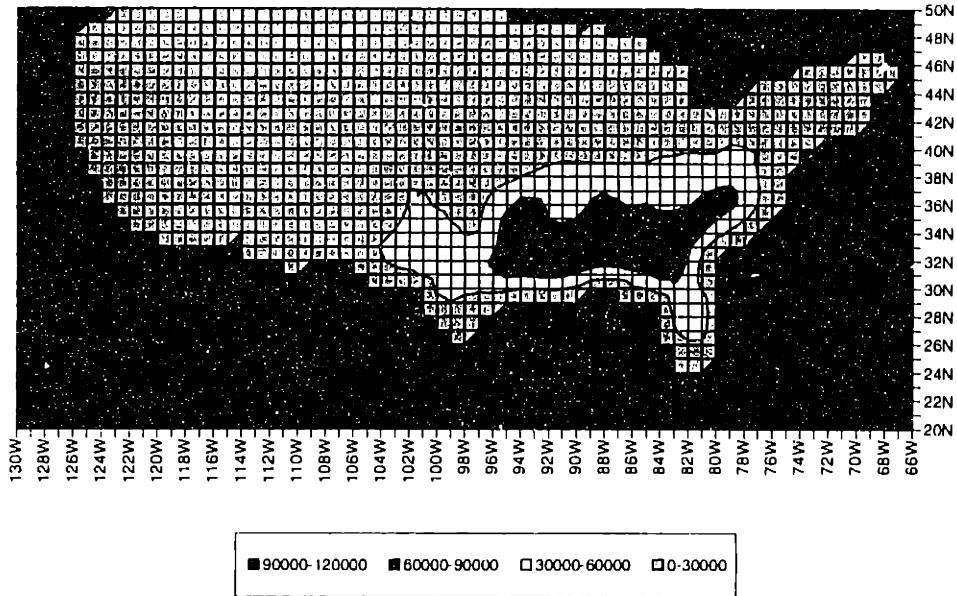
developed using dynamic emission factors by sector and fuel and the energy use and elasticity factors related to energy-intensive sector outputs from the EPPA model (Figure 5.8).

The emissions factors, or the amount of pollutant emitted per unit fuel burned, are decreased over time to account for improved technology and regulation. In the United States, the Clean Air Act Amendments of 1990 (CAAA 1990) regulate air quality using health-based standards, performance standards, technology standards, and tradable permits. These regulatory efforts have restrained the amount of per unit emissions and are expected to limit future emission rates (EPA, 1995).

Table 5.1 CO₂ Reduction Policy Scenarios

Reference Case: The reference case represents the air pollutant emissions in the absence of climate policy. This case provides the basis for comparison for the climate change policy cases.

AOSIS Case: This case represents a CO_2 stabilization policy and is modeled after the proposal of the Alliance for Small Island States (AOSIS). The actual protocol proposes a commitment by Annex I parties[24] to reduce CO_2 emissions by 20% below 1990 levels by 2005 (UNFCCC, 1995). In this model's version of AOSIS, 1990 levels of CO_2 emissions are cut by 10% by 2005 and again cut by 10% by 2010 and held thereafter. These constraints are placed on the OECD countries only.

TAX Case: This scenario imposes a 20% BTU tax in 1990. This constraint is placed on the OECD countries only.
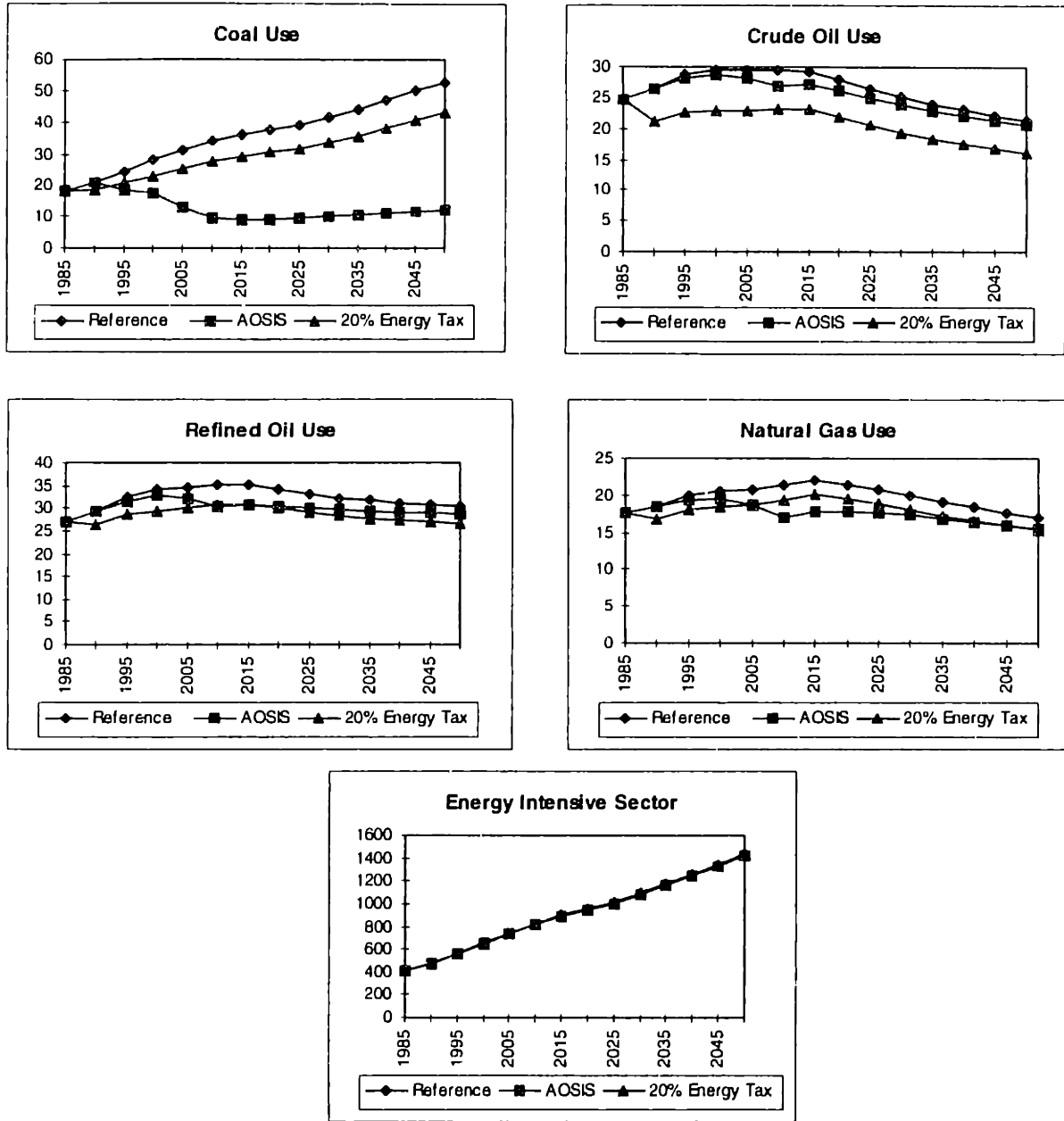
These emission forecasts show that the emission reductions relative to the Reference Case differ between the AOSIS Case and the TAX Case and this difference changes over time. This change is because each policy impacts different fuel uses and hence results in different energy use scenarios. The AOSIS Case, which is essentially a carbon tax, impacts coal consumption, because coal has the highest carbon content (Table 5.3).

Table 5.3 Coefficients of Carbon Content (million ton/exajoule)

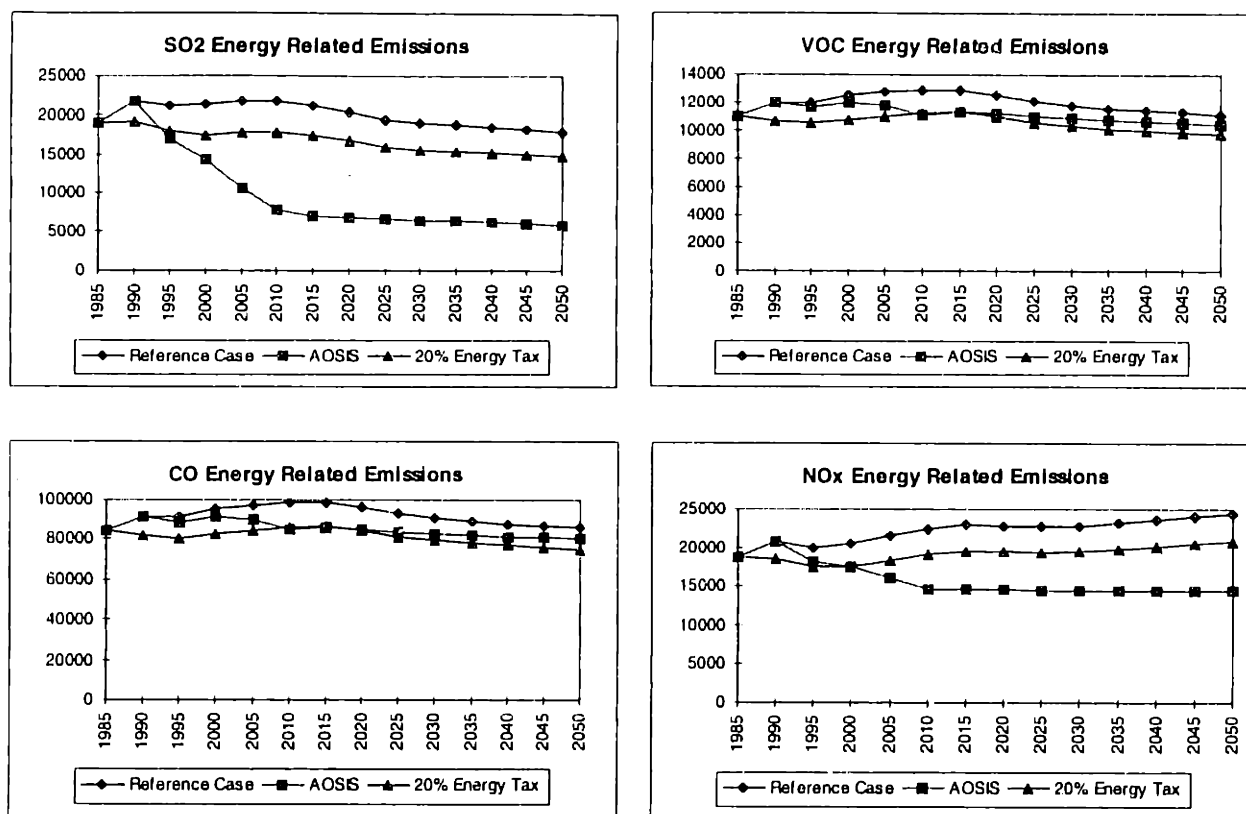| Fuel	Carbon Content
Coal	24.686
Oil	20.730
Gas	13.473
Refined Oil	20.730

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[24] ANNEX I includes OECD countries, the Former Soviet Union, and Eastern Europe

Figure 5.7 EPPA Energy Use Forecast for the United States

Coal Use

(line chart, x-axis 1985–2045, y-axis 0–60; series: Reference, AOSIS, 20% Energy Tax)

Crude Oil Use

(line chart, x-axis 1985–2045, y-axis 0–30; series: Reference, AOSIS, 20% Energy Tax)

Refined Oil Use

(line chart, x-axis 1985–2045, y-axis 0–40; series: Reference, AOSIS, 20% Energy Tax)

Natural Gas Use

(line chart, x-axis 1985–2045, y-axis 0–25; series: Reference, AOSIS, 20% Energy Tax)

Energy Intensive Sector

(line chart, x-axis 1985–2045, y-axis 0–1600; series: Reference, AOSIS, 20% Energy Tax)

Therefore, pollutants emitted during coal combustion, in addition to CO_2, are reduced. Averaging across sectors, emissions per energy quantity of coal are 840 g SO_2/GJ, 303 g NO_x, /GJ, 25 g CO/GJ, and 2 g VOC/ GJ. Given this, SO_2 emissions are hardest hit under this policy. In contrast, the TAX Case impacts the use of fuels with the relatively high energy content, such as refined oil. Because 68% of refined oil is consumed by the transport sector, these activities and their related emissions are reduced under such a policy. These related emissions include NO_x, VOC, and CO.

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Figure 5.8 Primary Energy Related Emission Forecasts for United States

SO2 Energy Related Emissions

—◆— Reference Case —■— AOSIS —▲— 20% Energy Tax

VOC Energy Related Emissions

—◆— Reference Case —■— AOSIS —▲— 20% Energy Tax

CO Energy Related Emissions

—◆— Reference Case —■— AOSIS —▲— 20% Energy Tax

NOx Energy Related Emissions

—◆— Reference Case —■— AOSIS —▲— 20% Energy Tax

5.3   Implementing Policy in Megacities

In the investigation of the air quality effects of these climate change policies, it is neces-sary to choose a year for the analysis. The year 2000 is chosen in this study for four rea-sons. First, it is hypothesized that the benefits of climate change policies will accrue in the near-term as opposed to the long-term benefits of reducing the threat of climate change. Second, in the emission forecasts, the NO_x emission reductions are roughly similar for the two policies as are CO_2 emission reductions. Given the importance of the relationship of NO_x and VOCs in the atmosphere, holding one of the variables constant provides an examination of this relationship. Third, the year 2000 is chosen in order to be consistent with existing secondary benefit studies which choose this year for analysis. Fourth, this choice minimizes the problem of knowing where megacities and emissions are. The percent changes in emissions for the different policy cases compared to the Ref-erence case for the year 2000 are shown in Table 5.4.

Table 5.4 Percent Change in Emissions from the Reference Case in 2000

POLICY	SO_2	CO	VOC	NO_x
AOSIS	-30.0	-2.1	-3.9	-14.2
TAX	-17.2	-7.2	-12.4	-14.4

These regional emission reductions, consistent with other studies, must be distributed to the megacities. As discussed in Chapter 3, the emissions are forecast maintaining a constant region to megacity emission ratio as determined by the 1985 base year inventories. The impacts of policy implementation, measured in emission reduction of primary pollutants, are determined for the regions using EPPA. These emission reductions are applied uniformly to the megacities. Therefore, each megacity experiences identical emission reductions.

Given the annual megacity emissions, it is necessary to disaggregate these emissions over the year to reflect seasonal patterns in energy use. For this study, the emissions are simply divided by 365 to produce daily emissions. For the purposes of investigating the secondary benefits of CO_2 reduction policies, it is appropriate to first focus on a period of high pollutant concentrations or a pollution episode, usually a day or series of days. This approach is used by the EPA to investigate the effects of air pollution control policies. The underlying assumption is that if air quality compliance standards can be met under the worst case scenario, they will be achieved under all other scenarios (NRC,1991).

To simulate a pollution episode, the meteorological conditions where chosen which favor high pollutant concentrations: low mixing layer height, low wind speed, high temperature, and low cloud cover. Under these meteorological conditions and the megacity emission rates, elevated concentrations of primary and secondary pollutants are likely. These scenarios provide an indication of the possible secondary air quality effects.

However, given the complexity of atmospheric chemistry, evaluation of annual secondary benefits requires simulation over an entire year. It is possible to generate concentrations and fluxes of primary and secondary pollutants for a given megacity over an entire year on a day-to-day basis using the Metamodel. The Metamodel is run continuously for 365 days with variable meteorological and emission parameters. The concentrations at the end of day 1 are used to set the initial conditions for the day 2 and so on.

The Reference Case emissions for the ten U.S. megacities shows that emission rates vary considerably across megacities (Table 5.5). SO_2 emissions range from 1.4 to 19.8; CO range from 103.8 to 190.4; VOC range from 13.1 to 43.8; and NO_x range from 7.9 to 21.9 kg/km²/day.

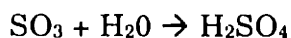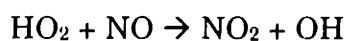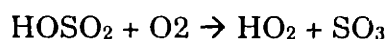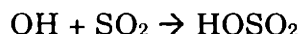### Table 5.5 Reference Case Emissions: Year 2000 (kg/km²/day)

	Megacity	CO	SO₂	VOC	NOₓ
1	New York City	12.1	190.4	43.8	21.7
2	Los Angeles	1.6	128.5	23.0	11.5
3	Chicago	12.7	159.7	24.0	16.0
4	Boston	8.3	129.6	18.1	10.3
5	Philadelphia	9.4	136.4	13.1	12.2
6	San Francisco	1.4	116.1	16.8	8.8
7	Jacksonville	5.8	103.8	12.5	7.9
8	Indianapolis	19.8	149.5	16.7	14.5
9	Dallas	5.0	132.4	17.7	12.6
10	Atlanta	16.3	120.5	17.0	10.2

5.4 Resulting Pollutant Concentrations

As noted earlier, existing literature on secondary benefits of CO_2 reduction policy translates primary emissions directly to monetized benefits. In contrast, this study presents air quality indexes by pollutant concentrations. The results are presented per primary pollutant to show the inadequacies of using emission reductions as a proxy for the benefits. The results highlight the need to examine concentrations as opposed to emissions, and therefore to investigate megacities as opposed to aggregating emissions across a nation. As described in Chapter 2 (Section 2.3), this study uses a reduced-form atmospheric model, the Metamodel, for this purpose.

5.4.1 Sulfur Dioxide

SO_2 emission reductions are around 17.2% and 30% for the TAX case and the AOSIS case respectively. SO_2 concentrations decrease slightly less than the emission reductions with slight variation across cities (Figure 5.9). This variation is expected because SO_2 is not highly reactive in the troposphere. However, SO_2 does react with the hydroxyl radical to produce SO_3 and H_2SO_4:

$$OH + SO_2 \rightarrow HOSO_2$$
$$HOSO_2 + O2 \rightarrow HO_2 + SO_3$$
$$HO_2 + NO \rightarrow NO_2 + OH$$
$$SO_3 + H_20 \rightarrow H_2SO_4$$

Given these relationships, H_2SO_4 concentrations are expected to diminish with SO_2 reductions. This is indeed the case (Figure 5.10).

While the percentage decrease in emissions is relatively constant, the secondary benefits are not. The level of benefit depends on the absolute magnitude of the change in concentration which varies from megacity to megacity depending on initial concentrations. The same percentage change in concentration results in magnitude changes that range from 0.0057 ppm (14.25ug/m) to 0.0005 ppm (1.25ug/m) (Figure 5.11).

Figure 5.9 Percent Change in SO₂ Concentrations:
Policy Scenarios Compared with Reference Case in 2000
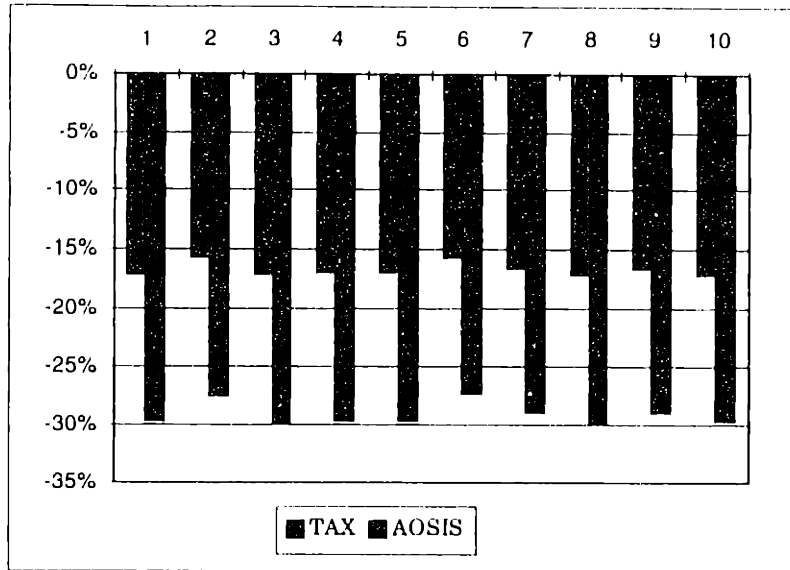
Figure 5.10 Percent Change in SO₂ Deposition:
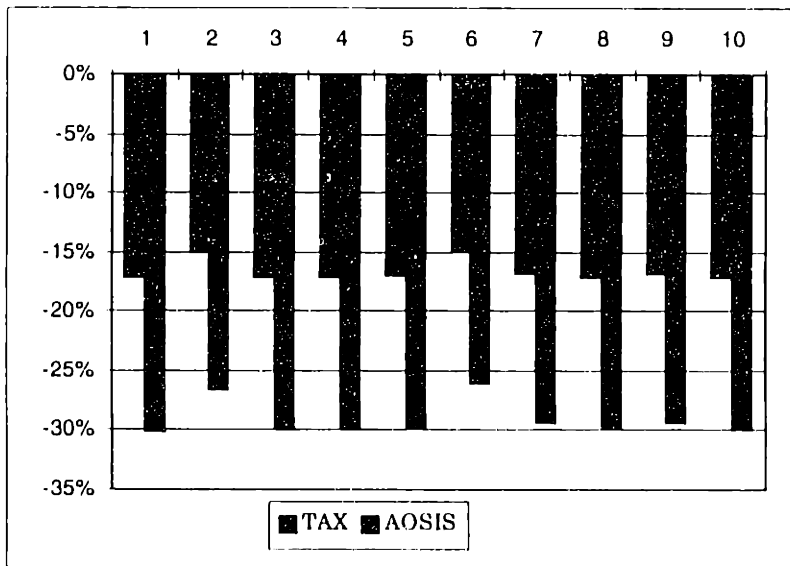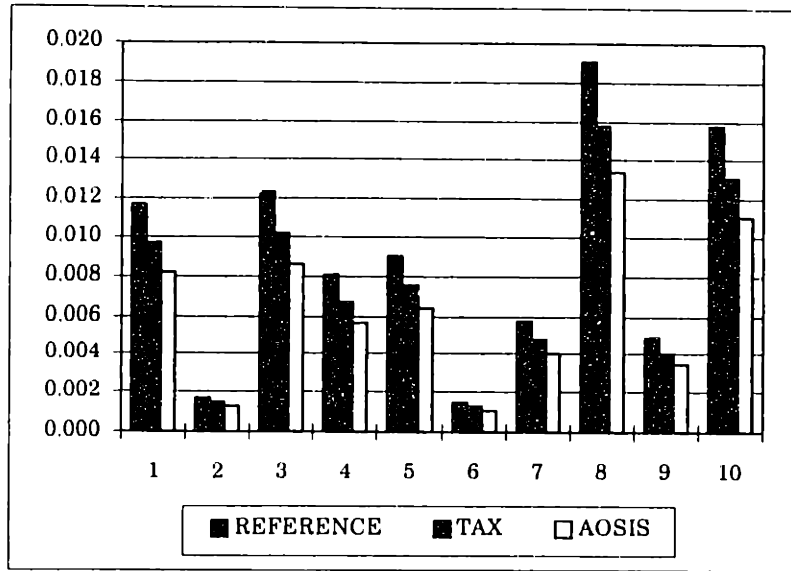**Policy Scenarios Compared with Reference Case in 2000**

**Figure 5.11 SO₂ Concentrations for 10 U.S. Megacities**

5.4.2 Carbon Monoxide

Similar to the SO₂ emission-concentration relationship, CO concentrations decrease by roughly the same percentages as the decrease in emissions (Figure 5.12). Recall the TAX case and the AOSIS case reduce CO emissions by 7.2% and 2.1% respectively.

Figure 5.12 Percent Change in CO Concentrations
Policy Scenarios Compared with Reference Case
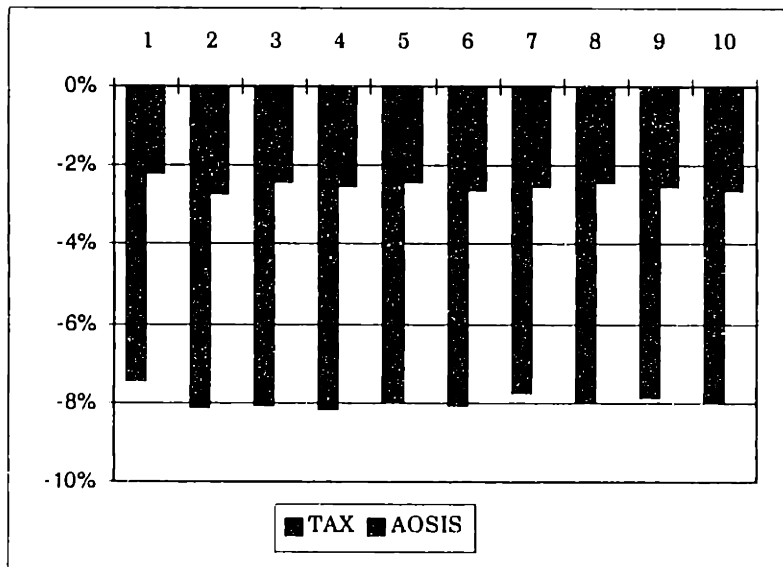
A slight variation in the CO concentration-emission ratio exists. CO is both created and destroyed in the troposphere. Methane and non-methane hydrocarbons react to form CO. At the same time, CO does react with OH to form CO_2. HO_2 is then reactive with NO to form NO_2 and to regenerate the OH radical and hence the potential to react further.

$$CO + OH \rightarrow CO_2 + HO_2$$
$$HO_2 + NO \rightarrow NO_2 + OH$$
$$OH + NO_2 \rightarrow HNO_3$$
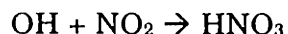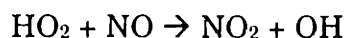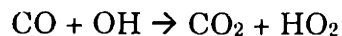
The variation between megacities can be explained by differing emission rates and pollutant mix. Due to the chemical reactions between pollutants, determining the cause of regional variability on a pollutant by pollutant basis is impossible. It is necessary to examine each city in context of baseline concentrations, emission rates, and meteorology.

5.4.3 Nitrogen Oxides

The percent changes in NO_x concentrations given the same emission reduction vary with policy and among megacities (Figure 5.13).

Figure 5.13   Percent Change in NO_x Concentrations Policy Scenarios Compared with Reference Case
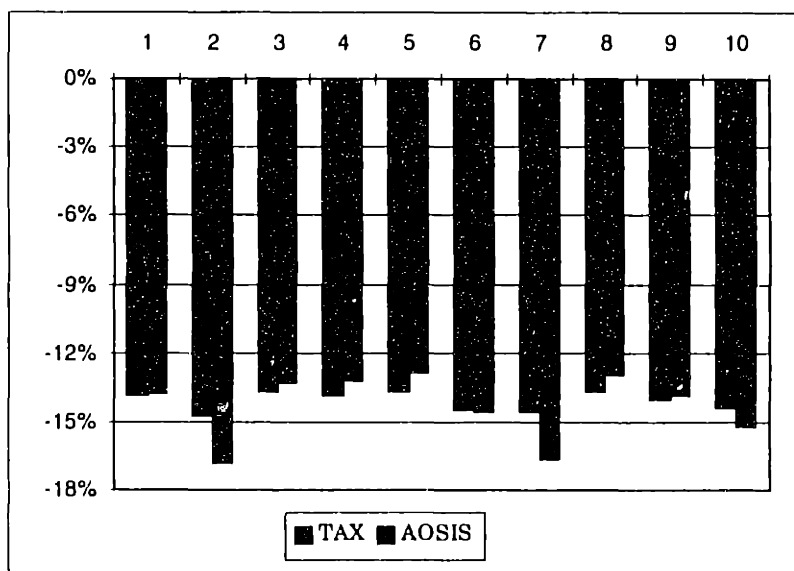
As discussed in Chapter 2, NO_x and VOC concentrations regulate critical aspects of the overall air quality. These species contribute to a number of secondary pollutants including ozone, nitric acid, PAN, and nitric aerosols. The relationship between primary emissions and secondary emissions is complex and depends on the specifics of the region, however relationships do exist and are discussed in the following section. For Los Angeles, it appears that ozone and PAN are reacting in similar ways (Figure 5.14). In addition, in all cases HNO_3, NO_x, and CO decrease with primary emission decreases.

**Figure 5.14 Percent Change in Pollutant Concentrations:
Summer Day, Los Angeles, 2000**
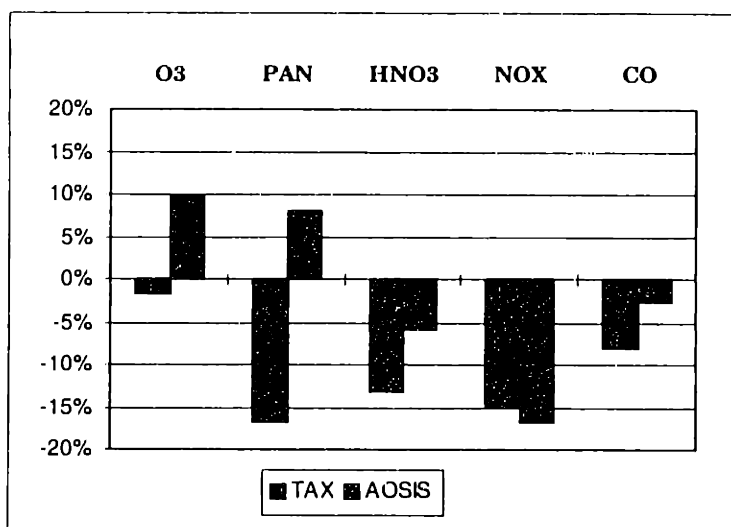
Photochemical Oxidants
Ozone

Controlling NO_x and VOC emissions may result in either an increase or decrease in ozone accumulation. This increase or decrease in ozone depends on the initial concentration levels and emission rates, and the reductions of NO_x emissions relative to the reductions in VOC emissions. This relationship between NO_x, VOC, and ozone can be graphically represented showing the ozone concentrations at different NO_x and VOC concentrations, known as ozone isopleths. Ozone isopleths are tools used by policy analysts to determine the possible outcomes of different control measures. Figure 5.15 shows an ozone isopleth generated using the Metamodel for "New York City"[25] during a pollution episode. Recall from the ozone isopleths presented in Chapter 2 (Figure 2.3)

that ozone isopleths are c-shaped curves. Therefore, given the characteristics of this region for this scenario, the regime of ozone concentrations is the upper-right hand portion of a typical ozone isopleth. It is in this regime that decreasing NOx without changing VOCs results in a increase in ozone.

In this graph, Point I represents the initial conditions at 2000 for the reference case. Point L represents the lowest ozone concentrations. Ozone increases on the contours towards a maximum at Point H. Any movement away from point I is achieved through emission reductions. This graph illustrates how a decrease in NOx alone causes an increase in ozone and a decrease in VOC alone causes a decrease in ozone. The combination of VOC/ NOx control determined by the policy determines whether ozone increases or decreases.

Figure 5.15 Ozone Isopleth for NYC
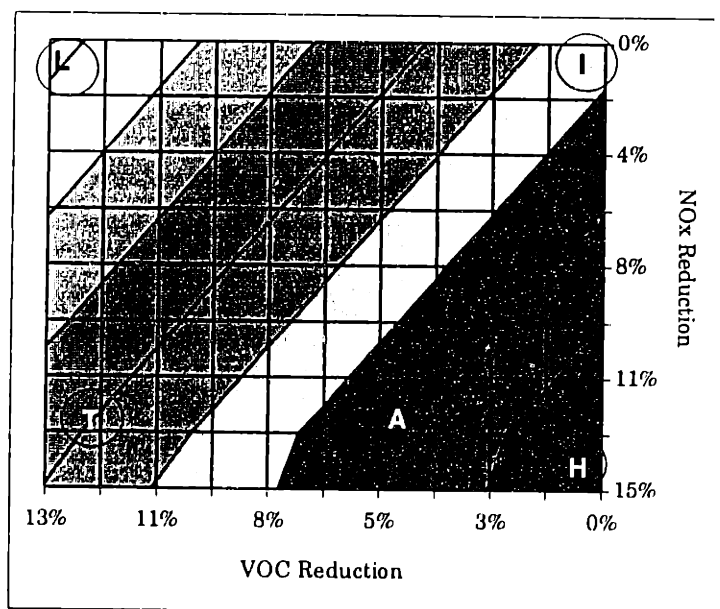
For example, Point A is the resulting ozone under the AOSIS Case, reductions in NOx and VOC emissions. The relative reductions in these precursor emissions lead to an increase in ozone concentrations. Point T is the resulting ozone under the TAX Case. Here, the relative reduction yields a decrease in ozone concentrations. Due to

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[25] Ozone isopleths vary for different regions within an air shed (Milford, 1989).

atmospheric reactions, a linear relationship between precursor emissions and secondary pollutant concentrations does not exist.

This relationship poses considerable challenges to a number of cities which are in non-attainment with the NAAQS. To complicate the matter further, the ozone concentration varies throughout the day because its formation is dependent on sunlight and precursor emissions, both of which follow a daily cycle (Recall Figure 2.5). In addition, uncontrollable characteristics of an area, such as meteorology and biogenic VOC emissions influence the creation of photochemical oxidants.

Due to a boundary condition problem in the metamodel, the prediction of ozone concentrations are currently not available. Therefore, this analysis relies on the underlying model to investigate the impact on ozone concentrations. Hourly maximum ozone concentrations for Los Angeles on a summer day for a pollution episode allow the exploration of ozone responses (Figure 5.16). For the time of peak ozone, implementing the AOSIS policy actually increases ozone concentrations. Policies can also shift the time of the peak ozone which in turn can change the magnitude of the population exposed. While the percentage increase appear small, considerable amounts of resources are allocated based on the potential to decrease ozone concentrations 1-2%.

**Figure 5.16 Ozone Concentrations : Summer Day, Los Angeles, 2000**
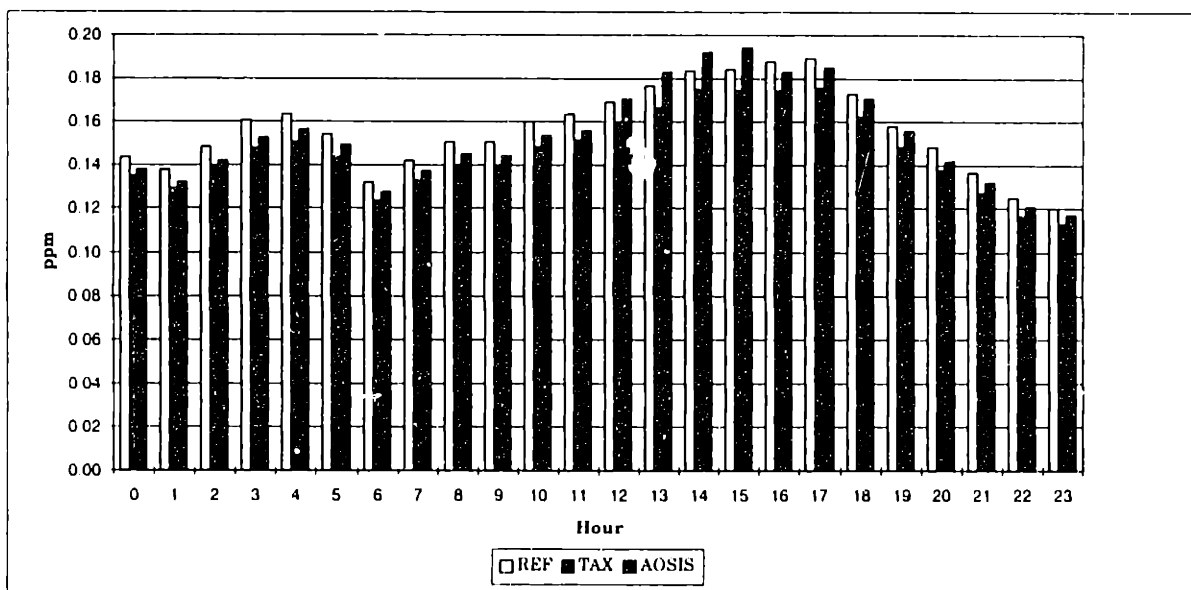
Other Photochemical Oxidants

Figures 5.17 and 5.18 show the PAN mass in the air shed and the flux out of the air shed respectively. Under the TAX there is a decrease in PAN formation and an increase under AOSIS for LA megacity region. These figures highlight the daily variation of the pollutant concentration due to daily emission and meteorological patterns.

The response patterns for PAN and ozone are similar (Figure 5.14). For this day in Los Angeles ozone and PAN increase under AOSIS and decrease under TAX. Both PAN and ozone depend on availability of VOCs and NO_x for their production. When PAN concentrations decrease as a result of increasing NO_x, as is the case under AOSIS, this leads to an increased rate at which VOCs decrease to OH via NO oxidation and therefore HNO_3 production is favored over PAN production. The HNO_3 flux out does not improve under AOSIS, but does under TAX. The mass in of HNO_3 in the air shed decreases under both AOSIS and TAX (Figure 5.19 and 5.20).

While NO_3- is not an output from this model, the level of O_3 gives an indication of how this pollutant would respond under different primary pollutant reductions. When ozone is decreased or ozone and NO_2 are reduced, lower concentrations of NO_3 and N_2O_5 could be expected (NRC,1991) due to the following reactions:

$$NO_2 + O_3 \rightarrow NO_3 + O_3$$
$$NO_3 + NO_2 \rightarrow N_2O_5$$
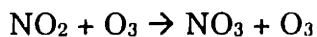
This reduction may result in a decrease in HNO_3 production because it is a product of a reaction between NO_3 and VOCs. However, there are a number of different reactions which yield HNO_3.

Figure 5.17 PAN Mass
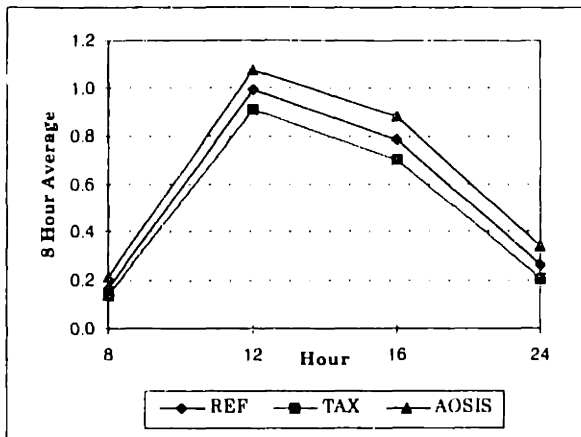
Figure 5.18 PAN Flux
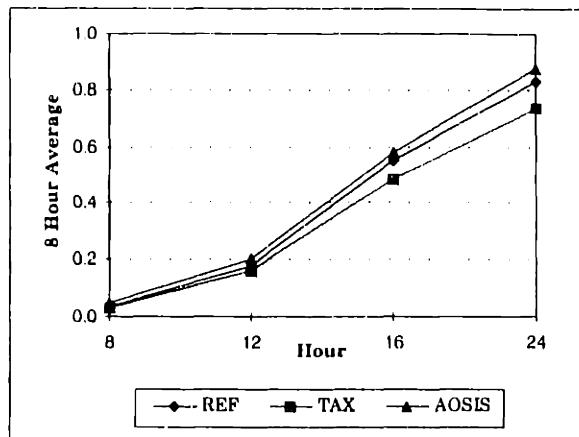
Figure 5.19 HNO₃ Mass
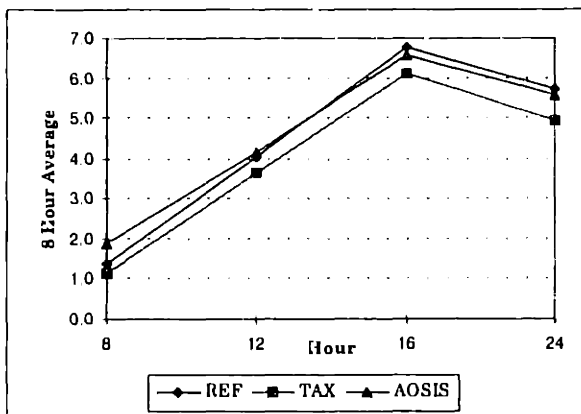
Figure 5.20 HNO₃ Flux
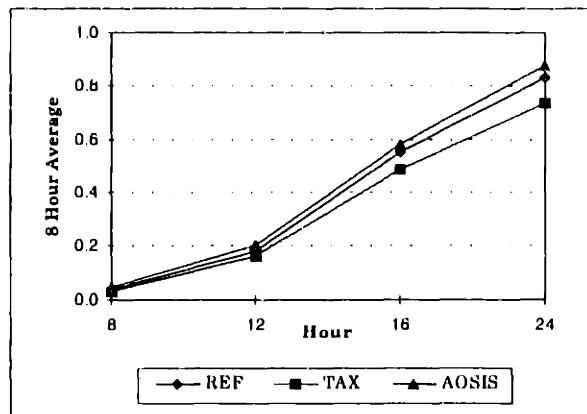
5.3.5 Volatile Organic Compounds

VOCs are highly reactive organic compounds and lead to the formation of ozone and PAN and acid aerosols. In addition, organics in the atmosphere such as formaldehyde, HCHO, have serious health effects, including carcinogenic effects. This study does not examine VOC concentrations, however a number of additional species can be determined using the Metamodel. It is important to consider these pollutants given their health impacts. Therefore, regardless of VOC's impact on photochemical pollutants, it is necessary to limit their output.

5.5 Determining Benefits to Human Health

Once the air pollutant concentrations under different policies are determined, the health effects can be quantified using the approach outlined in Chapter 4. The health impacts of sulfur dioxide, nitrogen oxide and ozone are presented for the polluted day in Los Angeles. The health effects of carbon monoxide are not presented because dose-response relationships do not currently exist. The health effects of particulate matter, which are expected to be great, are not presented because currently this model does not incorporate PM emissions.

High and low estimates are provided to explicitly characterize the range in which these health effects lie. The difference between the two policies are the difference between costs and benefits in the case of ozone, as ozone concentrations increase under AOSIS and decrease under TAX. The effects from nitrogen oxide given the uncertainty fall within the same bounds, roughly a decrease of 623 to 1697 respiratory symptoms for this day. The sulfur dioxide effects are relatively low, because the base line sulfur dioxide level is low.

Table 5.6 Health Benefits for Polluted Day in Los Angeles in 2000[26]

a) Sulfur Dioxide: AOSIS = -1.363 $\mu g/m^3$ Change in Average
TAX = -0.777 $\mu g/m^3$ Change in Average

Health Effect	Policy	High Estimate	Central Estimate	Low Estimate
Mortality	AOSIS	-0.05%	-0.02%	-0.01%
	TAX	-0.03%	-0.01%	0.00%
Respiratory Symptoms	AOSIS	-0.40	-0.28	-0.15
	TAX	-0.23	-0.16	-0.09
Chest Discomfort	AOSIS	-630	-420	-210
	TAX	-359	-240	-120

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[26] Assuming a total population of 15,327,700 (GEIA) with 27% children, 73% adults (US Census), and 5% asthmatics (Oslo, 1994).

b) **Ozone :** AOSIS = 0.004 ppm Change in 1-Hour Maximum
TAX = -0.014 ppm Change in 1-Hour Maximum

Health Effect	Policy	High Estimate	Central Estimate	Low Estimate
Hospital Admissions	AOSIS	2.02	1.29	0.64
	TAX	-7.05	-4.53	-2.23
Restrictions in Activity	AOSIS	8,567	5,711	2,856
	TAX	-29,983	-19,989	-9,994
Respiratory Symptoms	AOSIS	16,266	9,197	4,722
	TAX	-56,792	-32,188	-16,526
Eye Irritation	AOSIS	3,666	3,262	2,869
	TAX	-12,832	-11,416	-10,043
Asthma Exacerbation	AOSIS	1,594	575	325
	TAX	-5,579	-2,012	-1,137

c) **Nitrogen Oxides :** AOSIS = -4.803 µg/m³ Change in Average
TAX = -4.227 µg/m³ Change in Average

Health Effect	Policy	High Estimate	Central Estimate	Low Estimate
Respiratory Symptoms	AOSIS	-1697	-1203	-708
	TAX	-1493	-1058	-623

In order to highlight the regional difference and the importance of base line air quality, the health benefits from reducing sulfur dioxide in Indianapolis are presented. Because Indianapolis' initial air quality is higher for this pollutant, the identical reductions result in a larger absolute magnitude of pollutant concentrations. This difference resolves itself when comparing the magnitude of health benefits between the two megacities.

Table 5.7 Health Benefits for Polluted Day in Indianapolis[27] in 2000

a) Sulfur Dioxide   AOSIS: -16.251 µg/m³ Change in Average
                    TAX: -9.301 µg/m³ Change in Average

Health Effect	Policy	High Estimate	Central Estimate	Low Estimate
Mortality	AOSIS	-0.54%	-0.21%	-0.09%
	TAX	-0.31%	-0.12%	0.05%
Respiratory Symptoms	AOSIS	-2.24	-1.55	0.86
	TAX	-1.28	-0.89	-0.49
Chest Discomfort	AOSIS	-3543	-2362	-1181
	TAX	-2028	-1352	-676

5.6   Sensitivity Analysis

In order to validate the responses of the Metamodel under these highly polluted conditions, the CIT air shed model, the underlying model to the Metamodel, is run for the "Los Angeles" case using Metamodel emission inputs. The outputs of these models are consistent with the exception of the magnitude of ozone concentrations. As discussed previously, the problem has been determined and is currently being corrected.

In order to carry out this task, a number of assumptions have been asserted. The parameter with the biggest effect on uncertainty in this analysis has been the base year VOC emissions rates. While reliable databases provide the VOC emission rates, these inventories have limitations. The difficulty in measuring VOC emissions and the role natural emissions play, which is highly seasonal, has led to underestimated inventories (NRC, 1991). Since, VOC's are an extremely important component in the photochemical reactions and hence, the concentration of the primary and secondary species in the atmosphere, the sensitivity of the results to this parameter are tested.

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[27] Assumptions: Population – 7,227,698 (GEIA), Children –27% (US Census)

The base case megacity emissions drive future megacity emission scenarios because the regional-to-megacity ratio is held constant. For this reason, the uncertain impact of the VOC base year assumptions is tested. Using the Los Angeles case, the upper bound on the possible VOC emission estimates is set at 31.84 kg/km^2/day. The NO$_x$ emissions are held constant in order to analyze the impact a variable NO$_x$-VOC emission ratio has on the concentration results. This analysis shows that the sign of the percent emission change in ozone, PAN, and HNO$_3$ remains positive or negative under a range of base year VOC emission rates, and the magnitude varies only slightly. Ozone, PAN, and HNO$_3$ increases uniformly with increasing base line VOC emissions. To the contrary, NO$_x$ decreases. There is no perceptible change in emissions of concentrations of CO, SO$_2$, or SO$_2$ deposition.

CHAPTER 6

CONCLUSION

6.1 Policy Implications

This thesis develops a method to assess the potential air quality improvements that result from changes in air pollutant emissions associated with policies intended to reduce carbon dioxide emissions. A global emissions prediction and policy analysis model is integrated with an urban-scale atmospheric chemistry model to determine ambient pollutant concentrations. Concentrations, as opposed to emission rates, are utilized to determine the impact on air quality. Applying this method to urban agglomerations or megacities in the United States, findings suggest that existing claims heralding secondary benefits of the same order of magnitude as the intended benefits, may not only be overstated, but also overlook possible adverse effects. While primary pollutants decrease with reduced pollutant emissions, secondary pollutants such as ozone and PAN may increase rather than decrease, damaging human and environmental health in addition to raising the compliance challenges faced by urban areas. The policy implications are that, given the potential for limited benefits or costly consequences, flexibility in policy design is needed to address this uncertainty.

The possibility of increasing secondary pollutants with reductions in primary emissions is a challenge inherent in the management of urban air quality. Reducing ozone concentrations has been an elusive goal for a number of urban areas despite strict emission controls and technology measures (Zegras, 1995). However, the secondary pollutants are not limited to ozone and PAN and include hazardous pollutants and toxic substances, which may respond positively to a decrease in primary pollutant emissions. Because it is

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impossible to monitor the entire field of pollutants and determine their respective adverse impact on humans and ecosystems, pollutant emission reductions are viewed positively.

However, while the aggregate decrease in concentrations may improve air quality, increases in ozone and PAN may have a considerable impact on the public health of exposed populations and burden the urban region with additional compliance costs and stricter environmental regulations. For example, under the NAAQS, failure to attain ozone standards is costly and may result in additional percent reduction requirements and excess fee penalties. Currently, a number of cities, such as Los Angeles, are spending considerable resources to develop control strategies which promise even small reductions in ozone. The success of local air pollution policies may suffer if climate change mitigation policies are not considered in the design of such strategies. Decision-making on local air quality should incorporate flexibility and the potential impact of climate change mitigation policy on their region to the extent possible. For example, if climate policy reduces NO_x emissions while having little impact on VOC emissions, the local control strategy may focus on reducing VOC emissions. This calls for an integrated approach to managing local, regional, and global problems.

This study suggests that the limitations in the existing models used to assess secondary benefits may have led to an overstatement of benefits and directed attention away from the possibility of adverse effects. Evaluating benefits based on national aggregate emission rates overlooks the potential inconsistencies and trade-offs between climate change policies and air quality. While it is necessary to determine the impacts global policies will have on the local level, it is also essential to incorporate the secondary benefit analysis into the assessment of costs and benefits of climate change policies. These objectives require additional research into the secondary air quality benefits.

6.2 Recommendations for Further Research

The approach developed for this thesis represents a first step towards incorporating the analysis of secondary air quality benefits into an integrated assessment model. There-

fore, the method presented can benefit from improvements in both the models employed and the assumptions relied upon. Certain aspects of the EPPA model and the Metamodel need to be addressed in order to accommodate the needs of secondary benefit studies. In addition, the validity of the governing assumptions used to integrate these models should be examined by conducting a sensitivity analysis on the uncertain variables.

Metamodel

While the Metamodel adequately simulates the output parameters of the CIT model, the prediction of ozone concentrations while qualitatively similar, is quantitatively different for high pollution conditions. The cause of this discrepancy has been linked to the distribution of initial concentrations, and an improved distribution has been shown to yield outputs consistent with the underlying model. An effort to re-parameterize the Metamodel with the CIT model.

For further research, regional variations in pollutant concentrations across the domain would provide insight into the distributional impacts of the given policy. Currently, the Metamodel averages pollutant concentrations over the domain.

In addition, the Metamodel and climate model should be linked because of the relationships that exist between air pollutants, greenhouse gases, and meteorology (Chapter 2). The outputs of the climate model would incorporate temperature changes and wind velocities from potential climate change impacts into the megacity scenarios. Likewise, the Metamodel would provide for corrected pollutant fluxes into the global atmosphere. This coordination will improve the predictions of secondary benefits.

EPPA

The EPPA model is used in the prediction of energy use by fuel type and fuel sector under climate change policies. In order to fully benefit from the use of aggregate emission factors for the prediction of trace gases, the energy use in EPPA must be differentiated to include the transport sector.

In addition, this study develops aggregate emission factors only for the United States. It is necessary to develop these factors for the other eleven regions of the EPPA model. Difficulties will arise where existing country inventories are not as detailed and reliable as those of the United States and where regions incorporate more than one nation.

Region-to-Megacity Emission Distribution

As discussed in Chapter 3, the current distribution of regional emissions to megacities is accomplished by maintaining a constant megacity-to-region emission ratio. This method requires a base year inventory of pollutant emissions for the megacities within the region. Currently, the global databases of emission inventories include only NO_x, SO_2, natural VOC and population. Therefore, national databases have been relied upon in this study for VOC and CO emission inventories. Where these inventories are unreliable or non-existent, statistical relationships can estimate base year emissions from the megacity. Further methods for improvement have been discussed in Chapter 3 (3.2.2 *Distributing Regional Emissions to Megacities*).

Evaluation of benefits

Assigning values to human health benefits is a process which is laden with uncertainties and value judgments. Further epidemiological research on the causal links between human illness and pollutant concentrations will improve existing dose-response models and provide relationships where they are currently non-existent. In the absence of these models, attempts should be made to quantify these benefits in an alternative way so that they can be included in the policy analysis.

Scope of Pollutants

The scope of this analysis should be expanded in further research to investigate VOC concentration outputs and incorporate particulate matter emissions into the analysis. Currently, the Metamodel is capable of predicting VOC concentrations over the megacity domain, but these outputs are not investigated in this study. These pollutants contribute significantly to the air quality degradation and adverse human health effects. Inclusion in the analysis would provide a fuller picture of the climate change policy impacts on air quality.

Neither the EPPA nor the Metamodel incorporate particulate matter emissions, with the exception of acidic deposition at this time. Particulate matter contributes to increasing air pollution problems in megacities of the world, like Beijing and Shanghai, where coal combustion technologies do not have adequate emission controls. Decreasing these emissions is expected to contribute greatly to the level of secondary benefits.

Scope of Analysis

The analysis presented in this thesis investigates the impact of climate change policy on a given megacity for a day of a pollution episode. In order to determine the annual benefits for a given policy, it is necessary to run the model for a year. This requires meteorological fields and emission fields on a daily basis for a year for the different megacities of a region. Functions to simulate these parameters over the year must be defined.

A number of assumptions in this model are based on industrialized countries. However, as stated previously, the secondary benefits to air quality are likely to be highest in developing countries where baseline air quality and control technologies are low. Therefore, it is important to further develop the assumptions to address the characteristics specific to developing countries. With these improvements and increased scope, this method of analysis can provide further insight into the expected level of air quality impacts, the degree of regional variation, and inform policy-making on the local, national, and international level.

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