AIR POLLUTION, METEOROLOGY AND PUBLIC HEALTH IN NEW JERSEY: CASE STUDIES OF NEWARK AND THE MEADOWLANDS

by

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ABSTRACT OF THE DISSERTATION

Air Pollution, Meteorology and Public Health in New Jersey: Case Studies of Newark and the Meadowlands

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Air pollutant concentrations vary with local meteorological conditions and can have deleterious effects on human health. This dissertation adapts an integrated approach to improve understanding about the role of meteorological factors on air pollution concentrations and their cumulative effects on public health in Newark and the Meadowlands, New Jersey.

Filter-based samples of particulate matter (PM) were collected and meteorological variables were monitored in Newark from August to October 2009 and from December 2010 to January 2011. Measurements of daily maximum ground-level ozone (O_3) were obtained from Newark Firehouse, a monitoring station in the New Jersey Department network (NJDEP). Ambient concentrations of nitrogen oxides (NO_x) and O_3 were measured and meteorological variables were monitored at the Meadowlands Environmental Research Institute (MERI) from June 2007 to May 2008, to characterize the temporal and seasonal variations of gas-phase air pollutants. Health records of

respiratory hospital admissions were obtained from St. Michael's Medical Center in Newark and the New York State Department of Health and Senior Services (NJDHSS). Statistical analyses were conducted by using time series, multiple linear and principal component regression techniques.

The results show that ambient levels of $PM_{2.5}$ and O3 were influenced mainly by temperature and wind speed. Positive associations demonstrated among O3, PM and hospital admissions suggest that even below current federal standards, air pollutant levels could be associated with an increased risk for respiratory illnesses. Variability of NO_x and O_3 were altered by distinct atmospheric conditions and chemical inter-conversions of the pollutants. There was an inverse relationship between concentrations of NO_x and O_3; the latter was dominant in summer and specific time of the day (early afternoon). Seasonal variations of NO_x were less distinct with strong diurnal patterns of traffic-related peaks during the early morning rush hour. There was a strong association between NO_x and respiratory hospital admissions mainly in the winter season. For O_3, association with hospital admissions was strongest at 2 lag days. Both climate-induced and pollution-induced health effects of NO_x and O_3 suggest that current national standards may not adequately provide a safe threshold for air pollutants from a public health perspective.

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TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

LIST OF ILLUSTRATIONS

Chapter 1: Introduction

1.1 Overview

The natural composition of the atmosphere may be altered by dangerous substances or air pollutants in the form of particles and gases that come from both natural and anthropogenic sources. Air pollutants can have adverse effects on human health by aggravating pre-existing or chronic diseases, increasing susceptibility to infections, reduce lung function, inducing systemic inflammation and degrading the status of healthy individuals (Forastiere et al. 2005). Motor traffic is a leading source of air pollution in cities (Frumkin, 2002; Gurjar et al., 2008). Other major sources include fossil fuel combustion for electricity generation from power plants, industrial waste incinerators, cement plants, iron and steel mills, emissions of dust from urban construction and quarries, open burning at solid waste dumpsites and domestic heating.

Following their emission, a number of synoptic conditions influence the dispersion, secondary transformation and concentrations of air pollutants, particularly local meteorology to which pollution levels are inextricably linked (Vukovich et al., 2002; Triantaflyllou et al.; 2002; Hussein et al., 2006). The direct emission of primary pollutants such as nitrogen oxides (NO_x), contribute to the large-scale production of ground level ozone (O_3) and particulate matter (PM) that affect local air quality but they are also transported over long distances by prevailing winds (Gurjar et al, 2008). The highest pollution levels have been observed about 20 to 40 miles downwind from the main source (Frumkin, 2002). The public health impact of air pollution is better understood by studying the spatial effects and meteorological conditions on which pollutant levels depend (Varadarajan, 2007).

Variations in prevailing atmospheric conditions can result in corresponding differences in air pollution concentrations (Guttikunda and Gurjar, 2011). Since it is formed in the presence of heat and sunlight, O_3 is commonly referred to as a summertime air pollutant. The variations observed in PM concentrations are also strongly influenced by their emission source and subsequent interactions with meteorological variables on different time scales (Madsen and Motolla, 2003). In fact, PM is classified based on its aerodynamic properties which govern its transport, removal from the air and deposition into the respiratory system (Pope et al., 2002; Brook et al., 2004).

The respiratory system is the principal route of entry for air pollutants (Jamal et al., 2004). Thus, public health and scientific interest in air pollutants has resulted mainly from epidemiological studies. Previous research has found that hospital admissions for respiratory diseases may increase disproportionately on days with elevated air pollution levels (Frumkin, 2002, Jamal et al., 2004). Evidence of the relationship between air pollution levels and public health is, however, sparse in New Jersey. Knowledge of the mechanisms that enhance air pollution concentrations gained in this study can lead to better regulation and a novel target for minimizing respiratory and inflammatory diseases of the airways.

A number of hypotheses were tested in this dissertation to investigate the relationship between local meteorological conditions and air pollution concentrations as a plausible explanation of respiratory illnesses. The main premise is that, as precursors, weather variables contribute both to the production and variable concentrations of air pollutants, and provide evidence for the identification of respiratory health effect indicators. To this end, it demonstrates that an understanding of the relationship between meteorological parameters and pollutant levels may be a useful predictor of respiratory hospital admissions. The investigation was carried out in Newark and the Meadowlands, two metropolitan areas in New Jersey that are surrounded by numerous air pollution sources. The key pollutants, nitrogen oxides (NO_x), ground-level ozone (O_3) and fine particulate matter (PM) are criteria pollutants that are routinely monitored elsewhere and are known to have adverse effects on public health in US cities and major urban areas worldwide.

1.2. Research Objectives

There are three primary objectives of this study under which several specific objectives are listed as follows:

1. Evaluate the relationship between meteorological conditions and air pollution concentrations to account for respiratory health problems in Newark.
    a. Collect daily samples of fine and coarse size fractions of PM ($PM_{2.5}$ and $PM_{2.5\text{-}10}$) to estimate their mass concentrations with simultaneous rooftop observations of weather variables.

b. Monitor meteorological parameters to investigate their influence on PM and O_3 concentrations.

c. Obtain a record of daily maximum O_3 concentrations in close proximity to the PM sampling station to assess dependence on meteorological parameters and possible influence on daily respiratory hospital admissions.

d. Analyze the cumulative effects of variable weather conditions and air pollution levels by comparing the record of total daily respiratory admissions from a nearby hospital.

2. Characterize the relationships among ambient NO_x and O_3 concentrations and meteorological factors to identify their seasonal variations in the Meadowlands.

   a. Obtain measurements of NO_x and O_3 and meteorological variables from rooftop observations over a one-year period to distinguish seasonal patterns.

   b. Identify the critical characteristics of NO_x and O_3 and the influence of the primary meteorological parameters (temperature, wind speed, wind direction, relative humidity and atmospheric pressure) on their ambient concentrations.

   c. Investigate the fundamental mechanisms that drive nitric oxide (NO) when altered under variable atmospheric conditions to further explain its role both as a precursor to O_3 production and as a catalyst for O_3 consumption.

3. Evaluate the relationship between meteorological parameters and ambient levels of NO_x and O_3 for a better understanding of their combined effects on respiratory hospital admissions in the Meadowlands.

a. Examine the relationship between pollutants and hospital admissions through path analytic models, multiple regression analysis and bivariate correlations.
b. Determine whether hospitalization for respiratory illnesses is accelerated by short-term exposure to air pollutants.
c. Explain whether the seasonal and diurnal patterns of air pollutants can provide a guide for reducing risks to air pollution exposure.
d. Identify the diurnal patterns of air pollutants to explore the feasibility of establishing more than one averaging period to improve current air pollution standards from a public health point of view.

1.3. Major findings

In general, time series and regression analyses indicated that the two meteorological factors that accounted for the greatest variability in air pollution concentrations were temperature and wind speed. Elevated pollutant concentrations were characterized mainly by low wind speed although there were times when O_3 concentrations increased paradoxically by 0.6 units ($p<0.001$), with one unit increase in wind speed due to downwind accumulation from emission sources. Conversely, NO_x concentrations showed greater dependence on diurnal source emissions than on meteorological factors. The association between pollution concentrations and hospital admissions was, in many instances, both positive and statistically significant.

Meteorological influence on the formation and reaction rates of O_3 can potentially lead to greater health risk for exposure in summer, mid-days and on weekends. Higher respiratory hospital admissions mainly in winter and cool seasons rather than in the summer, were particularly related to exposure to elevated NO_2 concentrations. On the other hand, marked associations between O_3 concentrations and lagged hospital admissions were more apparent than a direct health effect based on the strong seasonal pattern of O_3.

The results of this study suggest that the underlying mechanisms of toxicity linked to air pollutants are more clearly understood when studied in the context of meteorological factors. It makes a contribution to current knowledge by reducing the complexities presented in previous air quality and epidemiological studies that obtain data from central monitoring stations that do not detect the synoptic effects of air pollutants. By linking air pollution and hospital admissions, this study has the advantage of explaining the unique qualities that characterize the relationship between atmospheric conditions and public health effects on a local scale. These findings may inform policy makers regarding the need for more practical regulations that can more effectively reduce the risk to air pollution exposure.

Chapter 2: Evaluation of air pollution, local meteorology and urban public health[1]

Abstract

The relationship between meteorological conditions and air pollution was assessed as a plausible explanation for respiratory health problems in Newark, New Jersey on the US east coast. Pollutants in both particle ($PM_{2.5}$ and PM_{10}) and gas phase (O_3) were collected and analyzed. We find that $PM_{2.5}$ concentrations decreased by 30.8% (95% CI: –43.2%, –15.8%) when the prevailing westerly wind speeds increased by 1 unit. A 0.01 ppm increase in O_3 was associated with a rise in daily respiratory hospital admissions by 2.16 (95% CI: 0.72, 3.59) on 1 lag day. A small decrease in the monthly mean $PM_{2.5}$ concentration was associated with a corresponding decrease in hospital admissions. These initial findings suggest that local meteorological conditions influence the concentrations of both particulate and gas phase pollutants which in turn, can affect respiratory health problems in this urban area. More comprehensive investigations will be followed based on this pilot study.

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[1] This chapter is in press as Roberts-Semple, D. and Gao, Y. (2012) 'Evaluation of air pollution, local meteorology and urban public health', *Int. J. Environmental Technology and Management*, Vol. X, No. Y, pp. 000-000.

2.1. Introduction

Air pollution is an environmental problem of global concern that has driven a considerable amount of field research worldwide, and the adverse health effects of exposure to ambient air pollutants are well-known (Frumkin, 2002; Bell, 2006; Brauer et al., 2008). Fuel combustion sources, primarily vehicular emissions (Alam et al., 2011; Vanderstraeten et al., 2011), produce large amounts of primary pollutants that can react in the atmosphere to form secondary pollutants, such as fine aerosol particles and ozone (O_3) (Bell, 2006; Peters et al., 2009). Ambient particulate matter (PM) comprises a heterogeneous mixture of solid and liquid fine particles with an aerodynamic diameter of ≤ 2.5 μm ($PM_{2.5}$) that can penetrate into the deepest parts of the human lungs (Pope et al., 2002; Madsen and Mottola, 2003). Coarse particles, with an aerodynamic diameter of 2.5 to 10 μm (PM_{10}) are generally deposited in the human upper respiratory system (Brook et al., 2004). Ground-level ozone, formed in intense heat and sunlight, can also be toxic to humans (Bell, 2006; Seinfeld and Pandis, 2006; Wei et al., 2007).

The primary objective of this study is to investigate whether ambient PM and O_3 levels are dependent on meteorological conditions, to explain an association with daily hospital admissions in Newark. The key pollutants discussed, O_3 and $PM_{2.5}$, are two of the six criteria pollutants assigned by the National Ambient Air Quality Standard (NAAQS) as a requirement of the EPA under the Clean Air Act (Madsen and Mottola, 2003; Dominici et al., 2006). Despite regulations, observed levels have repeatedly exceeded the federal standards that protect human health (Madsen and Mottola, 2003; Rosenzweig et al., 2005; Engler et al., 2011). Long-term exposure to O_3 can aggravate asthma, chronic bronchitis, reduce lung capacity, and cause permanent lung damage (Pope et al., 2002; Stieb et al., 2002; Tager et al., 2005; Bell, 2006). Short-term respiratory problems include pneumonia, irritation of lung airways, inflammation and breathing difficulties during deep inhalation and exercise (Seinfeld and Pandis, 2006; Bell, 2006; Wei et al., 2007). Even at levels below the Environmental Protection Agency (EPA) standard, exposure to O_3 can cause adverse health effects necessitating hospitalization and medical treatment (Weisel et al., 1995, 2002; Madsen and Mottola, 2003; Tager et al., 2005; Bell, 2006). Ren et al. (2010) found a 1.96% (95% CI: –1.83%, 5.90%) increase in respiratory disorders, with a 0.01 ppm increase in the four-day moving average of maximal eight-hour O_3.

In New Jersey, hourly air quality readings in 2003, 2004 and 2005 with 28, 19 and 29 days of exceeded O_3 levels, respectively, motivated a proposal to implement the California Low Emission Vehicle program in 2009 (NJDEP, 2006). In central and northern New Jersey, average O_3 levels greater than 0.06 ppm, resulted in emergency room visits 28% more frequently, demonstrating that O_3 can potentially have serious health effects at levels far below the EPA standard of 0.08 ppm during an eight-hour period (Weisel et al., 1995; Madsen and Mottola, 2003; USEPA, 2006). Although O_3 levels observed in the present study did not exceed this standard, based on previous findings, they may still pose a threat to public health. This investigation demonstrates that current standards do not necessarily guarantee protection nor provide a threshold below which health effects of air pollutants are not expected to occur.

High PM levels have been linked to a range of health problems (Jameson et al., 1997; Holman, 1999; Pope et al., 2002; Gurjar et al., 2008) including decreased lung function, bronchitis and emphysema (Frumkin, 2002; Pope et al., 2002; Wilson et al., 2005). In the bloodstream, $PM_{2.5}$ can cause cardio-respiratory diseases (Frumkin, 2002; Stieb et al., 2002; Bell, 2006); and among other effects, the relative risk is highest for respiratory hospital admissions and mortality (Bell et al., 2009; Brunekreef et al., 2009). According to Peng et al. (2009), in multiple-pollutant models, an interquartile range increase in $PM_{2.5}$ components was associated with a 1.01% (95% PI, 0.04% to 1.98%) increased risk of respiratory admissions on the same day. At least 50% of the PM_{10} mass in most urban centers in the US comprise $PM_{2.5}$ and estimates of the effects on hospital admissions largely reflect the $PM_{2.5}$ component with consistency for respiratory diseases (Dominici et al., 2006). While few studies have assessed the association between $PM_{2.5}$ and hospitalization (Dominici et al., 2006), even fewer studies have been conducted on the daily count of hospital admissions, since these data are not always readily available (Stieb et al., 2009).

Due to the potential effects on human health, there is a need to reduce ambient air pollution levels. To address this need, the EPA has implemented an alert system with published PM and O_3 levels, including next-day forecasts and real time air quality information for more than 150 cities in the USA (Brook et al., 2004; White, 2010). National and regional studies provide useful data from central monitoring stations but often present limitations of the real effects of air pollutants. This is partly because data collection on such a large scale is cumbersome, while spatial variability due to local meteorology and location is often overlooked (Forastiere et al., 2005; Varadarajan, 2007). In addition, some studies show that individual exposure to toxic agents of air pollution may vary within a single city as much as across different cities (Brook et al., 2004).

By linking air pollutants and meteorological data with hospital admission records from a common geographic location in Newark, this study provides an analysis of the spatial-temporal effects and meteorological conditions on which ambient air pollution levels strongly depend. Although extensive air pollution research has been conducted in New Jersey, including the characterization of aerosol particles and precipitation at the study site (Zhao and Gao, 2008a; Song and Gao, 2009), it remains unclear how toxic agents such as ambient PM and O_3, are influenced by meteorological and concomitant respiratory infectious agents. The lack of new evidence should not be treated as a lack of effect since this often impairs the identification of the most relevant health effect indicators (WHO, 2003). Knowledge gained from this study can serve as a useful reference to improve regulations and controls that lead to the mitigation of inhalable gaseous and particle pollutants in the ambient air.

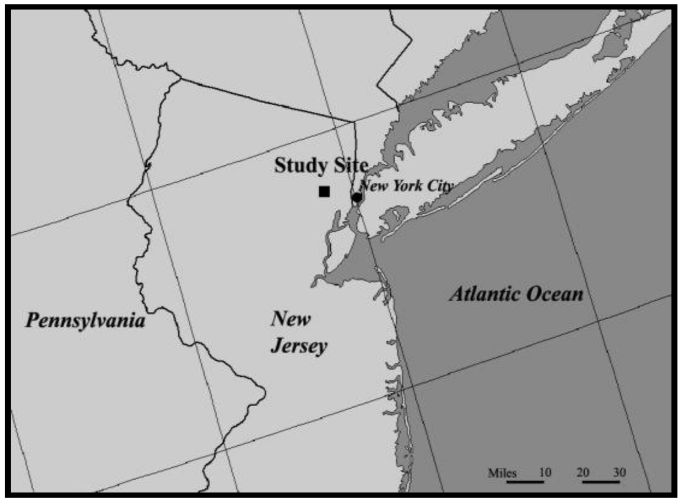
2.2. Materials and Methods

2.2.1 Study Site

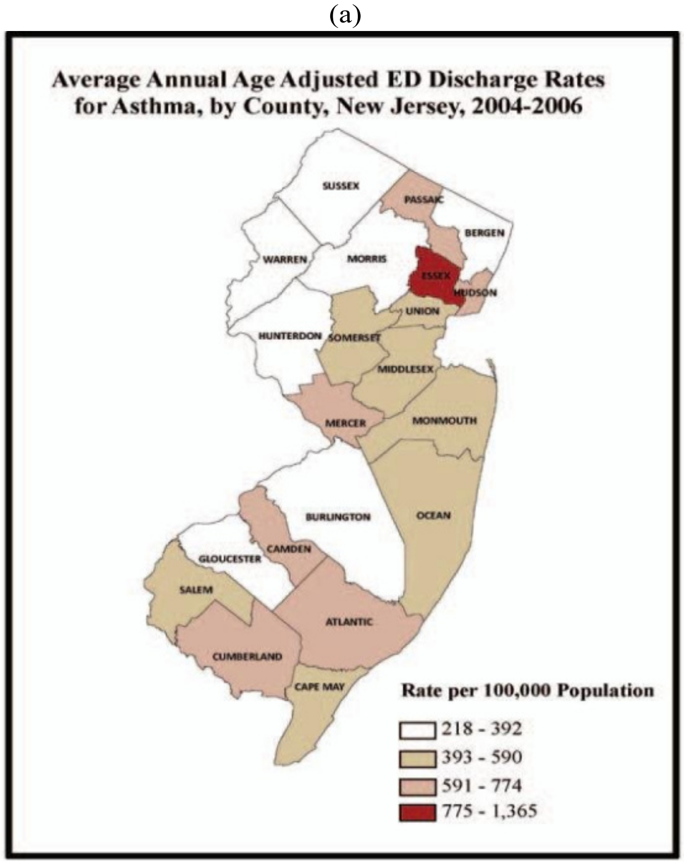
Aerosol sampling was conducted in Newark, New Jersey located at 40° 44′ N and 74°10′ W on the US Northeast coast [Figure 2.1(a)]. Newark lies at 83.3 m above sea level in Essex County, with humid summers and high temperature often exceeding 32°C. It is the largest metropolitan centre in New Jersey, where serious respiratory health effects have been observed [Madsen and Mottola, 2003; Figure 2.1(b)]. Surrounded by many highways and industrial activities, Newark is a heavily polluted environment. Densely populated urban areas and residential suburbs border the city centre to the south and west, while industrial areas are located in the north. The Newark Bay and the Passaic River form the eastern boundary.

2.2.2. PM Sampling and Meteorological Data Collection

Aerosol samples were collected using a Partisol Model 2000-D Dichotomous Air Sampler (ThermoElectron Corporation, East Greenbush, New York), mounted on the roof of a building at ~20 m above the ground on the Rutgers University campus. A total of 184 samples of fine ($PM_{2.5}$) and coarse PM ($PM_{2.5-10}$) were collected on Zefluor™ membrane filters (Pall Corp., 47 mm in diameter, 1 μm pore size) at 24-hour intervals daily for three consecutive months from August to October, 2009. Filters were prepared in a 100-class clean bench.



(a)



(b)

Figure 2.1 Map showing (a) situation of the study site in Newark, New Jersey (b) hospital discharge rates for asthma cases in New Jersey.
NJDHSS (2008)

The PM mass concentrations were obtained by pre- and post-weighing each filter under controlled temperature and humidity conditions in a laboratory at the Environmental and Occupational Health Sciences Institute (EOSHI) at Rutgers University in New Brunswick, using a microbalance (Model MT5 Mettler Toledo MT/UMT). The filters were stored in a refrigerator in pre-washed petri dishes both before and after weighing. These filters were placed in the weighing room at least 24 hours prior to weighing to be equilibrated at 20 ± 1°C and 35 ± 5% temperature and relative humidity, respectively. Meteorological parameters, mainly temperature, wind speed, wind direction and relative humidity, were obtained simultaneously from rooftop observations, with the Davis Vantage Pro2 Unit (Davis Instruments, Hayward, California), also mounted on the PM sampling platform. For continuous weather data, sampled data were supplemented with data from the Newark Museum meteorological station nearby.

2.2.3 Collection of O3 Data and Hospital Admission Record

To better evaluate the overall impacts of PM, O3 and meteorological factors on public health, the daily maximum 24-hour averages of O3 concentrations during the period of observation were obtained from the Newark Firehouse. It is a monitoring station in the New Jersey Department of Environmental Protection (NJDEP) network and a part of the National Photochemical Assessment Monitoring Station (PAMS) program. Hourly measurements of O3 precursors – 54 volatile organic compounds (VOCs) that affect O3 formation – were followed by daily retrieval and extensive review for subsequent validation (NJDEP, 2007). Hospital admissions were used as potential respiratory health indicators for O_3 and PM. Health records of daily total respiratory hospital admissions were obtained from Saint Michael's Medical Center, a nearby hospital that reports data the New Jersey Department of Health and Senior Services (NJDHSS).

The daily total hospital admissions in the study are based on International Classification of Diseases, Ninth Revision, clinical modification (ICD-9-cm) codes consistent with an application submitted to the Institutional Review Board (IRB), for use in a subsequent one-year dataset as a continuation of this pilot study. The data contained no individual identifiers, thus no patient consent was necessary. ICD-9 codes included acute upper respiratory infections of multiple unspecified sites, acute bronchitis [465] and bronchiolitis [466], chronic sinusitis [473], other diseases of upper respiratory tract [478], bronchitis (not specified as acute/chronic) [490], chronic bronchitis [491], emphysema [492], asthma [493], bronchiectasis [494], extrinsic allergic alveolitis [495], chronic airways obstruction, not specified elsewhere [496], respiratory conditions due to other and unspecified external agents [508], other respiratory diseases of the lung [518] and other diseases of respiratory system [519].

2.3. Results and Discussion

2.3.1 Trend Analysis

Figure 2.2 shows a continuous decline in temperature from August through

October with a corresponding decline in O_3, suggesting that the production of O_3 is temperature -dependent, consistent with previous studies (Weisel et al., 2002; Forastiere et al., 2005; Bell, 2006). Estimates of the daily measurements of O_3 in the multiple regression analysis indicated a reasonable performance with an R^2 value of 0.55, further explained by Forastiere et al. (2005).

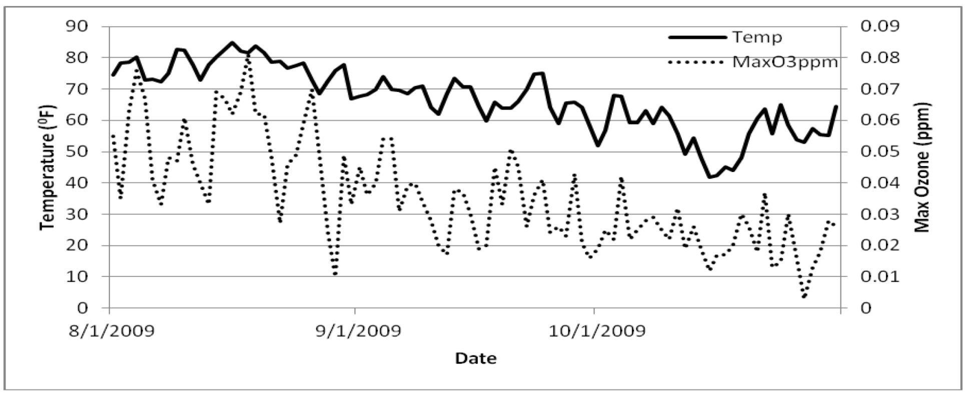


Figure 2.2 Time series of average daily temperature and maximum ozone from August to October 2009.

As shown in Figure 2.3, the highest concentrations of both $PM_{2.5}$ and PM_{10} occurred early September, although not the warmest month in the study period. It is noteworthy, that relative humidity did not seem to influence PM concentrations probably due to counteracting effects of nucleation, condensation and other related processes (Hussein et al., 2006; Varadarajan, 2007). These are determined by measuring the particle number concentrations (Hussein et al., 2006) which was not done in this study.

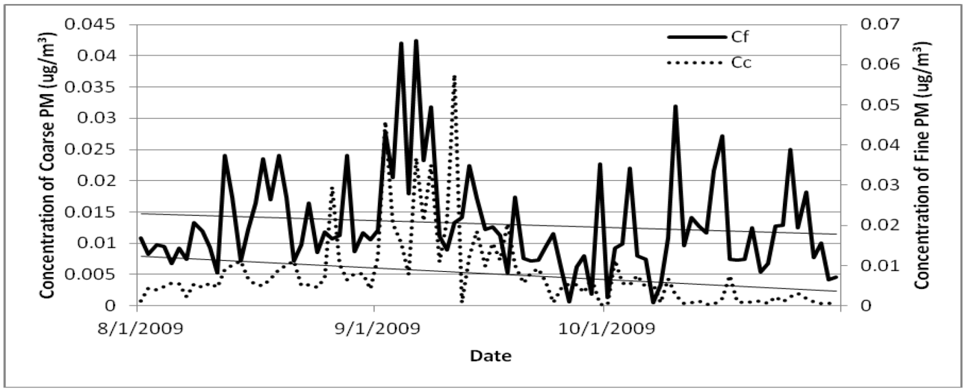


Figure 2.3 Concentrations of $PM_{2.5}$ (Cf) and PM_{10} (Cc).

Temperature was presumably a primary factor contributing to the elevated PM concentration levels. This observation is further illustrated in Figure 2.4. As indicated by the oval rings, low temperature and high concentrations of $PM_{2.5}$ occurred in early September and October. This trend was also observed by Hussein et al. (2006), over a three-year study period and may also be related to higher particle number concentrations. Particle number concentrations increased with low ambient temperatures, attributed to stable atmospheric stratification and controlled by atmospheric pressure (Hussein et al., 2006). Similar results were found in Erfurt, Germany (Ebelt et al., 2001) where higher $PM_{2.5}$ concentrations occurred in colder months than during the summer months. This suggests that stable stratification, related to atmospheric pressure, may influence ambient PM concentrations indirectly (Ebelt et al., 2001; Hussein et al., 2006 Engler et al., 2011); however, a stronger influence is observed when combined with temperature inversions.

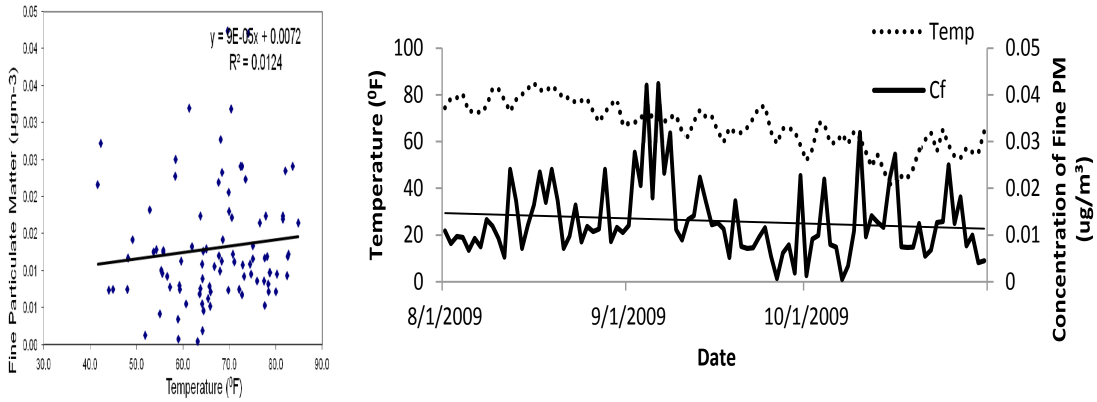


Figure 2.4 Relationship between concentration of $PM_{2.5}$ (Cf) and temperature (a) Slope, $R^2 = 0.0124$ (b) Time series.

In addition, the higher PM concentrations during colder periods may be explained by higher vehicular emissions associated with an increase in commute to avoid exposure to colder temperatures, and less complete combustion under colder climatic conditions (Ebelt et al., 2001). Although statistical analysis was not done to confirm this relationship, similar studies that conducted both visual and statistical analyses to assess this relationship over time, have linked higher PM concentrations with increased traffic and energy production during colder months (Engler et al., 2011). On the other hand, the trend line in Figure 4 indicates that there was a gradual overall decline in $PM_{2.5}$ levels, with a corresponding decline in temperature during the study period, from August to October. This finding is consistent with several air pollution studies that examine the influence of temperature on fine PM concentration (Lee et al., 2002; Rosenzweig et al., 2005; Lai and Cheng, 2009).

Higher temperatures are also enhanced by tall buildings which retain heat over longer periods (Rosenzweig et al., 2005). This confirms that temperature is one of the major factors controlling the concentrations of fine PM in urban areas, and further suggests that the higher levels of $PM_{2.5}$ that occurred with warmer temperature, were partly due to temperature inversions, a feature that is common under stagnant conditions in urban areas (Rosenzweig et al., 2005; Hussein et al., 2006; Engler et al., 2011). High day-time surface temperatures generate convective winds within the urban boundary layer. At night, however, a reversal of this daytime atmospheric mixing is evident due to the absence of solar heating; atmospheric convection decreases with increasing stabilization in the urban boundary layer and the development of an inversion layer. The formation of a shallow radiation inversion in Newark, was due to the development of a stable boundary layer attributed to limited cooling rates in summer 2008 (Thuman CC, unpublished master's thesis).

Table 2.1 presents a summary of the primary variables which were all normally distributed. Appropriate ranges were selected for each variable to match the number of observations in each month. The most frequent temperatures 16.6°C to 27.7°C (62°F to 82°F), occurred in August and September when O_3 was greater than 0.04 ppm. There was little variation in wind speed in the selected categories (< 4 mph and > 4 mph) but the prevailing south westerly winds in sector 271° to 360° occurred most frequently.

Table 2.1 Descriptive data of primary air pollutants and meteorological variables

Variable	Range	Number of Observations per month		
		August	September	October
Temperature (^{0}F)	41-61	0	2	23
	62-82	26	27	8
	83-103	6	0	0
Wind Speed (mph)	<4	12	15	11
	>4	19	15	20
Wind Direction	0-90^0	2	4	5
	91-180^0	5	6	4
	181-270^0	13	2	8
	271-360^0	11	17	15
$PM_{2.5}$ (μg/m^3)	<0.006	1	5	7
	>0.006	30	25	24
O_3 (ppm)	<0.04	8	7	1
	>0.04	23	23	30

Note: Total number of observations = 92

The monthly arithmetic mean, median and standard deviation of the different variables are presented in Table 2.2, indicating an increase in the concentrations of both $PM_{2.5}$ and PM_{10} from August to September followed by a decline in October. A similar trend was observed for hospital admissions; however, an inverse relationship existed between PM and wind speed. Conversely, consistent with temperature, O_3 showed a steady decline while wind speed and relative humidity both decreased from August to September and increased in October. Based on these initial observations, the hypothesis to evaluate the relationship between meteorological conditions and air pollutants was tested.

Table 2.2 Monthly concentrations of ambient air pollutants and meteorological variables

Variable	Monthly mean (median)			Standard Deviation		
	August	September	October	August	September	October
O_3 (ppm)	0.051 (0.049)	0.034 (0.035)	0.02 2(0.022)	0.017	0.011	0.008
$PM_{2.5}$ ($\mu g/m^3$)	0.013 (0.011)	0.015 (0.012)	0.011 (0.011)	0.005	0.010	0.007
PM_{10} ($\mu g/m^3$)	0.007(0.006)	0.015 (0.011)	0.003 (0.001)	0.005	0.014	0.003
Admissions	68.26 (69)	70.23 (69)	65.68 (67)	12.26	10.15	10.53
Temp (^{0}F)	77.4 (77.7)	67.33 (68)	56.06 (55.8)	10.24	4.47	7.22
Wind Spd (mph)	3.91 (3.8)	3.89(3.6)	4.62 (4.2)	1.63	1.63	2.23
R.H. (%)	60.07 (57)	57.87 (55)	59.45 (56)	18.84	19.50	23.45

In general, there was a positive association between local meteorological factors and air pollutants, with the exception of wind speed (Table 2.3). The investigation determined the statistical significance of the apparent relationships demonstrated and whether ambient air pollution levels were sufficiently influenced by local meteorology to result in increased hospitalizations. The mean hospital admission could be explained by a lag time in O3 concentrations which showed a consistent decline from August to October with means of 0.05, 0.03 and 0.02, respectively.

2.3.2 Statistical Analysis

Statistical analyses were conducted primarily with the SPSS (Version 11.0) software package for the estimated Pearson correlation coefficient (r), a time series regression analysis in which the p-value was in agreement with Pearson correlation.

Table 2.3 Correlation matrix of pollutants, wind speed, temperature and admissions.

Pollutant	Wind Speed	Temperature	Hospital Admissions
$PM_{2.5}$	-0.128	0.111	0.011
PM_{10}	-0.146	0.190	0.215*
O_3	-0.134	0.74	0.195

*Correlations are significant at 0.05 level (2-tailed)

There is no direct measure for attributing changes in hospital admission to changes in meteorological variables. Therefore, the association between meteorological variables and ambient air pollutants was first determined and then linked to hospital admissions using stepwise multiple regression, to determine whether the statistical significance varied from day-to-day. In this method, the important coefficients were automatically selected and statistically significant variable was inserted into the model in turn. Once the significance level was determined for each variable, any variable with a statistical significance of more than 0.05 was removed from the model. This procedure was repeated until all variables with possible relationships were added to the model.

The dependent variable was the daily total number of hospital admissions for all respiratory illnesses and the independent variables: PM and O_3 were linked to demonstrate possible relationships. Lag analyses were automated by SPSS and SAS to reduce the potential confounding effect of meteorology, similar to Dominici et al. (2006). A lag of 0 days, corresponding to the association between concentrations of $PM_{2.5}$, PM_{10} and O_3 on a given day, and the risk for hospital admission on the same day was first applied, followed by tests of up to four lag days. In other words, we sought to find out whether there was a lag effect of $PM_{2.5}$, PM_{10} and O_3 on hospital admission. This means that exposure to an air pollutant on a particular day, may result in hospital admission on that same day or up to four days later, denoted as lags 0, 1, 2, 3 and 4, respectively. Dominici et al. (2006) found positive associations between day-to-day variation in $PM_{2.5}$ concentration and hospital admissions for all respiratory outcomes, with the largest effect occurring at lags 0 and 1 for chronic obstructive pulmonary disease (COPD) and lag 2 for respiratory tract infections.

Table 2.4: Correlation between pollutants and respiratory hospital admissions

Pollutant	Respiratory Hospital Admissions				
	Lag 0	Lag 1	Lag 2	Lag 3	Lag 4
$PM_{2.5}$	0.01	0.02	0.18	-0.03	0.04
PM_{10}	0.22	0.17	0.01	-0.01	0.04
O_3	0.21	-0.02	-0.11	-0.01	-0.03

2.3.3 Air Pollutant Concentrations and Meteorological Parameters

Studies involving a range of meteorological parameters have demonstrated that wind speed is one of the most important meteorological parameters influencing ambient PM concentration levels (Hussein et al., 2006; Hu et al., 2009). The daily means of ambient air pollutant concentrations and meteorological data show meaningful relationships in both multiple regression analysis (p-value) and Pearson correlation coefficient (Table 2.4). The value was identical in both tests. The lowest PM concentrations were associated with high wind speed (Figure 2.5). $PM_{2.5}$ was negatively correlated with wind speed as suggested by the Pearson correlation of r = –0.13 which agreed with the multiple regression analysis (R^2 = 0.02). This relationship was not statistically significant.

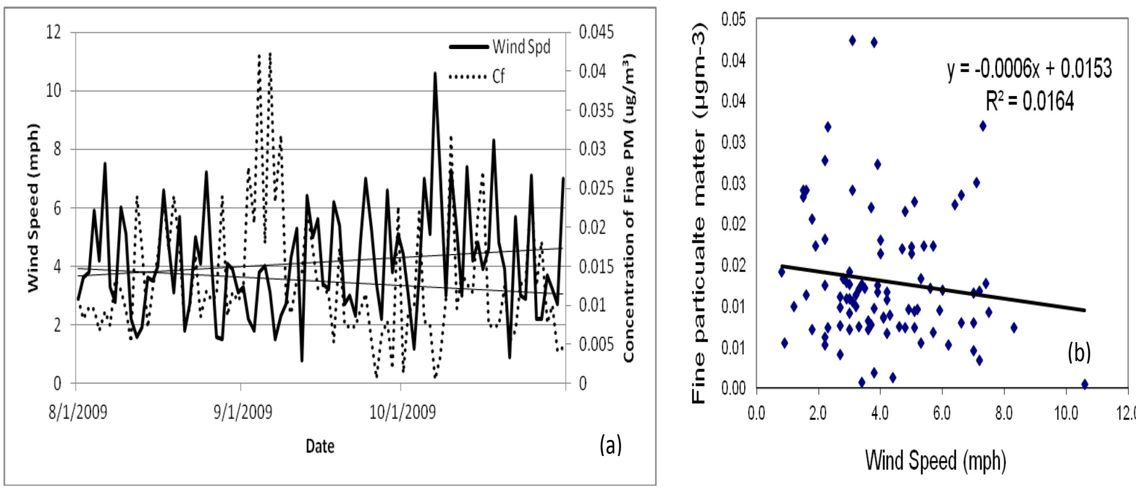


Figure 2.5 Relationship between concentration of $PM_{2.5}$ (Cf) and wind speed (a) Time series (b) Slope

These results are consistent with recent studies in Southeast Kansas, Helsinki, Finland and Santa Monica, Southern California (Guerra et al., 2006; Hussein et al., 2006; Hu et al., 2009). Generally, high wind speeds result in the removal of PM_{10} from the atmosphere. Hussein et al. (2006) showed that there was a significant dependence of PM concentrations on wind speed: elevated concentrations coincided with low wind speeds and vice versa. Hu et al. (2009) also found a steady pattern of increased pollutant dispersion with higher wind speeds, consequently reducing the concentrations of ultra-fine particles during afternoon hours. In their study, Guerra et al. (2006) found that the second highest PM concentrations were recorded when winds were calm. There is also a significant association between fine PM and wind direction which may indicate the source regions of transported particles (Hussein et al., 2006; Hu et al., 2009). Accordingly, higher concentrations of fine PM occurred with calm conditions and under the influence of prevailing south westerly winds in Newark (Figure 2.6).

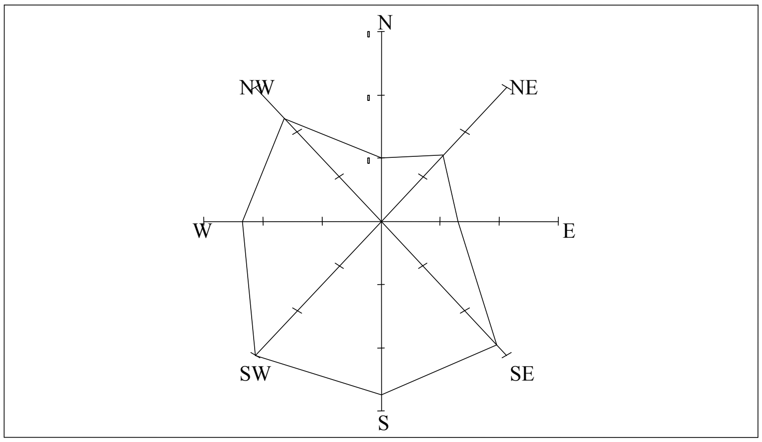


Figure 2.6 Wind rose highlighting prevailing south westerly winds in Newark

Previous studies have found that higher temperatures are associated with increased air pollutant concentrations (Lee et al., 2002; Rosenzweig et al., 2005; Lai and Cheng, 2009). In this study, there was a strong positive correlation between temperature and O_3, r = 0.74. The correlation between $PM_{2.5}$ and PM_{10} with temperature was also positive but not statistically significant; there was low co-linearity among these variables: $PM_{2.5}$, r = 0.11 and PM_{10}, r = 0.19. The log of both $PM_{2.5}$ and PM_{10}, however, were statistically significant, r= 0.22 (p=0.04) and r=0.5 (p=0.0001), respectively. Similarly, Stafoggia et al. (2008) found that high temperature levels during the summer were associated with higher PM levels than in the winter. Large variations in temperature have also been shown to correspond with variations in air pollutant concentrations in several studies (Vukovich and Sherwell, 2002; Hussein et al., 2006); however, meteorological variables, notably atmospheric pressure and temperature inversions, can also influence this trend (Hussein et al., 2006; Hu et al., 2009).

2.3.4 Hospital Admission and Pollutant Concentrations

Daily hospital admissions for respiratory symptoms attributed to PM and O_3 exposure were linked to these pollutant concentration levels for an associated health effect. Hospital admission was not very well explained by pollutant concentrations; however, as indicated by the trend lines in Figure 2.7, in general, hospital admissions decreased as $PM_{2.5}$ concentrations decreased. The analysis showed a positive correlation but poor co-linearity among ambient air pollutants and hospital admission: $PM_{2.5}$ and PM_{10}, $r = 0.01$ and $r = 0.22$, respectively, and for O_3, $r = 0.21$. These values were identical to R^2 in the regression analysis but were not statistically significant, except for the log of PM_{10}, $r=0.23$ ($p=0.03$). The corresponding decline in hospital admission and fine PM concentration levels indicated by the trend lines complement evidence found in other studies based on hospitalization for respiratory illnesses due to $PM_{2.5}$ exposure. In their investigation of the effect modification of short-term effects of $PM_{2.5}$, temperature and O_3, Dominici et al. (2006) found that both county and regional average temperature positively modified the association between $PM_{2.5}$ and hospital admission rates for two respiratory outcomes. An estimated 18 and nine additional hospital admissions in 10,000 individuals per 10-

μg/m^3 increase in $PM_{2.5}$ for COPD and respiratory tract infections, respectively were found in the warmer of two regions that differed by 1°C (Dominici et al., 2006). Brook et al. (2004) found that there were increases in hospital admission of 0.7% and 0.8% with a corresponding increase in PM_{10} by as much as 10-μg/m^3. Stafoggia et al. (2008) also found that higher PM levels in summer resulted in a 2.54% increase in risk of death (95% CI: 1.31, 3.78) compared with 0.20% (95% CI: –0.08, 0.49) in winter.

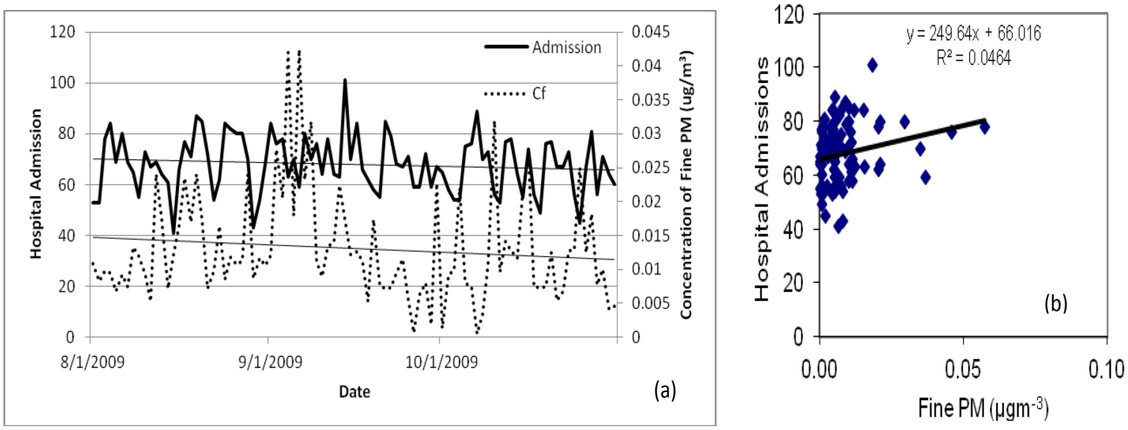


Figure 2.7 Relationship between $PM_{2.5}$ (Cf) and hospital admissions (a) Time series (b) Slope

The relationship between hospital admission and O_3 are presented in Figure 2.8. Despite the overall low co-linearity between hospital admission and air pollutants over the three-month period, a more significant relationship is evident in August for O_3, for which relatively high concentrations were observed. This is followed by a gradual decline associated with temperature variations in September and October. The trend line indicates more distinctly, however, that with a decline in maximum O_3 concentration levels, there was a corresponding decline in daily total hospital admissions. There were more frequent hospital admissions when the mean O3 levels were > 0.04 ppm than when they were < 0.04 ppm. This finding contributes to current knowledge of the potential health risks of O3, and the meteorological conditions that may enhance or aggravate respiratory health conditions.

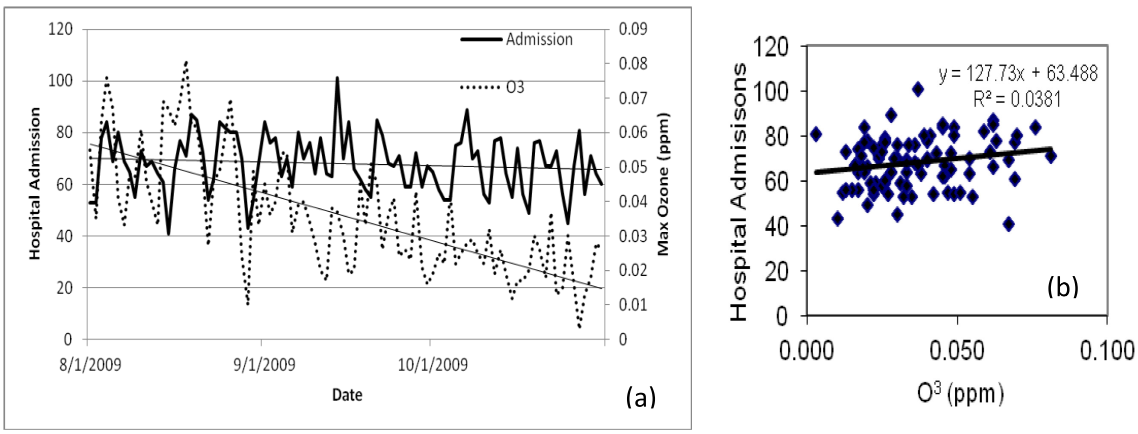


Figure 2.8 Relationship between ozone and hospital admissions (a) Time series (b) Slope

To demonstrate the adverse short-term health effects in relation to day-to-day variations in air pollution concentrations, some studies have employed lag times with varying levels of significance due to the susceptibility of populations observed, meteorology, commute and exposure times, etcetera (Brook et al., 2004). The effect between exposure and response can be spread over several days. Therefore, specifying the lag association would allow better comparison and provide greater insight into the relationship between concentration of a specific air pollutant and hospital admission (Schwartz, 2000; Stieb, 2009).

The automated lag association used in this study was done with the consideration that the number of admissions on a single day may be influenced by exposure to air pollution both on that day and on previous days. The statistical significance between hospital admissions and air pollutants, however, did not improve with an increase of up to four lag days for $PM_{2.5}$, r = 0.02, 0.18, –0.03 and 0.03 and for O_3, r = –0.02, –0.11, –0.01 and –0.03 (Table 2.4). The only exception was that a rise in O_3 was associated with an increase in daily respiratory hospital admissions at one lag day, consistent with similar short-term studies. The strongest lag association automated by the multiple stepwise regression model, showed that one unit increase in O_3, was associated with an increase in hospital admissions on one lag day (R^2 = 0.091). In other words, with an increase in O_3 by 0.01 ppm on 1-lag day, there was an associated increase in daily respiratory hospital admissions by 2.159 (95% CI: 0.721, 3.596).

Chan et al. (2009) also found that the effect of four air pollutants, including O_3 and PM_{10}, on emergency room visits of up to two lag days was not statistically significant. One of the largest studies in the USA, NMMAPS, reported no difference in health effects of particle pollutants when 0 to 2 lags were employed, and therefore they based their estimates only on one lag day instead (Brook et al., 2004). Yet another study in Canada of O_3, PM and other pollutants; O_3 was most consistently associated with respiratory hospital emergency department visits over a period of two lag days particularly in the warm season (Stieb et al., 2009). These findings suggest that the effect of O_3 and PM did not worsen with time after initial exposure to the air pollutants. It eliminates the assumption of a linear relationship or that the respiratory health conditions in response to O_3 and PM is transient.

Weisel et al. (1995) identified a consistent pattern in the magnitude of reported effects, time lag between exposure and response for four health indices, including hospital admissions for respiratory response to O_3 exposure in Central New Jersey. To reduce error in the association between asthma and emergency visits, they made a comparison with the mean O_3 of lags 1 and 2, respectively, using stepwise multiple regression analysis. Their findings indicated no significant relationship between O_3 exposure and emergency department visits when the time lags were employed during a five year period. Peng et al. (2009) found an increase in risk of respiratory admissions on the same day: lag 0 (95% PI, 0.04% to 1.98%) with exposure to $PM_{2.5}$. In addition, a study conducted in Rome showed that the effect of air pollution was strongest on the event day: lag 0 (Forastiere et al., 2005).

Despite slight variations, these studies have demonstrated that even in the short term and at significantly low concentrations, both ambient particulate and gas-phase pollutants are capable of affecting human health (Weisel et al., 2002; Brook et al., 2004). Epidemiological studies demonstrate statistical association between air pollutants and hospital admissions for respiratory problems but they do not link the effect to meteorological influence. In a specific high-risk population such as Newark where such data are lacking, and where an investigation of this nature is unprecedented, the results from this study are valuable.

2.4. Conclusions

The data presented in this pilot study demonstrated that ambient PM and O_3 levels are dependent on meteorological conditions, particularly temperature and wind speed. The combined effects of weather parameters and their interaction with ambient air pollutants indicate an association with daily hospital admissions. There was a positive association among O_3, PM and hospital admission suggesting that air pollutant concentration levels, even below current EPA standards, could be associated with an increased risk for respiratory illnesses in New Jersey. The study contributes to a fundamental understanding of the key characteristics and controls of these variables in Newark. Admittedly, there were other variables that could potentially influence these results, which render our understanding of the relationship between ambient air pollutants and public health incomplete.

Unlike other studies which utilize modeling techniques to predict the association between air pollution and health effects, however, the present study linked real hospital admissions and sampled air pollution data in a local geographic location. The in-situ sampling method used is useful in environmental health research and practice. It can stimulate interest among other institutions to carry out their own analyzes to aid in the reduction of respiratory illnesses in similar urban areas. Further, it provides a rationale for continued investigation of the characteristics of particle and gas-phase pollutants in the ambient air. Future longer-term studies at this site would determine the chemical composition and toxicity of air pollutants to improve understanding of the interaction between meteorology and respiratory toxic agents. This will strengthen current evidence that warrants the reduction of PM and O_3, and hence, respiratory hospital admissions in Newark.

Chapter 3: Seasonal characteristics of ambient nitrogen oxides and ground-level ozone in metropolitan northeastern New Jersey[1]

Abstract

Nitrogen oxides (NO_x) and ground-level ozone (O_3) were measured and meteorological parameters were monitored to determine the seasonal variations of gas-phase pollutants in the Meadowlands. O_3 and NO_x were inversely related; the highest average NO_x concentration (29ppb) occurred in winter while average O_3 concentrations peaked in summer up to 36.2ppb. The seasonal variations of O_3 were more distinct than NO_x. In multiple linear and principal component regression analysis, ambient levels of NO_2 and O_3 were influenced primarily by wind speed. In time series and regression analysis, NO_2 and O_3 displayed an inverse relationship with wind speed but O_3 paradoxically increased with wind speed downwind.

The seasonality of O_3 was amplified mainly by wind speed and temperature, while NO_x displayed stronger dependence on diurnal source emissions. Concentrations of NO_x and O_3 were also influenced by differences in chemical processing through their complex emission-production and consumption mechanisms, making them interdependent. In intense solar radiation, O_3 was NO_x-dependent but as O_3 levels were inhibited by lower temperatures, NO_x concentrations increased. Higher O_3 on weekends indicated an

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[1] This chapter is published as Roberts-Semple, D., Fei Song, Gao, Y., 2012. Seasonal characteristics of ambient nitrogen oxides and ground-level ozone in metropolitan northeastern New Jersey, Atmospheric Pollution Research 3, 247–257. DOI: 10.5094/APR. 2012.027

apparent sensitivity to VOC precursors. This study provides a basis for improved air quality standards primarily in summer and during daily O_3 peaks. Additionally, plans for protection against health problems caused by O_3 and NO_x are feasible in this region.

3.1. Introduction

Nitrogen oxides (NO_x) and ozone (O_3) are two of the six criteria pollutants assigned by the National Ambient Air Quality Standard (NAAQS), regulated by the United States Environmental Protection Agency (USEPA). NO_x comprise a mixture mainly of nitric oxide (NO) and nitrogen dioxide (NO_2), prominent in air quality studies (Brook et al., 2004; WHO, 2003). Although NO is a simple molecule, its production secondarily generates strong oxidizing agents and reactive nitrogen species notably NO_2, that may modulate the development of chronic inflammatory airway diseases (Ricciardolo et al., 2004). Nitrogen dioxide has been generally kept at low levels through alert systems with published levels, next-day forecasts, real time air quality information and the Low Emission Vehicle program (Brook et al., 2004; NJDEP 2006; USEPA, 2006). As a precursor pollutant of O_3, however, NO_2 poses a threat to public health (Ito et al., 2007; López-Villarrubia et al., 2010; Linares et al., 2010; Berti et al., 2009; Kelly et al., 2011; Namdeo et al., 2011).

Produced from both natural and anthropogenic sources, NO_x is formed in the atmosphere by the action of lightning on molecular oxygen and other chemical reactions involving naturally occurring nitrogen and volatile organic compounds (VOCs) (Sillman, 1999;

Godish, 2004; Brown et al., 2006). Previous studies have shown, however, that NO_x emissions from anthropogenic sources exceed natural sources (Godish, 2004). Increased combustion of fossil fuels and exhaust fumes may be extremely pervasive, causing serious environmental degradation, illnesses and deaths (Farrell et al., 1999; Ricciardolo et al., 2004; Godish, 2004; Gurjar et al., 2008). In the eastern United States, vehicular traffic contributed more than half of the total NO_x emissions with an increasing trend since the 1990s (Butler et al., 2005; Parrish, 2006). High temperature causes the oxidation of atmospheric N_2, first to NO and then to NO_2 (Sadanaga et al., 2008; Geddes et al., 2009), which plays a major role in the formation of ground-level O_3 (Farrell, 1999; Godish, 2004).

$$NO_2 + h\nu \longrightarrow NO + O \qquad R1$$
$$O + O_2 + M \longrightarrow O_3 + M \qquad R2$$
$$O_3 + NO + h\nu \longrightarrow NO_2 + O_2 \qquad R3$$
$$NO + RO_2 \longrightarrow NO_2 + RO \qquad R4$$

Although O_3 originates mainly from the upper stratosphere via convection movements, tropospheric O_3 production is largely dependent on the concentration of NO_x (Clapp and Jenkin 2001) in photochemical reactions, as shown above (R1-R2). The complexity of these series of reactions results both in the production and destruction of O_3 (R3). The photolysis of NO_2 to NO and oxygen atoms, is the first in the reaction series. By oxidizing NO to NO_2, a reactive oxygen-containing molecule (RO_2) such as a VOC (R4), helps to promote O_3 production in the presence of bright sunlight or high energy radiation (Seinfeld and Pandis, 2006; Brown et al., 2006; Song et al., 2011). While many VOCs are short-lived or exist in trace amounts well below the detection limit of current sampling methods (Godish, 2004), NO_x and O_3 rapidly interconvert within minutes to hours via photochemical reactions (Seinfeld and Pandis, 2006; Wei et al., 2007; Sadanaga et al., 2008; Geddes et al., 2009), posing major risks to public health (Blanchard and Tanenbaum, 2003; Torres-Jardon and Keener, 2006; Bigi and Harrison, 2010).

Over the past decade, there has been a steady decline in NO_x emissions but despite regulations, the national health-based O_3 standard has been repeatedly exceeded in many parts of the country (Sillman, 1999; Godish, 2004; Sadanaga et al., 2008; USEPA, 2006), including New Jersey (Madsen and Mottola, 2003; NJDEP, 2006). It is unclear how NO_x and O_3 interact with atmospheric conditions in northeastern New Jersey. The main objective of this study, therefore, is to assess the meteorological conditions on which ambient concentrations of NO_x and O_3 may strongly depend in this region. It attempts to characterize the seasonal patterns of NO_x and O_3 when concentrations are enhanced by local meteorology, and highlights the fundamental mechanisms that drive NO when altered under variable atmospheric conditions. These aspects of air quality research have not been fully studied particularly in the Meadowlands. An understanding of such mechanisms may lead to better regulation and provide a novel target in the reduction of air pollutants that cause respiratory and inflammatory diseases of the airways.

3.2. Materials and Methods

3.2.1. Study Region

The Meadowlands District is located in northeastern New Jersey (41ºN, 74ºW) approximately four miles west of New York City. This heavily industrialized and densely populated region comprises residential developments and occupies 83 km^2 mostly at sea level (Fig. 3.1). The prevailing winds blow from the southwest in summer and from the northwest in winter. On hot summer days, southwesterly winds are laden with air pollutants from the Washington, Baltimore and Philadelphia metropolitan areas that reach the Meadowlands and other areas in New Jersey (http://www.co.hunterdon.nj.us/mun/Holland/Climate.pdf). Air quality in the Meadowlands has been affected by surrounding major highways and transit systems, including the New Jersey Turnpike in the south.

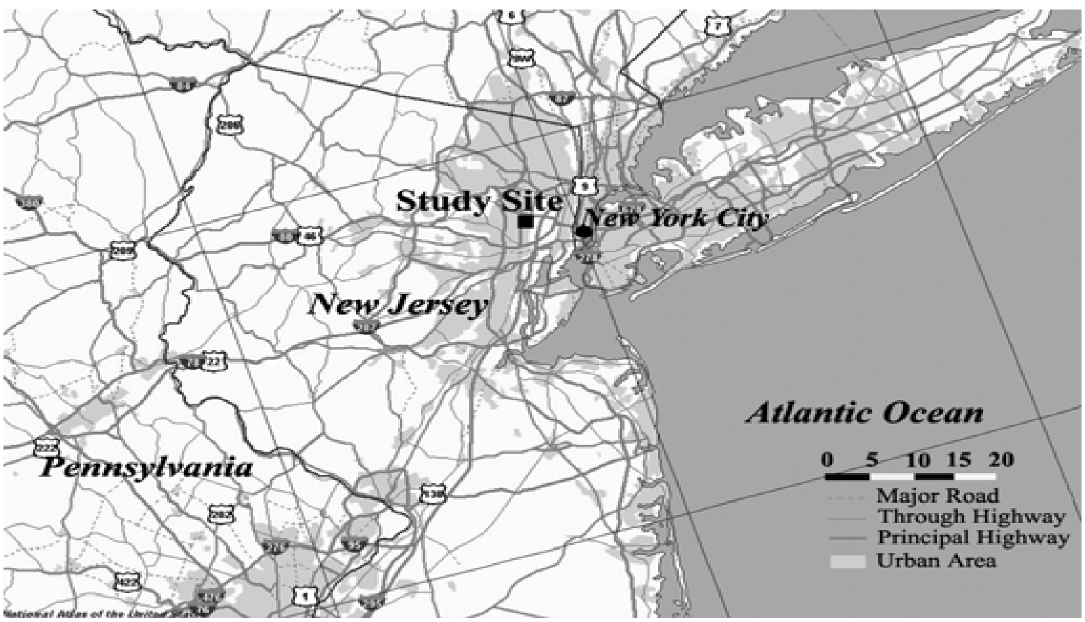


Figure 3.1 Situation of study site/monitoring station in proximity to highways and pollutant source regions.

3.2.2. Sampling Methods

Measurements of NO_x and O_3 were obtained from June 1, 2007 to May 31, 2008 at the Meadowlands Environmental Research Institute (MERI) in Lyndhurst, New Jersey (41ºN, 74ºW), ~ 1 km away from major highways. A 42i-D NO_x analyzer (Thermo Electron Corporation, Franklin, MA) and 49i O_3 analyzer (Thermo Electron Corporation, Franklin, MA) were installed on the roof of the MERI building 8 m above the ground in spring 2007. Air was pumped into the instruments from inlets via two separate plastic tubes. To assure data reliability, reduce bias and associated errors, the sampling and monitoring instruments (analyzers) were calibrated bi-weekly (Munger et al., 1998). Inevitably, due to the large data set, there are minor gaps in the data collection resulting primarily from routine instrument calibrations.

Nitrogen oxides were determined on line by detecting the chemiluminescence in a range of 600 nm to 3,000 nm, and O_3 was measured by the UV photometric method since O_3 molecules absorb infrared radiation at a wavelength of 254nm. The relationship between absorbance of UV-light and O_3 concentration follows the law of Lambert-Beer. Hourly averaged data were derived from the original 5-minute interval data. Based on the Chauvenet's criterion, the data point in each 1-hour range was treated as an outlier if it fell outside of 3 times the standard deviation from the mean: 0.006%, 0.17% and 0.03%, for O_3, NO and NO_x respectively. In addition, consistent with EPA guidelines for NO_2 and O_3, 8-hour, daily and yearly averages were generated from the hourly averaged data. In this case, a valid day is defined as one with at least 75% of the possible 8-hour averages in the day. A day that is less than 75% complete is only considered valid if the daily maximum is greater than the standard (www.epa.gov).

For each day, the hourly averages were grouped into daytime (7:00 hrs to 19:00 hrs) and nighttime (19:01 hrs to 6:59 hrs) based on average solar radiation changes during a day in the spring time of N.J. (www.underground.com). Data collected for these two periods were distinguished mainly by the photochemical activity of O_3 and its precursors, similar to the approach used by Abdul-Wahab et al (2005). Meteorological data obtained from MERI were supplemented with data from Weather Underground, a reliable and accurate web source containing specific localized weather conditions. The primary meteorological parameters were temperature, wind speed, relative humidity and atmospheric pressure.

3.2.3. Data Analyses

Time series plotting techniques were used to visualize the seasonal, monthly, diurnal and weekday versus weekend patterns of NO, NO_2, NO_x and O_3 concentrations. Auto-correlations of the daily pollutant concentrations were conducted to estimate pollutant persistence, and cross-correlations between meteorological variables and pollutants were conducted to test the same day or lagged relationships between them. Multiple linear regression techniques are known to be effective in explaining significant relationships between weather parameters and seasonal characteristics (Ainslie and Steyn, 2007; Hatzianastassiou et al., 2007; Abe et al., 2009). This approach has also been central to the regulatory policy process and is useful in evaluating the risks of air pollution (Dominici et al., 2004; Ainslie and Steyn, 2007). In this study, therefore, the stepwise regression modeled the association between air pollutants and meteorological parameters, similar to the method used by Weisel et al. (1995) and Abdul-Wahab et al. (2005). The regression model was given as follows:

$$y = \beta_0 + \beta_i x_i + \varepsilon + \beta_2 x_2 + \beta_3 x_3 + \ldots + \beta_p x_p$$

where y is the dependent variable (O_3/NO_2), xi is the ith independent variable, β_1 is the regression coefficient, β_0 is the intercept, p is the number of independent variables, and ε is the error with mean zero. To overcome possible multicollinearity among the independent variables in the stepwise regression and to validate the relationship between air pollutants and meteorological variables, the principal component analysis (PCA) described by Liu et al. (2003) and Abdul-Wahab et al. (2005) was employed. The PCA, a special case of factor analysis, was conducted by transforming highly correlated independent variables, particularly O_3 precursors and wind speed, into principal components that are independent of each other (Liu et al., 2003; Abdul-Wahab et al., 2005; Gvozdić et al., 2011). Statistical analyses in this study were conducted with Statistical Package for the Social Sciences (SPSS) version 18.0, Statistical Analysis System (SAS) 9.2, and MINITAB 16 software packages.

3.3. Results and Discussion

3.3.1. Seasonal Patterns

The seasonal cycles of O_3 and NO_x were driven by their response to local meteorological conditions and resultant chain reactions among them. The average monthly/annual concentrations and variations of O_3, NO_x, NO and NO_2 (Table 3.1), and time series graph show opposing trends in NO_x and O_3 concentrations (Fig. 3.2), either amplified or diminished by a summer maximum and winter minimum.

Table 3.1 Summary of monthly and yearly average concentrations and variations

Month	Concentrations (ppb)	O_3	NO	NO_x	NO_2
1	Average± StdEv	14.1±6.3	12.1±19	29.0±24.7	16.9±7.1
	Minimum	1.6	0.9	7.0	6.0
	Maximum	24.9	79.6	109.9	30.3
2	Average± StdEv	15.2±8.8	7.3±7.3	23.2±13.8	15.9±7.2
	Minimum	1.1	1.1	7.0	5.5
	Maximum	29.4	30.0	58.0	29.0
3	Average± StdEv	23.2±8.4	6.4±8.0	22.7±16.3	16.3±9.1
	Minimum	7.2	0.6	4.6	4.0
	Maximum	36.2	38.0	72.8	34.7
4	Average± StdEv	25.0±8.8	7.5±6.7	27.7±16.3	20.2±10.8
	Minimum	7.3	0.8	4.3	3.4
	Maximum	41.7	26.0	73.9	53.5
5	Average± StdEv	27.4±9.3	6.3±7.8	24.0±17.1	17.7±10.4
	Minimum	3.8	0.6	5.2	4.7
	Maximum	45.4	31.7	60.1	37.4
6	Average± StdEv	36.2±11.6	3.7±4.0	19.3±9.5	15.5±6.2
	Minimum	15.3	0.5	4.6	4.1
	Maximum	63.0	15.7	46.9	33.3
7	Average± StdEv	30.5±11.9	2.7±2.3	17.0±9.3	14.3±7.6
	Minimum	9.9	0.2	4.4	4.1
	Maximum	55.6	9.6	38.4	34.0
8	Average± StdEv	29.0±12.9	5.0±3.4	19.6±8.2	14.6±5.9
	Minimum	5.2	0.3	5.4	3.3
	Maximum	52.0	13.7	39.9	28.6
9	Average± StdEv	22.5±9.4	5.0±5.0	20.6±11.5	15.5±7.0
	Minimum	5.5	0.6	7.2	5.8
	Maximum	46.6	17.8	52.1	34.3
10	Average± StdEv	16.3±8.0	7.4±7.9	23.8±14.3	16.4±6.7
	Minimum	6.7	0.5	8.2	7.2
	Maximum	38.1	32.4	65.5	33.1
11	Average± StdEv	11.5±6.0	5.4±6.5	18.2±10.3	12.9±4.9
	Minimum	0.2	0.8	6.0	5.1
	Maximum	20.7	31.0	47.7	24.8
12	Average± StdEv	11.1±7.2	9.1±12.3	25.8±17.1	16.7±8.0
	Minimum	0.3	0.7	6.1	5.4
	Maximum	27.2	64.0	94.4	46.5
Yearly	Average± StdEv	21.9±11.9	6.5±8.9	22.6±15.0	16.1±7.9
	Minimum	0.2	0.2	4.3	3.3
	Maximum	63.0	79.6	109.9	53.5

The monthly average diurnal variations of NO_x and O_3 display a gradual increase in O_3 variations from March to June, with pronounced variations until September, followed by a decrease through December. Such variations are apparently caused by photochemical differences across seasons. In months with relatively lower diurnal variations, the more prominent O_3 peaks were apparently obscured by lower peaks in months with relatively higher variations. Conversely, NO_x showed no pronounced diurnal variations except in April and May.

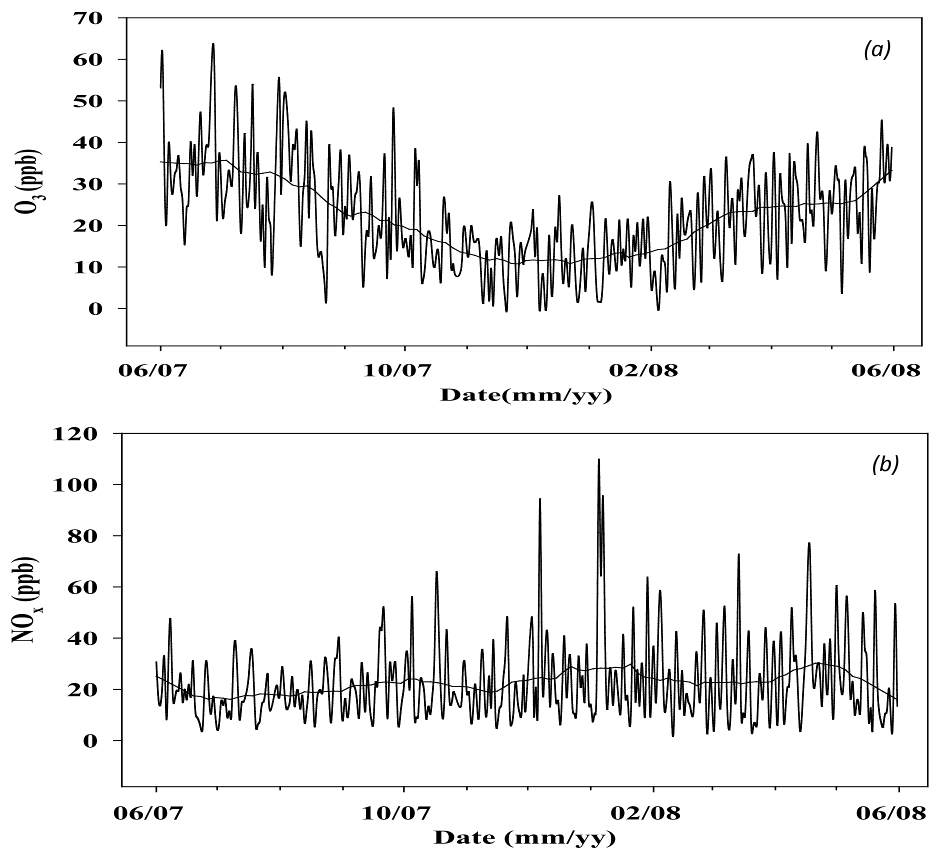


Figure 3.2 Temporal variations of the daily average (a) O_3 and (b) NO_x concentrations during the one-year period (June 1, 2007 to May 31, 2008). Ozone displays summer maxima and winter minima with pronounced variations in the colder months; NO_x displays the opposite trend. Variations in NO_x are greater in the summer than in winter.

The annual average O_3 concentration was 21.9ppb, with pronounced variations mainly in summer. Daily mean O_3 values displayed summer highs with a gradual decline into spring (March to August) and winter lows (December to February). The monthly NO_x concentrations ranged from 17.0 to 29.0ppb and the annual average was 22.6ppb with an increasing trend towards the winter. The seasonal variations in NO_x were common to several studies (Khoder, 2009; Smith et al., 2011; Guttikunda and Gurjar, 2011), attributed to the poor dependence of NO_x on meteorological conditions. Similar trends observed for NO and NO_2 consisted of a winter peak and summer minima (Khoder, 2009; Bigi and Harrison, 2010; Kan et al., 2010; Smith et al., 2011). This enhanced seasonality of NO_x levels in winter may be partly attributed to increased fossil fuels for domestic heating and driving. Therefore, anthropogenic sources seem to play a greater role in NO_x build-up than local meteorological conditions in winter; NO from traffic emissions is converted to NO_2 while the photochemical formation of O_3 is inhibited by the lack of intense solar radiation (Sadanaga et al., 2008; Geddes et al., 2009).

3.3.2. Monthly Patterns

Figure 3.3 shows variations in monthly average NO_x and O_3 concentrations. Average O_3 ranged from 11.1ppb to 36.2ppb, with wide distribution ranges from June through August, and increased in the winter. The distribution ranges displayed for NO_x were narrower in summer, June to August, than in winter and spring, December to May. Table 3.2 compares the average monthly concentrations of NO_x and O_3 with other locations in the

US East Coast. Maximum O_3 values in the study area, 94.3, 94.0, 93.0 and 92.0 ppb, respectively, were relatively higher than most sites within the EPA network.

Table 3.2. Comparisons of O_3 concentrations with other nearby sites on US East Coast

Sites	8-Hour Mean (ppb)					
	1st Max	2nd Max	3rd Max	4th Max	Days > Std	# Days
[1]Lyndhurst, NJ (This Study)	94.3	94.1	92.7	92.1	10	282
[2]EPA Standard	75 (8-hour average)					
[3]Oceanville, NJ	76	72	72	72	1	173
[4]Leonia, NJ	89	89	85	82	6	172
[5]Bayonne, NJ	90	86	81	81	7	175
[6]East Brunswick, NJ	94	89	86	83	13	183
[7]West Long Branch, NJ	89	86	86	83	10	183
[8]Chester, NJ	86	84	82	81	9	162
[9]Jackson, NJ	100	90	85	85	15	182
[10]Albany, NY	88	84	78	77	5	195
[11]New York, NY	84	81	78	77	5	203
[12]New York, NY	90	87	82	82	6	207
[13]East Farmingdale, NY	94	93	85	83	8	202
[14]White Plains, NY	101	91	86	82	10	186
[15]Greenwich, CT	105	102	90	88	14	172
[16]Middletown, CT	91	83	83	82	8	182
[17]Hagerstown, MD	84	80	78	75	3	212
[18]Baltimore, MD	82	65	62	62	1	212
[19]Lynn, MA	86	81	79	78	5	182
[20]Boston, MA	83	73	72	72	1	178
[21]Cooleemee, NC	89	84	82	81	6	214
[22]Greensboro, NC	88	83	83	81	5	195
[23]Pittsburgh, PA	84	79	79	79	7	214
[24]State College, PA	81	77	74	74	2	209
[25]East Providence, RI	88	86	85	77	4	175
[26]Mclean, VA	102	90	81	80	6	213
[27]Hampton, VA	88	82	79	79	4	179

[1]Present study, [2-27] http://www.epa

The 8-hour average O_3 concentrations calculated for 8 consecutive hours on a given day, denoted as "Days > Std", exceeded the EPA standard of 75ppb on 10 days at the study site and were higher than levels at EPA sites. On the other hand, the annual average NO_2 concentration (53ppb) satisfied the EPA requirement (Table 3.3). Despite high maximum NO_2 levels, the average concentration (16.1ppb) was within the EPA standard and there were no exceedances ("# Exceed"). The 1st and 2nd maximum and mean concentrations were also comparable to other sites.

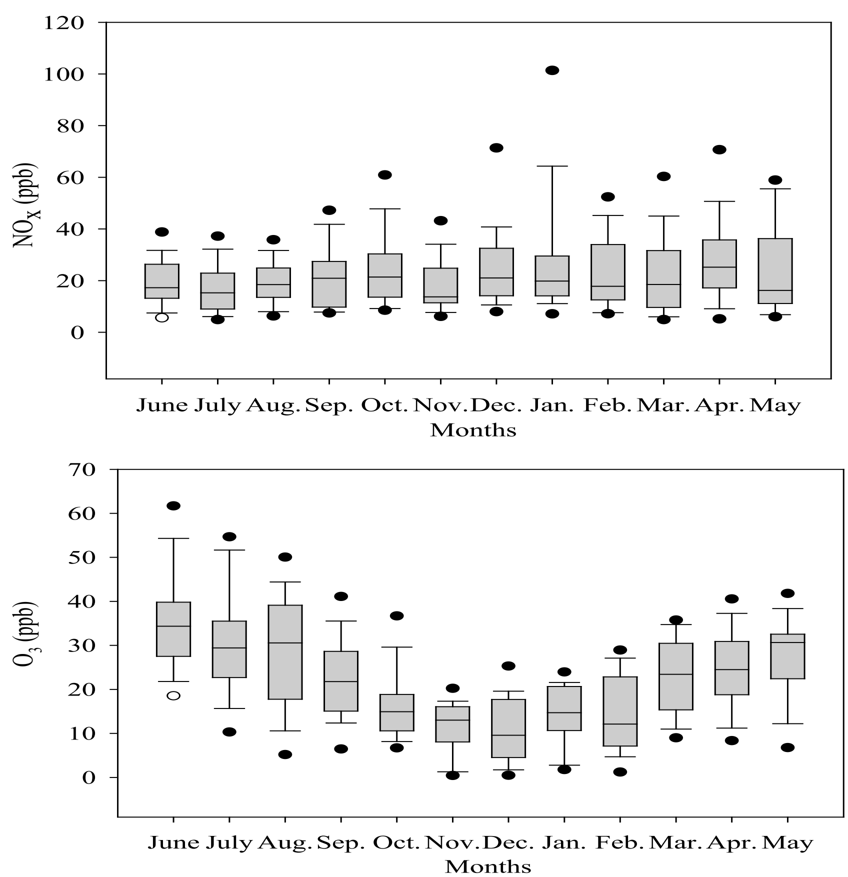


Figure 3.3. Seasonal variations of monthly average NO_x and O_3 concentrations. Box plots show (a) NO_x and (b) O_3 concentrations from June 1 2007 – May 31, 2008.

Table 3.3. Comparisons of NO_2 concentrations with other sites on US East Coast

Sites	1-Hour Mean (ppb)			Annual Mean
	1st Max	2nd Max	Mean	# Exceed
[1]Lyndhurst, NJ (This Study)	77.4	72.7	16.1	0
[2]EPA Standard	-	-	53	-
[3]Leonia, NJ	84	83	19	0
[4]East Orange, NJ	79	76	21	0
[5]Bayonne, NJ	82	80	18	0
[6]East Brunswick, NJ	56	53	11	0
[7]Chester, NJ	49	48	6	0
[8]Elizabeth, NJ	93	89	27	0
[9]New York, NY	89	87	25	0
[10]New York, NY	97	90	36	0
[11]Lynn, MA	61	61	8	0
[12]Boston, MA	54	50	7	0
[13]Pittsburgh, PA	113	94	14	0
[14]State College, PA	42	41	6	0
[15]East Providence	31	31	6	0
[16]Mclean, VA	72	68	13	0
[17]Westport, CT	62	60	12	0
[18]New Haven, CT	64	62	15	0
[19]Beltsville, MD	49	49	9	0
[20]Baltimore, MD	78	73	18	0
[21]Winston-Salem, NC	61	61	11	0
[22]Charlotte, NC	59	58	11	0

[1]Present study, [2-22] http://www.epa

3.3.3. Diurnal Variations of NO_x and O_3

Figure 3.4 presents the diurnal variations of NO_x and O_3. The major O_3 peak lasted from late morning until early afternoon. This photochemically active period was apparently accentuated by a preceding build-up of precursors and increased electrical loads for air conditioning particularly in the summer. This feature demonstrates O_3 dependence both on emissions and meteorological conditions. The lowest O_3 concentration (~12ppb) occurred around 6:00 A.M. but it increased steadily to its highest peak (~33ppb) around 2:00 P.M. High levels were maintained until 4:00 P.M. but decreased rapidly until 8:00 P.M., then gradually until 6:00 A.M. of the next day.

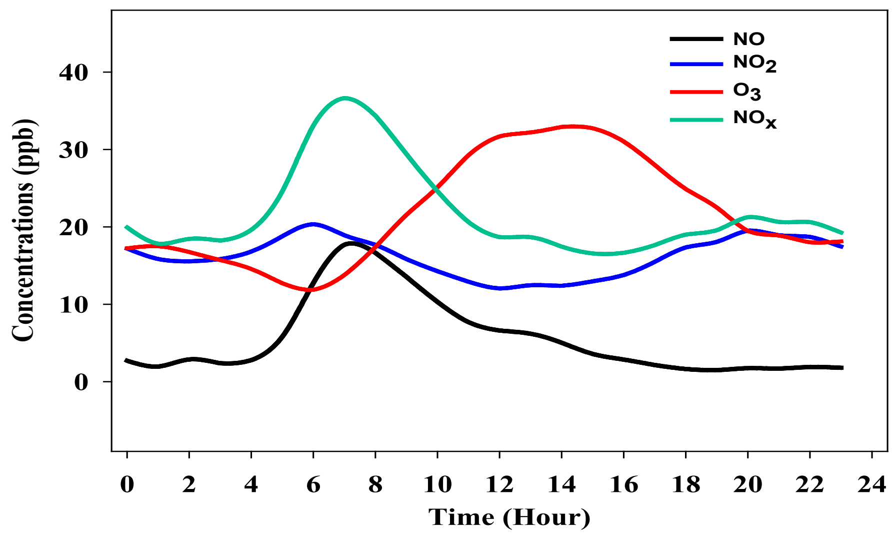


Figure 3.4. Diurnal variations of NO, NO_2, NO_x and O_3 measured over 24-hour period. The morning NO, NO_2, NO_x peaks are followed by an O_3 photochemical peak later in the day.

Conversely, NO_x displayed morning and late afternoon peaks which coincided with rush-hour traffic. The accumulation process commenced from 4:00 A.M., reached a peak of 38ppb around 7:00 A.M., then diluted rapidly until 10:00 A.M. to ~18ppb; however, while NO_2 was also characterized by two daily peaks, NO was devoid of a second peak. The earlier NO_2 peak might be caused by nighttime balancing, where concentrations were higher than NO at night due to the oxidation of NO to NO_2 by O_3, in the absence of solar radiation (Song et al., 2011); these levels were maintained until early morning hours. The accumulation of NO_2 contributes to O_3 formation; however, O_3 is “scavenged” where there is an abundance of NO, resulting in lower O_3 concentrations in heavy traffic (Abdul-Wahab et al., 2005; Ainslie and Steyn, 2007; Sadanaga et al., 2008; Geddes et al., 2009).

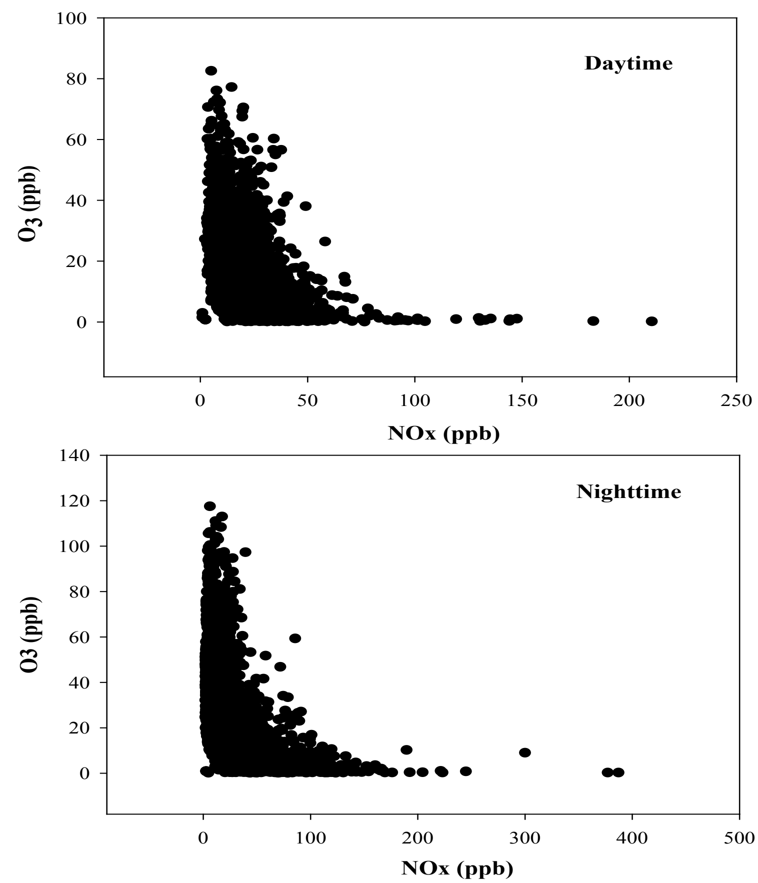


Figure 3.5 Daytime/nighttime comparison of the correlations between NO_x and O_3. Maximum O_3 concentrations are enhanced by NO_x precursors in the day but are inhibited at night.

The effect of NO_x on O_3 is further demonstrated when compared to specific NO_x concentrations. Maximum O_3 levels during the day were at times much lower than in the nighttime (Figure 3.5), indicating that NO_x could inhibit O_3 concentrations more significantly in the daytime than at night. Hence, photochemical reactions greatly inhibited at night, were apparently the primary factor responsible for these differences. Abdul-Wahab et al. (2005) found in their stepwise multiple regression analysis, that solar levels contributed significantly to high daytime O_3 concentrations with NO as the principal precursor, while at night, NO_2 was the primary influence.

Temperature inversions, a feature that is common in urban areas, may also contribute to this process (Hussein et al., 2006). These are caused when high daytime surface temperatures generate convective winds within the urban boundary layer, while at night a reversal of daytime atmospheric mixing occurs with the absence of solar heating (Hussein et al., 2006). Atmospheric convection decreases with increasing stabilization in the urban boundary layer and the development of an inversion layer. Atmospheric stability restricts vertical motions and increases pollution concentrations near the surface, particularly when accompanied by radiation induced inversions during early morning hours, thus enhancing NO_x concentrations (Ainslie and Steyn, 2007; Guttikunda and Gurjar, 2011).

3.3.4. Weekday/Weekend Comparisons

In Figure 3.6, both NO_x and O_3 displayed smaller variations on weekends than on weekdays. NO_x and O_3 levels were similar on Saturday and Sunday and their variations were higher on Saturday than on Sunday. Of the two components of NO_x, however, NO_2 showed less pronounced weekday variations than NO (Fig. 3.7). The contrast between

NO_x and O_3 concentrations on weekdays and weekends were further demonstrated by their diurnal variations, often influenced by cyclical patterns of local meteorological factors. While O_3 concentrations on weekdays and weekends displayed a unimodal peak in the early afternoon hours, bi-modal peaks were observed for NO_x and NO_2; there was a single NO peak. The first NO_x peak was more significant than the second, particularly for weekday concentrations.

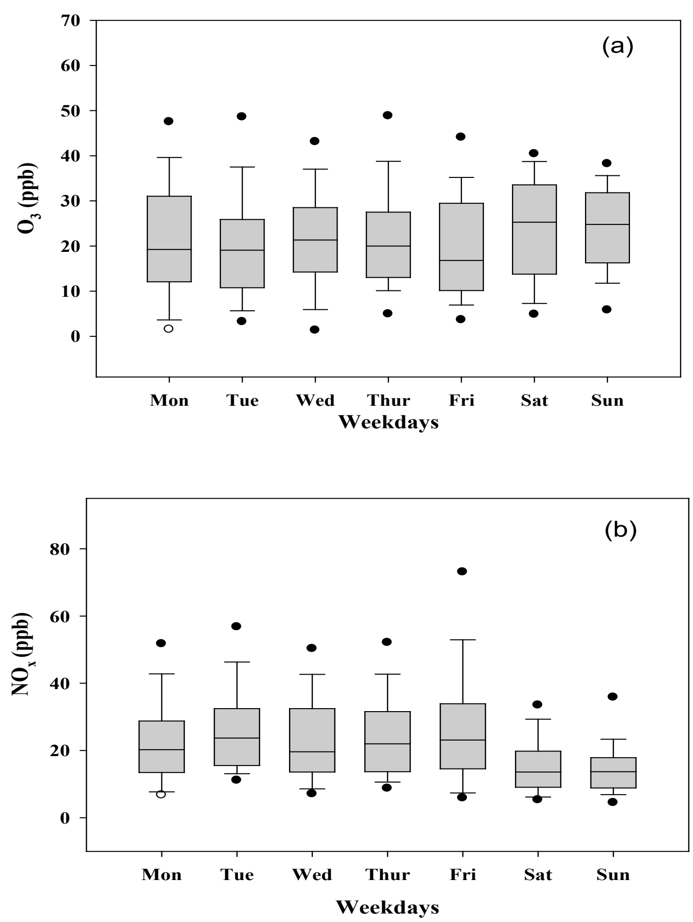


Figure 3.6 Variations in yearly average NO_x and O_3 concentrations on weekdays. There are smaller variations on weekends (Sat. and Sun.) than on weekdays especially for NO_x.

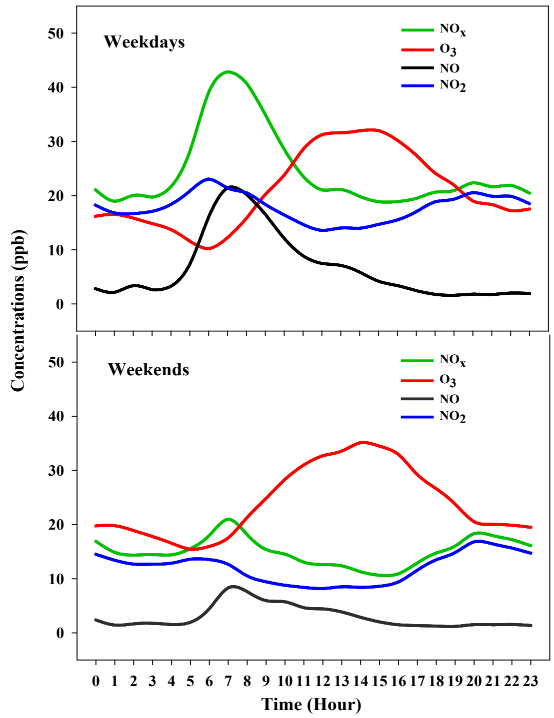


Figure 3.7 Diurnal variations of NO_x and O_3 on weekdays and weekends. While the O_3 peak is similar for both weekdays and weekends, NO, NO_2 and NO_x display much smaller morning peaks on weekends than on weekdays.

The average NO_x concentrations were much lower on weekends than on weekdays, reflecting reduced levels of vehicular emissions on weekends. Conversely, average O_3 concentrations on weekends were higher than on weekdays. The negative correlation in hourly average NO_x and O_3 concentrations suggested that VOC rather than NO_x, contributed to elevated O_3 concentrations, by the so-called "weekend effect". Lower NO levels and VOC emissions on weekend mornings consume less O_3 which accumulates later by photochemical reactions (Pudasainee et al., 2006). While a direct relationship existed between NO_x emissions and ambient concentrations on weekdays, the reduction of NO_x did not automatically lead to a proportional decrease in O_3 levels on weekends. In fact, several studies have shown that O_3 levels in the ambient air paradoxically increased when emissions of NO_x decreased (Heuss et al., 2003; Bernstein et al., 2004; Sadanaga et al., 2008). Similar observations were made in a potential nonattainment area of Cincinnati, Ohio where a reduction in NO emissions contributed to an increase in local O_3 (Torres-Jardon and Keener, 2006). Khoder (2009) also found many sites with elevated O_3 on weekends when traffic and O_3 precursor levels were substantially reduced, since motor vehicle emissions near busy streets are known to contribute to high local NO_x concentrations.

Apparently, temporal changes, proximity to emission source and meteorological factors, particularly temperature differences play a smaller role in weekend O_3 behavior; however, they do not necessarily explain the weekend effect but may modify it (Munger et al, 1998). Weekday/weekend differences in O_3 are intricately related to interactions with its chemical precursors: NO_x and VOC, respectively (Sadanaga et al., 2008). Compared to weekdays, however, O_3 could range much smaller for specific concentrations of NO_x on the weekends, even though NO_x concentrations were relatively lower than O_3 concentrations (Fig. 3.8). This might be an indication that on weekends, the inhibition effect of NO_x on O_3 concentrations was more significant.

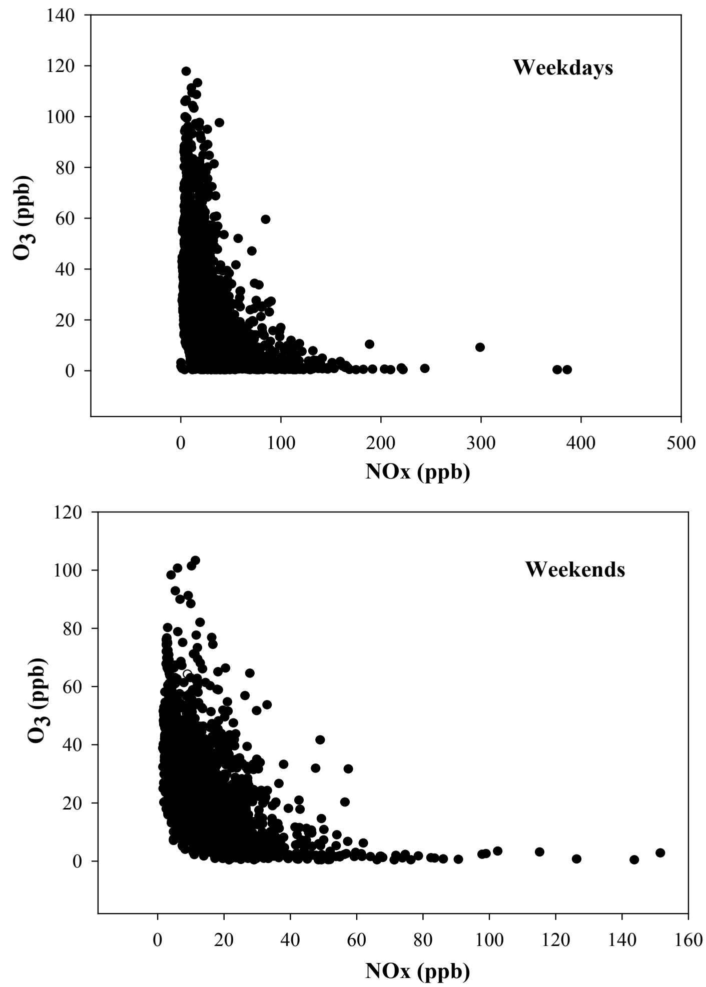


Figure 3.8 Weekday/weekend comparison of the correlations between NO_x and O_3.

3.4. Meteorological Influences

When examined by individual series with lagged observation to test for autocorrelations, the most significant correlation was at 24 hour lags. Except for O_3 and temperature which showed a consistent pattern of positive autocorrelations, there were slight variations among the other variables after 24 hour lags. On the other hand, in the test of cross-variability O_3 showed a negative correlation with all meteorological variables except temperature. While O_3 demonstrated a negative correlation with NO, there was a positive correlation with NO_2 up to 48 hour lags.

Table 3.4. Model summary of variables associated with pollution concentrations

Independent Variables Coefficients (Cum R^2)	Dependent Variables O_3	O_3	Log O_3	NO_2	log NO_2
Temperature	.412***	.388***	.009***	.029	.000
	(.260)	(.260)	(.169)	(.005)	(.006)
Relative humidity	-.375***	-.361***	-.011***	.070**	.004***
	(.456)	(.456)	(.411)	(.109)	(.156)
Barometric pressure	-7.349***	-.736***	-.172**	1.729	.090*
	(.499)	(.494)	(.448)	(.165)	(.235)
Wind speed	.598***	.422*	.015**	-.449***	-.019***
	(.514)	(.514)	(.489)	(.333)	(.419)
NO	—	-.253*	-.018***	.657***	.015***
		(.522)	(.527)	(.537)	(.537)
NO_2	—	.059	.009***	—	—
		(.523)	(.544)		

p-values < 0.05*
p-value <0.01**
p-value <0.001***

Table 3.4 presents a summary of the variables associated with O_3 and NO_2 concentrations in the stepwise regression model. The analysis showed that with one unit increase in temperature and wind speed, mean O_3 concentrations increased by 0.4 and 0.6 units respectively, when the other weather variables were kept constant. The high numerical value of the coefficient corresponding to barometric pressure (-7.349), demonstrated the sensitivity of O_3 concentrations to a single unit change; the maximum, minimum and average values were 30.74Hg, 28.90Hg and 30.02Hg, respectively. Variability in O_3 with the meteorological parameters increased slightly, from 51% to 52% after NO and NO_2 were included in the regression model, and the log of O_3 showed greater variability (54%); variations in the log of NO_2 were similar (>53%), although NO_2 was apparently less dependent on meteorological variables. With one unit increase in wind speed, NO_2 concentrations decreased by 0.4 units.

In the stepwise regression the diagnostic tests for regular normality assumption of the residuals for NO_x and O_3 confirmed independent normal distribution (Fig. 3.9); however, significant correlation among certain independent variables, notably NO, NO_2 and wind speed, was expected. Therefore, these variables were transformed into principal components, making them independent of each other, to improve the estimate of variability among them. From principal component regression equations, partial regression coefficients and constants were computed. The transformation of the standardized to general linear regression for $LogO_3$ and $LogNO_2$ was computed, with temperature (Temp), wind speed (Wind), relative humidity (Hum) and atmospheric pressure (Press); the final general partial regression model given as:

$$\mathrm{Log}(O_3) = 2.085584 + .007906\ \mathrm{Temp} - .009584\ \mathrm{Hum} - .01629\ \mathrm{Press} + .022426\ \mathrm{Wind} - .025547\ \mathrm{NO} - .011489\ \mathrm{NO_2}$$

$$\mathrm{Log}(NO_2) = .55519229 + .00098969\ \mathrm{Temp} + .006073206\ \mathrm{Hum} + .009186\ \mathrm{Press} - .03839\ \mathrm{Wind} + .02248296\ \mathrm{NO}$$

Consistent with the linear regression analysis, PCA demonstrated similar relationships between the log of O_3 and meteorological variables. There is a positive association with temperature and wind speed, whereas the coefficients of relative humidity and barometric pressure are negative. Similarly, there were positive associations between the log of NO_2 and all variables except with wind speed, comparable to the coefficients in the linear regression model. In both time series and regression models, wind speed accounted for the greatest variability in the concentrations O_3 and NO_2 than any other meteorological variable.

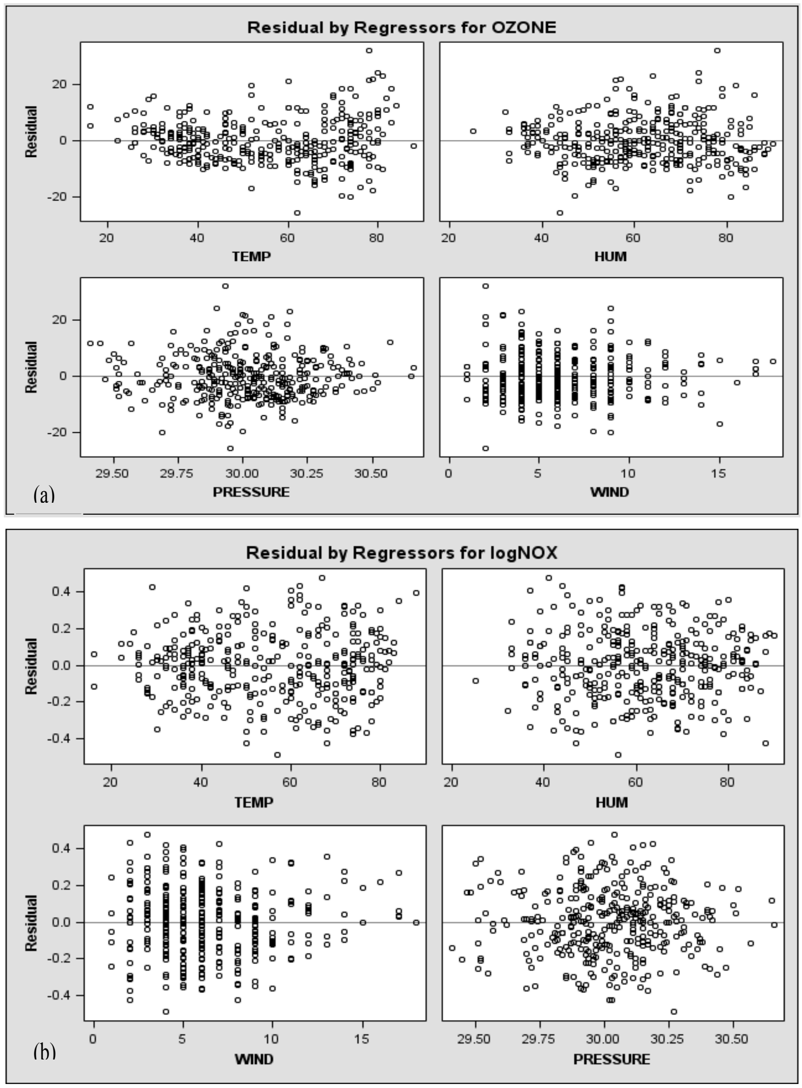


Figure 3.9 Residual plots of the regression model for (a) O_3 and (b) NO_x. Residuals show symmetry and are along a straight line with no specific pattern against meteorological variables around zero.

The overall trend was an inverse relationship between wind speed and O_3 concentrations (Fig. 3.10). The dispersal of pollutants by strong winds through vertical mixing and forced convection apparently contributed to lower O_3 and NO_2 concentrations. Ito (2007) found that NO_2 exhibited a very strong negative correlation with wind speed; however, while pollutant concentrations are generally inversely proportional to wind speed, concentrations may increase downwind even at high wind speeds (Kim et al., 2004; Ainslie and Steyn, 2007). The monitoring site was downwind of constant traffic on the New Jersey Turnpike and other heavily traveled roadways. While NO_2 concentrations decreased with wind speed, the increase in O_3 concentrations could be due to prevailing southwesterly winds from east coast urban areas, the direction associated with transport of precursor pollutants over hundreds of miles downwind from their original emission sources before being transformed into O_3 (Farrell, 1999; Ainslie and Steyn, 2007).

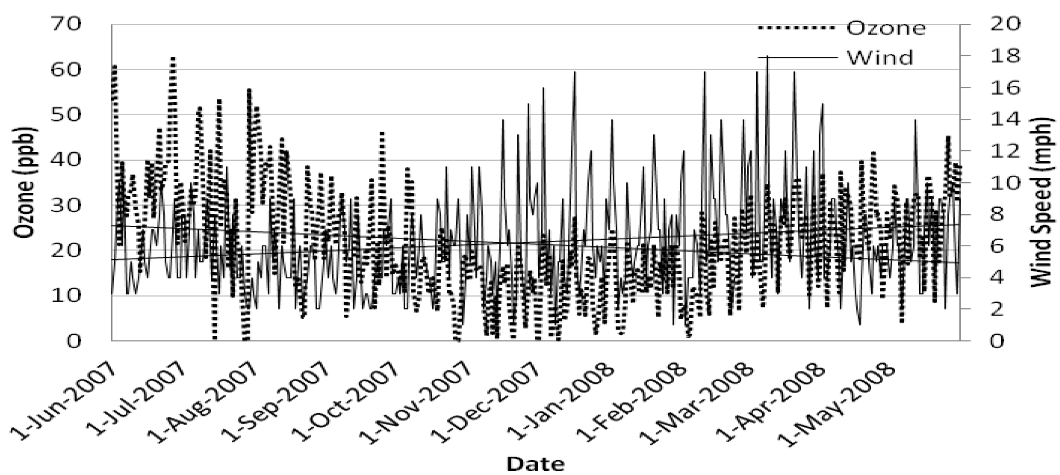


Figure 3.10 Inverse relationship between O_3 and wind speed. Ozone concentrations decrease as wind speeds increase.

While high O_3 concentrations are generally associated with conditions that suppress vertical mixing, such as relatively light winds and thermal inversions in the atmosphere

(Sillman, 1999, Godish, 2004), previous studies have shown a non-linearity between concentrations of gas-phase pollutants and wind speed (Bigi and Harrison, 2010). In a two-stage model, Kim et al (2004) found that downwind direction was an important determinant of increased exposure to traffic pollutants. Abdul-Sabal et al. (2005) reported that O_3 was weakly correlated with wind speed but was positively correlated with wind direction. Similar observations were made by Ainslie and Steyn (2007) of higher O_3 concentrations downwind in a multiple linear regression model.

3.5. Conclusions

In this study, the temporal variability of NO_x and O_3 concentrations was shown to be governed by the seasonality of atmospheric processes which acted interdependently. Based on the analysis, the seasonal patterns of NO_x and O_3 are altered under variable atmospheric conditions and chemical mechanisms that are inextricably related. Elevated O_3 concentrations are primarily influenced by wind, solar energy and temperature as well as chemical inter-conversions with NO_x. These chemical characteristics exist on short timescales in concert with emission sources, particularly the mechanisms that drive NO either to destroy or enhance ambient O_3 levels. Although adjustments were made to reduce bias caused by such factors, the methods used in this study could not authenticate the level of uncertainty pertaining to the role of these factors. Despite this limitation, the evidence is adequately consistent and plausible to draw reasonable conclusions.

If these results can be replicated at other locations, it may be possible to reduce future exposure to pollutants by developing a health and weather risk-warning system, based on the patterns of high pollutant concentrations. In terms of minimizing concentrations of pollutants from the perspective of abatement, this approach presents additional challenge to the attainment of in-state controls. It can increase the benefits of improved ambient air quality that have not been completely achieved. The health implications of O_3 exposure in the summer and on weekends when its association with atmospheric conditions and chemical processing are the strongest, should be explored in future studies.

Chapter 4: Impact of gas-phase air pollution on public health in northeastern New Jersey[1]

Abstract

To characterize the impact of urban air pollution and local weather conditions on human health, ambient concentrations of nitrogen oxide (NO_x) and ground-level ozone (O_3) were measured over the Meadowlands in Lyndhurst, NJ (41ºN, 74ºW) from June 1, 2007 to May 31, 2008. Meteorological parameters, including temperature, wind speed, relative humidity and barometric pressure, were obtained from Weather Underground, to analyze the dependence of air pollutants on atmospheric conditions. Respiratory health data were used to examine the relationship between pollutants and hospital admissions through path analytic models by using stepwise multiple regressions and bivariate correlations.

There was a positive relationship between hospital admissions and personal exposure to NO_2 (r=0.359) and NO_x (r=0.317) over the short-term. With one unit increase in O_3, respiratory hospital admissions increased by 0.7 (95% CI: 0.254, 1.23) at 2 lag days and by 0.5 (95% CI: 0.107, 0.969) for NO_x. Unlike O_3, NO_x did not show consistent seasonal behavior; however, respiratory hospital admissions were not strongly distinguished by high O_3 levels in the summer. The lowest NO_x concentrations occurred in months with maximum photochemical activity. There were 31 more hospital admissions in the fall, 29 in winter and 32 in spring than in summer, if other factors were kept constant. This study provides evidence for the need of developing additional plans for protection against respiratory illnesses and for setting improved air quality standards in this region.

[1] This chapter is currently in review by the Environmental Research Journal.

4.1. Introduction

Previous studies have shown that nitrogen oxides (NO_x) and ozone (O_3) pose major risks to public health (Blanchard and Tanenbaum 2003; Torres-Jardon and Keener 2006; McConnell et al. 2010; Namdeo et al. 2011). While there has been a steady decline in NO_x emissions typically below the National Ambient Air Quality Standard (NAAQS), O_3 levels have consistently exceeded the national standard in many areas (Godish 2004; Williams et al. 2009; USEPA 2010). Lower levels of NO_x can result in higher O_3 levels (Kelly et al. 2011). Nitrogen oxides comprise a mixture mainly of nitric oxide (NO) and nitrogen dioxide (NO_2) (Samet and Utell 1990; Brook et al. 2004; WHO 2003). They form in the atmosphere through the reaction of lightning with molecular oxygen and other processes involving naturally occurring nitrogen and volatile organic compounds (VOCs) (Sillman 1999; Godish 2004; Brown et al. 2006); however, emissions from anthropogenic sources exceed natural sources (Godish 2004). In the eastern United States, vehicular traffic contributed more than half the total NO_x emissions. Therefore, concentrations are known to be higher in heavily travelled areas (Godish 2004; Tramuto et al. 2011; Washam 2011).

Ground-level O_3 comes from two major sources: regionally from the upper stratosphere via convection movements (Godish 2004), and locally by photochemical reactions. The latter is largely dependent on NO_x concentrations that act as a catalyst in cycles to produce O_3 and to oxidize VOCs (Brown et al. 2006; Song et al. 2011). With the addition of RO_2, a reactive oxygen-containing molecule such as a VOC, NO is consumed when oxidized to NO_2, increasing O_3 formation in bright sunlight or high energy radiation (Atkinson 2000; Seinfeld and Pandis 2006; Yerramilli et al. 2011). Many VOCs produced by photochemical reactions are short-lived or may be found in trace amounts well below the detection limit of current sampling methods (Godish 2004); however, NO_x rapidly interconvert within minutes to hours (Seinfeld and Pandis 2006; Wei et al., 2007; Sadanaga et al. 2008; Geddes et al. 2009).

Although NO_x and O_3 have varied reaction properties, they may cause similar acute and chronic effects to certain systems and organs in the human body, ranging from minor upper respiratory irritation to chronic and acute respiratory infections (Fusco et al. 2001; Peel et al. 2005; Orazzo et al. 2009; Ji et al. 2011; Washam 2011). They can aggravate pre-existing heart, lung disease and asthmatic conditions causing premature mortality and reduced life expectancy (WHO 2003; Peters et al. 2009; Qian et al. 2010; Kan et al. 2010). The production of NO under oxidative stress conditions secondarily generates strong oxidizing agents and reactive nitrogen species that may modulate the development of chronic airway diseases and/or amplify inflammatory response (Ricciardolo et al. 2004; Ji et al. 2011). Clinical data indicate that NO_x and O_3 levels cause adverse physiologic responses and functional deficits from effects on airways and chronic respiratory conditions in both healthy individuals and asthmatic patients (Samet and Utell 1990; Torres-Jardon and Keener 2006; Orazzo et al. 2009; Ji et al. 2011). Fewer studies, however, have analyzed NO_x and O_3 (Godish 2004) to characterize the underlying mechanisms of toxicity and respiratory impacts from the perspective of spatial and seasonal patterns (Ito et al. 2007). In addition, many monitoring networks lack local information, particularly for traffic-generated pollutants known to vary on small scales (Kan et al. 2010; Giles et al. 2011).

The health-based O_3 standard has been exceeded in many states (Sillman 1999; Godish 2004) including New Jersey (Madsen and Mottola 2003), where 19 and 29 unhealthy O_3 pollution days occurred in 2004 and 2005, respectively (NJDEP 2006). In central and northern New Jersey, average O_3 levels greater than 0.06 ppm, caused emergency room (ER) visits 28% more frequently, suggesting that O_3 can have serious health effects at levels below the EPA standard (0.075 ppm/8-hour period) (Weisel 1995; Madsen and Mottola 2003; USEPA 2010). This chapter assesses the impact of NO_x and O_3 on human health due to the influence of local meteorology in the Hackensack Meadowlands. It may provide insight and consequent alleviation of respiratory and inflammatory airway diseases through further reduction of targeted gas-phase pollutants. It will also highlight the fundamental mechanisms driving NO when altered under atmospheric conditions, its role in enhancing O_3 concentrations and potential health effects that are not fully studied in New Jersey.

4.2. Materials and Methods

4.2.1 Study Site

The Meadowlands District is located in northeastern New Jersey, a heavily industrialized and densely populated region of the United States. It comprises residential developments and occupies 83 square kilometers approximately four miles west of New York City (Fig. 4.1). The prevailing winds are from the northwest in winter. On summer days, winds are usually from the southwest laden with air pollutants from the Washington, Baltimore and Philadelphia metropolitan areas. Air pollution in the Meadowlands has worsened with extensive highways and transit systems (Madsen and Mottola 2003), such as Route 46 in the east, Routes 1 and 9, and a freight rail line in the south, the Port Authority Trans-Hudson (PATH) commuter rail lines; and Route 17, Pascack Valley Line and Kingsland rail line in the west.

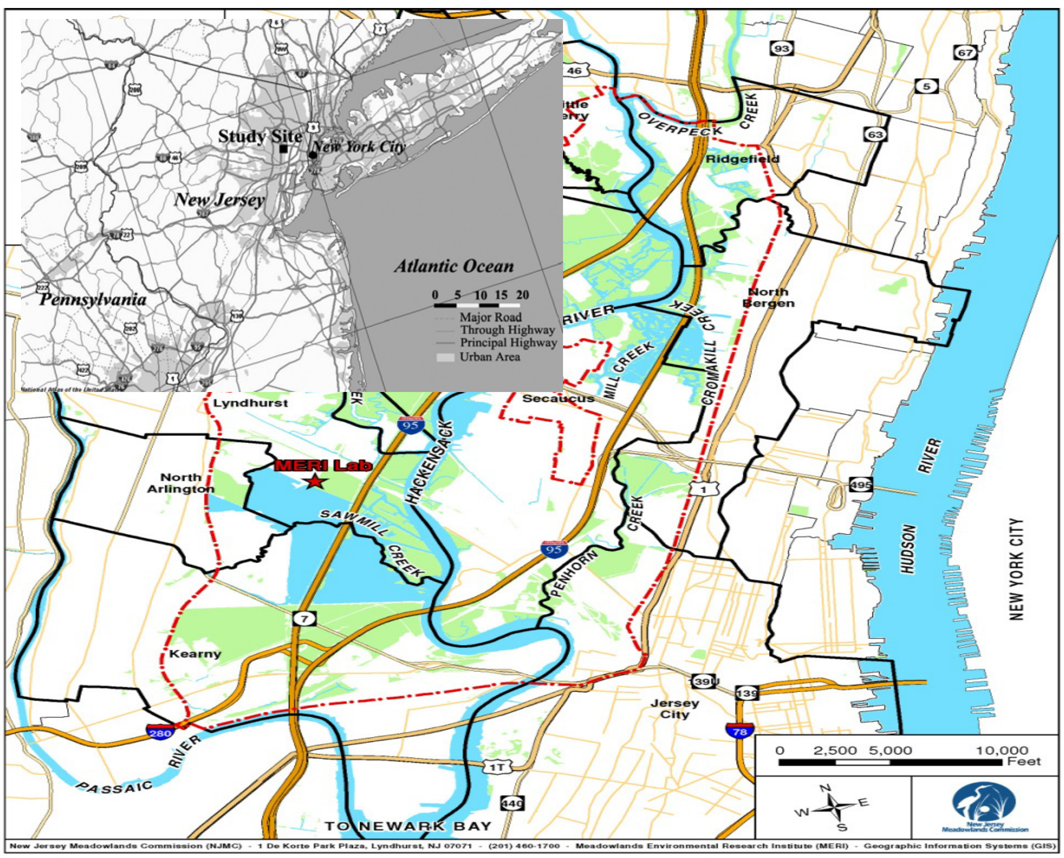


Figure 4.1 Map of study site showing situation of monitoring station in proximity to highways and pollutant source regions.

4.2.2 Sampling Methods

To obtain ambient concentrations of NO_x and O_3, a 42i-D NO_x analyzer (Thermo Electron Corporation, Franklin, MA) and 49i O_3 analyzer (Thermo Electron Corporation, Franklin, MA), were installed on the roof of the Meadowlands Environmental Research Institute (MERI) at Lyndhurst, New Jersey (41ºN, 74ºW), ~ 1 km away from major highways from June 1, 2007 to May 31, 2008. Air was pumped into the instruments from inlets. To assure data reliability, the analyzers were calibrated bi-weekly (Munger et al. 1998). Inevitably, there were minor gaps in data collection resulting from routine instrument calibrations.

Nitrogen oxides were measured by chemiluminescence and O_3 by ultraviolet photometric absorption. Hourly averaged data were derived from the 5-minute interval data. Based on the Chauvenet's criterion, the data point in each 1-hour range was treated as an outlier if it fell outside the 3 times standard deviation from the mean: 0.006%, 0.17% and 0.03%, for O_3, NO and NO_x, respectively. Consistent with EPA guidelines for NO_2 and O_3, 8-hour, daily and yearly averages were generated from the hourly averaged data. A valid day was defined as one with at least 75% of the possible 8-hour averages. Any day less than 75% complete was considered valid only if the daily maximum was greater than the standard (www.epa.gov).

4.2.3 Meteorological and Public Health Data

Meteorological data from the Lyndhurst site were supplemented with data from Weather Underground (www.underground.com). The primary meteorological variables were temperature, wind speed, wind direction and barometric pressure. The daily counts of both emergency department (ED) and hospital admission data were obtained from the New Jersey Department of Health and Senior Services (NJDHSS). Health records that contained International Classification of Diseases, Ninth Revision (ICD-9) clinical modification codes, were made available after approval by NJDHSS by the Institutional Review Board (IRB). Since the data included non-confidential hospital records, individual identifiers were not necessary and IRB approval was granted without consent from patients.

Respiratory conditions for both newly diagnosed and chronic cases were considered for specific zip codes in the Meadowlands with ICD-9 diagnosis codes for the following fourteen conditions: acute upper respiratory infections of multiple unspecified sites [465] acute bronchitis and bronchiolitis [466], chronic sinusitis [473], other diseases of upper respiratory tract [478], bronchitis (not specified as acute/chronic) [490], chronic bronchitis [491], emphysema [492], asthma [493], bronchiectasis [494], extrinsic allergic alveolitis [495], chronic airways obstruction, not specified elsewhere [496], respiratory conditions due to other and unspecified external agents [508], other respiratory diseases of the lung [518] and other diseases of respiratory system [519]. The number of ED visits and hospital admissions were determined by summing the daily counts of each health condition using only primary diagnoses.

4.2.4 Data Analyses

Data analysis was conducted with SAS 9.2 and MINITAB 16 software packages; SPSS 18.0 was used for intermediate data processing. Time-series analysis using stepwise multiple regression was applied to reduce potential confounding bias in the association between air pollution and seasonal fluctuations in health outcomes, day-of-week, weekend effects and meteorological factors. The dependent variable, total number of hospital admissions for all respiratory illnesses and the independent variables, NO_x and O_3, were linked by day to estimate possible relationships. This approach has been central to the regulatory policy process and has provided epidemiological evidence used to evaluate the risks of air pollution levels (Dominici et al. 2004; Ainslie and Steyn 2007).

The effect between exposure and response can be spread over several days; lag analyses were automated since admissions on a single day may be influenced by exposure to air pollution both on that day and preceding days. A lag of 0 days corresponds to the association between NO_x and O_3 levels on a given day and risk for hospital admission on that day. This was followed by tests of up to five lag days to allow better comparisons for greater insight into the relationship between the concentration of each pollutant and hospital admission (Steib 2009). It suggests adverse short-term health effects of O_3 and NO_x in relation to day-to-day variations at varying levels of significance based on time of day, ambient meteorology, etc. (Brook et al. 2004; Dominici et al. 2004).

4.3. Results and Discussion

4.3.1 Seasonal Variations

Low O_3 concentrations occurred from September to May during which NO_x displayed a single moderate peak. There was a broad O_3 peak from June to August. Conversely, NO_x concentrations were at their lowest during the summer months but did not display a strong seasonal pattern (Fig. 4.2), probably attributed to poor dependence on meteorological conditions (Khoder 2009; Smith et al. 2011). Similarly, NO and NO_2 consisted of a moderate winter maximum and summer minimum (Khoder 2009; Bigi and Harrison 2010; Smith et al. 2011). In a source-apportionment study conducted by Guttikunda and Gurjar (2011), pollutant concentrations including NO_x, were invariably 40% to 80% higher in winter. Smith et al. (2011) observed higher NO_2 levels in winter apparently when anthropogenic sources play a greater role in NO_x build-up than meteorological conditions. Enhanced seasonality may be partly attributed to increased fossil fuels for domestic heating and commuting. Traffic emissions (NO) are converted to NO_2 while the photochemical formation of O_3 is inhibited by the lack of intense solar radiation (Sadanaga et al. 2008; Geddes et al. 2009).

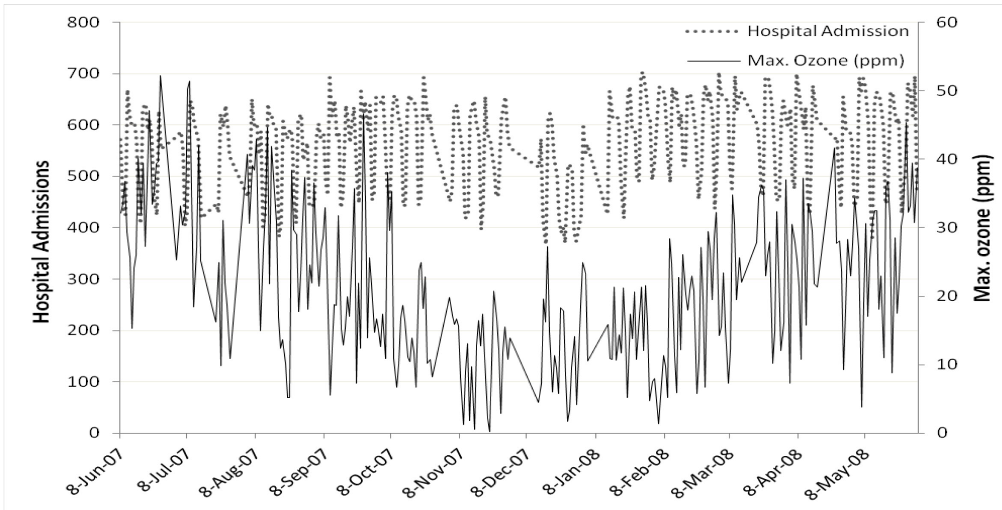


Figure 4.2 Temporal variations of the daily average O_3 and hospital admissions over the one-year period (June 1, 2007 to May 31, 2008). Ozone displays summer maximum and winter minimum with pronounced variations in the colder months with no apparent comparison to hospital admission trends.

The relationship between O_3 and NO_x is non-linear. O_3 molecules produced per molecule of NO_x consumed or production efficiency increases as NO_x decreases (Seinfeld and Pandis 2006; Yerramilli et al. 2011). Maximum photochemical generation and O_3 production in summer has made it difficult for local and regional regulatory agencies to meet a stringent 8-hr O_3 standard (Williams et al. 2009). Therefore, assessing uncertainty of future estimates related to photochemical processes and NO_x sources remains a challenge. Nevertheless, quantifying the magnitude and chemical evolution of NO_x is fundamental to our understanding of the health impact of anthropogenic emissions.

4.3.2. Seasonal Effect on Hospital Admissions

The greatest effect of air pollution on respiratory hospital admissions is evident in the winter season, consistent with previous investigations. Figure 4.3 (a) shows a steady increase in hospital admissions in winter, followed by a gradual decrease in spring (Fig. 4.3b) and fluctuations in summer (Fig. 4.3c). In the fall (Fig. 4.3d), hospital admissions showed an increasing trend followed by a slight decrease. The average respiratory hospital admissions were 564 (fall), 560 (winter), 584 (spring) and 549 (summer). When compared to summer, there were approximately 23, 18 and 31 more individuals hospitalized everyday in the fall, winter and spring, respectively. These results are consistent with previous studies utilizing multiple approaches to analyze both long- and short-term air pollution data. Qian et al. (2010) found statistically significant ($p<0.05$) cause-specific respiratory mortality in generalized additive models that estimated the seasonal effect of air pollution including NO_2, over a four-year period in China. They showed a clear seasonal pattern with the strongest effects in winter.

Kan et al. (2010) in natural-spline, single- and multi-pollutant models over a four-year time-series study, found more pronounced associations between air pollution, including NO_2 and O_3, in the cool season than in the warm season. Seasonal variations for health effects of NO, NO_2, O_3 and NO_x were observed in a multilevel logistic regression model for respiratory symptoms with no obvious differences in health when winter air pollution was generalized (Linares et al., 2010). In contrast, Jariwala et al. (2011) analyzed a one-year data set of daily NO_x, O_3, SO_2, and pollen counts at seven major Bronx hospitals in New York City and, similar to the present study, found increased health effects in winter, spring and fall. The spring peak also coincided with high tree pollen counts; however, Rossi et al. (1993) observed much higher ED visits in winter than in summer in Finland,

and found no association with airborne pollen levels. Instead, the most significant correlations were found between asthma visits and NO_2 levels over the one-year period.

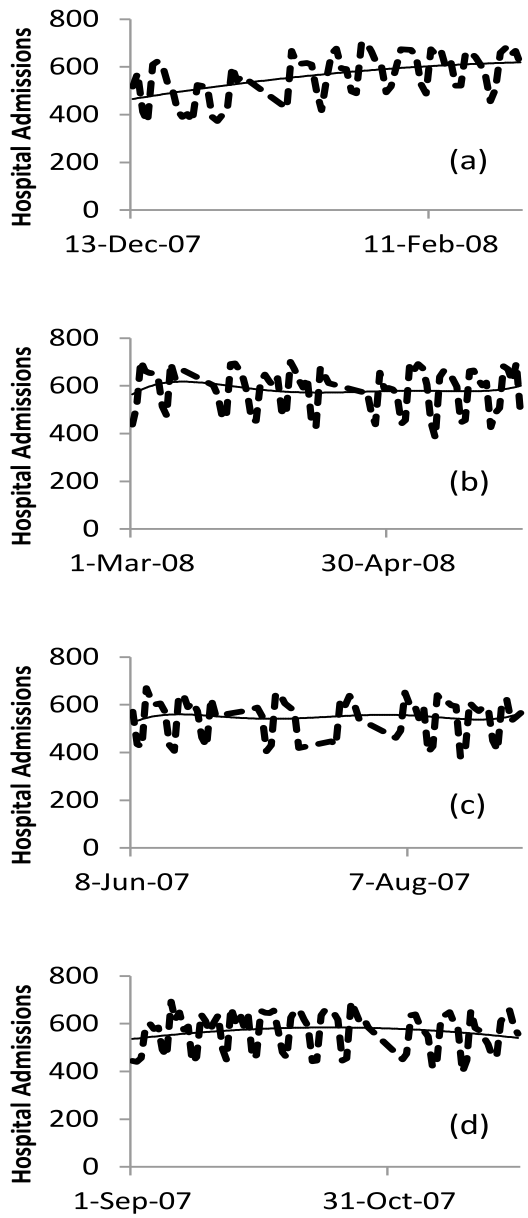


Figure 4.3 Time series of hospital admissions show (a) pronounced increase in the winter, (b) fluctuations in the spring and (c) summer seasons, and (d) an increase followed by a gradual decrease in the fall season.

In Leipzig, Germany, Mutius et al. (1995) found that NO_2 exacerbated respiratory health problems in winter when concentrations were at their highest. Daily mean NO_x concentrations showed a significant association with upper respiratory illnesses in winter, although concentrations only slightly exceeded those in summer. A previous study in Munich showed that NO_x might irritate the lung by altering the immune system. Human volunteers who inhaled weakened influenza virus after exposure to NO_x were more susceptible to infection than a control group that did not inhale NO_x (Mutius et al. 1995). Abe et al. (2009) measured daily air pollution levels including NO_x and meteorological data in Tokyo and observed that colder temperatures were related to an increased risk for significant exacerbation of asthma in adults necessitating ED visits. Furthermore, NO_x decreases the ability to generate antibodies when challenged by pathogens, reducing the ability of the respiratory system to remove foreign particles such as bacteria and viruses from the lung, particularly in the cold season (Abe et al. 2009). The inflammatory response to infections exacerbated by oxidants and aggravated symptoms can lead to an increase in respiratory-related hospital admissions (Fusco et al. 2001).

4.3.3. Air Pollution Concentrations and Daily Hospital Admissions

The gradual increase in hospital admissions during the winter and fall seasons might be associated with higher NO_2 concentrations. There was an increasing trend in the monthly mean NO_2 concentrations and a corresponding significant increase in hospital admissions for respiratory illnesses during the same period (Fig. 4.4a-b). Increased NO_2 concentrations can cause changes in lung function and susceptibility to lung-related illness in children and the elderly (Mutius et al. 1995; Abe et al., 2009). The present study did not examine the age-related effect of air pollutants; however, daily gaseous pollutant levels may be associated with hospital respiratory admissions at all ages, and may be more severe without the increased use of medication by asthmatic subjects, which effectively mask the adverse effects, especially of O_3 (WHO 2003).

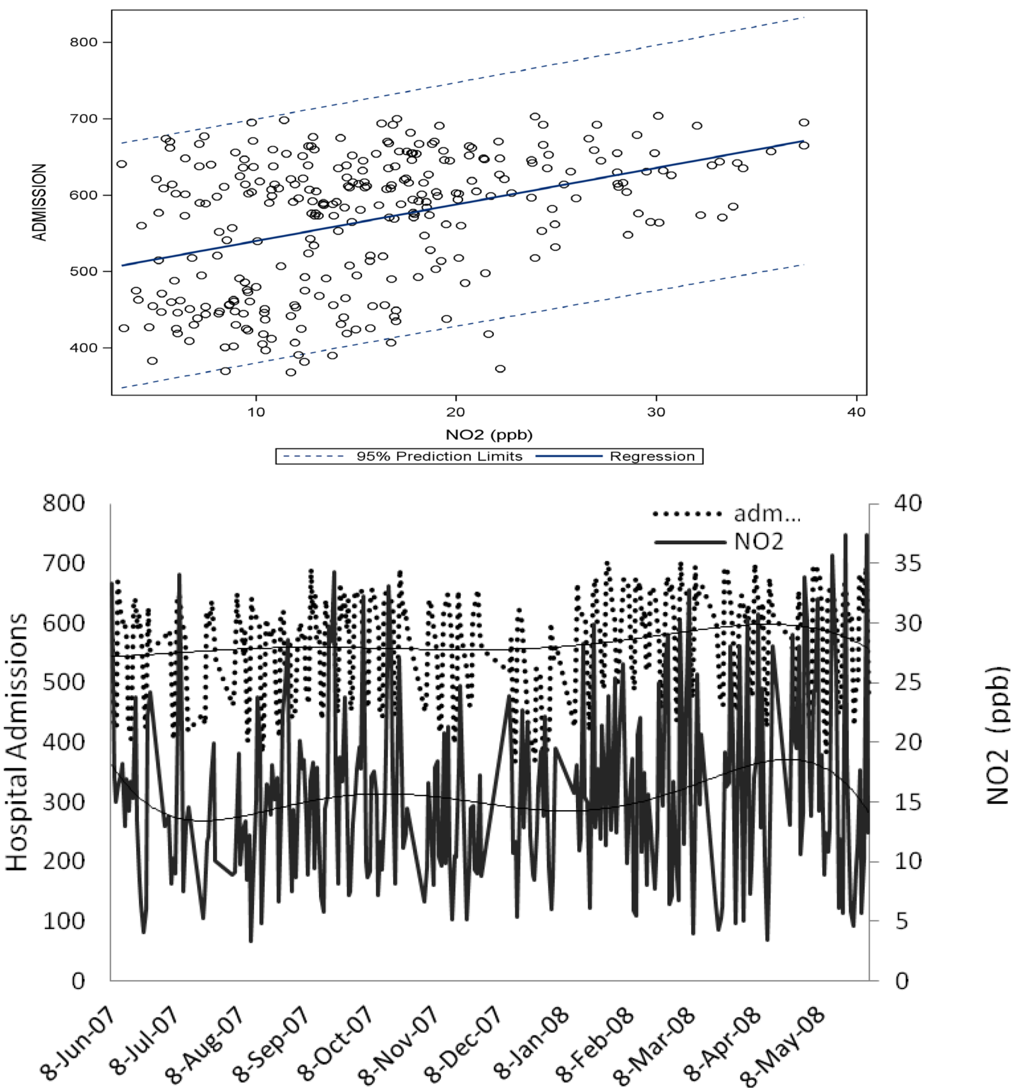


Figure 4.4 Respiratory hospital admissions show (a) a corresponding increase with monthly NO_2 concentrations (b) from late fall through early spring season.   .

The association between air pollutant concentrations and health effects may occur at different time lags that spread over several days, depending on response and exposure (Steib 2009). The number of admissions on a single day may be influenced by exposure to air pollution on the same day or up to four days later, further explained under Section 2.3.4. In this study, one unit increase in O_3 at 2 lag days was associated with an increase in hospital admission by almost 0.7 units, and 0.56 for NO_x, if other variables were kept constant. Ozone and NO_x are known to induce lung function impairment and airways inflammation during episodes of elevated concentration (WHO 2003; Grineski et al. 2010; Kelly et al. 2011).

Weisel et al. (1995) compared the mean O_3 on one and two previous days: lag 24 and lag 48 (hours), respectively using a stepwise multiple regression analysis and found no significant relationship between O_3 exposure and ED visits in time lags during the five year period. Similar findings were observed in Rome on the event day (Forastiere et al. 2005). Chan et al. (2009) found no statistical significance in the effect of air pollutants, including O_3 on ED visits, up to two lag days. Conversely, in Canada O_3 was most consistently associated with respiratory hospital ED visits up to two lag days (Steib et al. 2009). In addition, estimates from the National Morbidity, Mortality and Air Pollution Study (NMMAPS) reported 0.41% increase in mortality associated with an increase of 10 ppb (20µg/m^3) in daily O_3 concentrations on the same day in the summer. A larger effect was found at lag 2, independent of other pollutants.

4.3.4. Weekend Effect on Hospital Admissions

Consistent with previous studies including the Meadowlands, the average NO_x concentrations were lower on weekends than on weekdays (Song et al. 2011); the opposite was true for O_3 concentrations (Fig. 4.5). This weekend cycle indicates a possible negative influence of NO_x on O_3 due to elevated VOC concentrations, a phenomenon commonly known as the "weekend effect" (Paschalidou and Kassomenos 2004; Khoder 2009; Song et al. 2011). Changes in the timing and location of emissions and meteorological factors play smaller roles in weekend O_3 behavior but temperature differences might modify rather than explain the weekend effect (Munger et al. 1998).

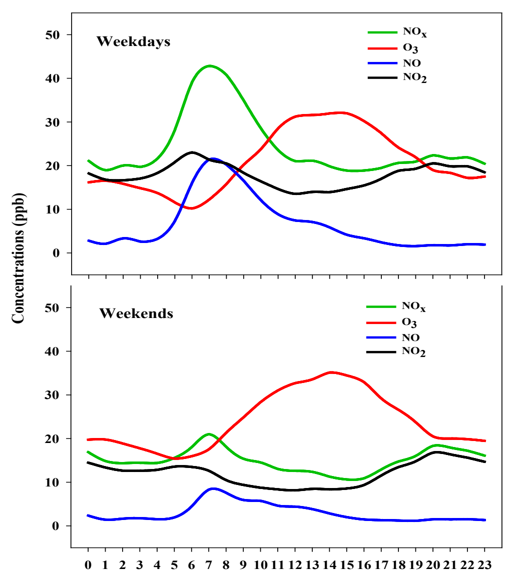


Figure 4.5 Diurnal variations of NO_x and O_3 on weekdays and weekends. The O_3 peak is similar for both weekdays and weekends whereas NO, NO_2 and NO_x display much smaller morning peaks on weekends than on weekdays.

The weekend cycle, however, did not correlate with hospital admissions as a response to higher O_3 concentrations in the study area; 166 more people were admitted to the hospital on weekdays than on weekends. This difference might not be attributed to the relative proportions of air pollutant concentrations per se but may be explained by varying hospital staffing levels, usually lower on weekends than on weekdays, despite a presumably consistent day-to-day burden of respiratory diseases (Bell and Redelmier 2001; Arabi et al. 2006). Based on staffing levels, it is also uncertain whether hospital admissions differ according to severity of a respiratory illness or whether personal factors affect the way respiratory patients are admitted on weekends versus weekdays such as the choice to remain at home on weekends.

4.3.5. Meteorological Factors and Hospital Admissions

Concentrations of NO_x and O_3 vary temporally and spatially with local meteorological parameters (Abe et al. 2009; Khoder, 2009). In health effect models, Ito et al. (2007) analyzed spatial variations, temporal relationships, extent and nature of multi-collinearity among air pollutants including O_3 and NO_2 at multiple monitors in New York City from 1999-2002. They concluded that the health effects regression on pollutants would provide better results if analyzed by season with the most meaningful results for asthma cases (Ito et al. 2007). This limitation was corrected in the present study to link meteorological and seasonal variations in air pollution concentrations to health effects.

The results showed that one unit increase in relative humidity and barometric pressure was associated with a decrease in hospital admissions by 0.52 and 32, respectively; however, these variables indirectly influenced hospital admissions primarily through the effect of wind speed. The coefficient corresponding to barometric pressure was very high in numerical value due to its small range (29.41 - 30.66), making one unit change very sensitive to hospital admissions. Low temperature (discussed in section 4.1. above) and wind speed, appeared to have the greatest effect on air pollutant concentrations and consequently, respiratory hospital admissions. Temperature differences partly explain the opposing trends in NO_x and O_3 concentrations, either amplified or diminished by a summer maximum and winter minimum. Obviously, the lowest temperatures occurred from December to February (Fig. 4.6a), when NO_x concentrations were highest and O_3 concentrations were lowest.

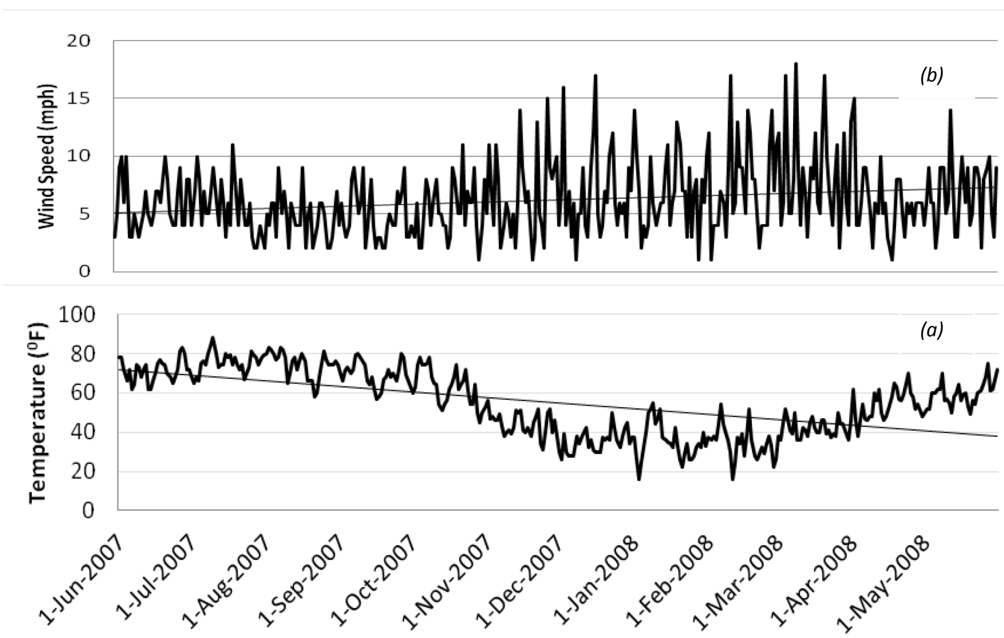


Figure 4.6 Meteorological parameters: temperature (a), show a decreasing trend and wind speed (b), show an increasing trend and from June 1, 2007 to May 31, 2008.

In a time series analysis and multivariable-adjusted autoregressive integrated moving average model, Abe et al. (2009) compared daily ED visits for asthma with air pollution levels including NO_x and meteorological data. They found that minimum temperatures were significantly associated with air pollution increase and exacerbation of asthma, requiring emergency care (Abe et al. 2009).

The lowest wind speeds were observed in late July to September when O_3 concentrations peaked (Fig. 4.6b). Conversely, the highest wind speeds occurred when O_3 concentrations declined to a minimum, generally from October to May. Wind speed explained more than 43% variability of NO_2 concentrations. The prevailing southwesterly winds were most dominant from December through February, corresponding to the highest peak in NO_x concentrations and hospital admissions. With one unit increase in wind speed, two additional individuals were admitted to the hospital for respiratory illnesses. When the lag 2 value of O_3 was proportionate to hospital admissions; that is, when O_3 increased by one unit, hospital admissions also increased by one unit.

Kim et al. (2004) conducted a school-based cross-sectional study in the San Francisco Bay Area in a two-stage multiple logistic regression model. Associations were found between respiratory symptoms and exposure to relatively low traffic-related air pollutants including NO_x and NO_2. The downwind direction was an important determinant of increased exposure to traffic pollutants (Kim et al. 2004). Ainslie and Steyn (2005) made observations of higher downwind O_3 concentrations in a multiple linear regression model.

In the present study, this was particularly true for periods with prevailing southwesterly winds from east coast urban areas, which presumably enabled the transport of precursor pollutants over hundreds of miles downwind from their original emission sources, and later transformed them into O_3. These trans-boundary characteristics make the study of O_3 a more complex problem as well as a challenge to attain in-state controls and federal O_3 standard (Farrell et al. 1999, Sillman 1999).

4.3.6. Statistical Analysis

Associations among meteorological parameters, O_3, NO_x and respiratory hospital admissions were analyzed in a three-stage modeling strategy. A multiple linear regression equation was fitted by entering or removing the "candidate variables" in a stepwise manner to arrive at a set of variables with the greatest strength of prediction. The best possible subset of predictors that explain variability in hospital admission was selected in sequence and fitted in both forward and backward stepwise regression. This process was finalized by the goodness of fit of the regression model, established by several diagnostic analyses of residuals and partial correlation graphs that confirm multi-collinearity. Similar methods were used in previous air pollution and health related studies (Weisel et al. 1995). The regression model was given as

$$y = \beta_0 + \beta_i x_i + \beta_2 x_2 + \beta_3 x_3 + \ldots + \beta_p x_p + \varepsilon$$

where y is the dependent variable (respiratory hospital admissions), xi is the ith independent variable, βi is the regression coefficient, β_0 is the intercept, p is the number of independent variables, and ε is the error term with mean zero. The final model (R^2) is:

Hospital admissions = 1310.926 + 165.798 weekday - 0.456 relative humidity - 29.758 barometric pressure + 0.742 O_3 lag 2 + 0.538 NO_x + 30.607 fall + 28.612 winter + 32.089 spring + 0.459 NO_x lag 5

After deleting all missing values, the first model selected explained 69% of the total variability. In the diagnostic test, all residuals were along a straight line but the histogram did not display complete symmetry due to a few outliers; however, the normal distribution of residuals fit well and confirmed the assumption of constant variance and independence. The outlier detection identified 5 leverages and 8 large standardized residuals.

Table 4.1 Model summary of variables associated with hospital admissions

Variable	Model 1	Model 2	Model 3
Hospital Admission	1441	1417.5	1311
Weekday	156	165.2	166
Relative humidity	- 0.73	- 0.52	- 0.46
Barometric pressure	- 33.0	- 32.18	- 29.76
O_3 lag 2	-	-	0.74
NO_x	1.26	0.57	0.54
NO_x lag 4	0.43	-	-
Fall	-	22.97	30.61
Winter	-	18.48	28.61
Spring	-	31.37	32.09
NO_x lag 5	-	-	0.46
R^2 [1]	69.0%	80% [2]	81% [3]

[1]Variability of hospital admissions explained
[2]344 data pts without lag values
[3]295 data pts with lag values

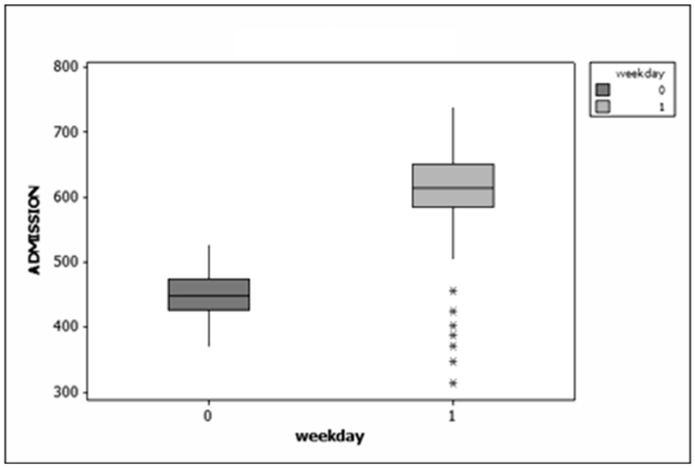


Figure 4.7 Boxplots of respiratory hospital admissions on weekday and weekends. (1) = weekday or the day is neither Saturday nor Sunday, (0) = weekend or the day is either Saturday or Sunday.

When compared to other weekdays, there were 4 unusually low (Fig. 4.7) and a few missing values of O_3, NO_x, NO and NO_2, that occurred on holidays (Fig. 4.8). These were combined with the weekend variable; however, 14 unusually high values among the leverage values of O_3, NO_x, NO and NO_2; presumably caused by calibration problems, were removed from the analysis to avoid bias. In the second model, we used 344 data points (Table 4.1) excluding the original 7 missing values and 15 outliers detected in the first model.

There was no problem of multi-collinearity and the p-value was less than .0001, confirming a good fit (Figs. 4.9; 4.10). Additional variables: weekday/weekend factor, 5 lag days for O_3, 5 lag days for NO_x, and 4 seasons were included to estimate their influence on hospital admissions.

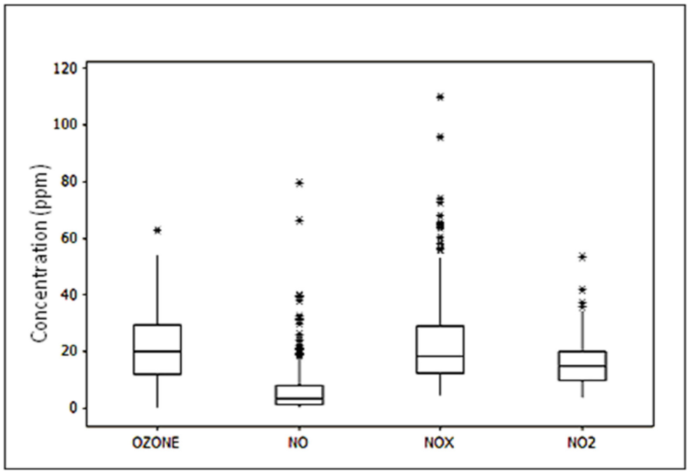


Figure 4.8 Boxplots of O_3, NO, NO_x and NO_2 depicting the distribution of each variable over the study period.

Weekday explained more than 77% of variability in hospital admissions, followed by spring with 1% and the remaining 7 factors altogether explained approximately 80% variability in hospital admissions (R^2 = 0.804). Finally, air pollutants with lags, were added with weather variables in the third model.

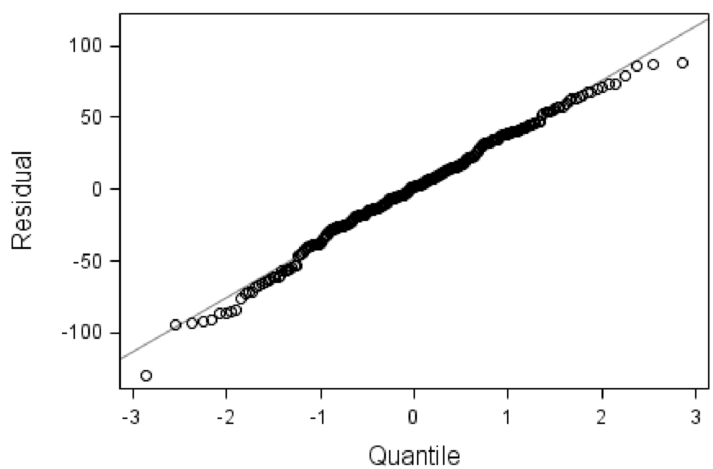


Figure 4.9 Residual diagnostic Q-Q plot for hospital admissions. All residuals are along a straight line.

Time lags (0 to 5 days) between daily pollution concentration and daily respiratory admissions were tested. Unusually high values corresponding to lag values of NO_x and O_3 were ignored, leaving 295 data points. Altogether, NO_x lag 5, fall, winter, O_3 lag 2, barometric pressure, NO_x and relative humidity explained more than 80% of total variability in hospital admissions (Table 4.1).

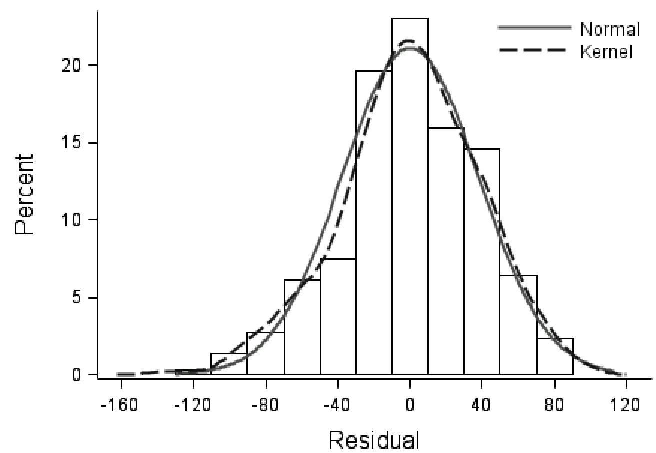


Figure 4.10 Residual diagnostic distribution of hospital admissions. The symmetrical histogram plot of residuals confirms the normality assumption.

The path model (Fig. 4.11) illustrates the relative dependence of respiratory hospital admissions on the key air pollutants and meteorological variables. Since O_3 had the least overall direct impact on respiratory hospital admissions, it was eliminated from the final model.

Among the pollutants, NO_2 showed the strongest association with hospital admissions. Of the meteorological factors, wind speed had the strongest effect on hospital admissions but this association was strongly influenced by wind direction as well.

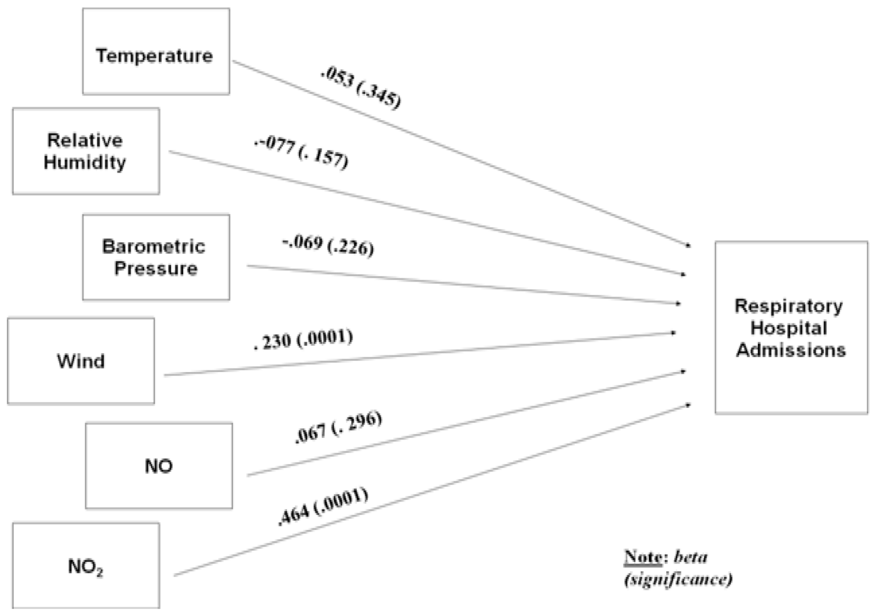


Figure 4.11 Path analysis of predictors of respiratory hospital admissions. Model R = .421, F =12.623, p =.0001. The most significant pollutant predicting respiratory hospital admissions was NO_2 and the meteorological parameter with the greatest effect was wind speed.

4.4. Conclusions

The statistical associations between levels of individual or combined gaseous air pollutants and hospital admissions are attributable to respiratory diseases. We agree with previous studies, that though useful, a complete analysis of the effect of atmospheric changes on air pollutants and hospital admissions for respiratory symptoms is a challenge. This is mainly due to the multiple variables related to health outcomes which no single or combined model can completely take into account. With consideration of some multifactorial effects linked to air pollution exposure, however, meaningful conclusions about the health impacts can be drawn. Although the present study could not completely corroborate the significant associations between outdoor NO_x and O_3 concentrations with adverse health outcomes, it presents evidence that is adequately consistent, coherent, and plausible. It demonstrates that at ambient levels, NO_x and O_3 may be enhanced by certain meteorological conditions with resultant health effects.

Personal exposure to respiratory health problems show association for O_3 concentrations in the summer but there is a stronger association for NO_x in the fall, winter and spring seasons. This finding is consistent with previous studies; however, despite the O_3 production efficiency which limits precursor pollutants, NO_2 seemed to have a greater overall effect on respiratory hospital admissions, a valuable contribution to current air pollution literature. Variations in O_3 occurred by season and specific time of day, with the highest concentrations in the summer and early afternoon, whereas traffic-related pollutants, NO and NO_2, spiked mainly during rush hour and in cold weather conditions. Future studies should incorporate pollen and other health effects data to further confirm these findings.

Chapter 5: Conclusions

5.1. Overall conclusions and contributions

First, this study demonstrated that ambient levels of PM and O_3 may be enhanced by certain meteorological conditions with resultant health effects. The positive relationship between temperature and PM, and the statistical significance of log$PM_{2.5}$, r=0.22 (p=0.04) and logPM_{10} concentrations, r=0.5, (p=0.0001) suggest that there may be a higher health risk with exposure to PM due to its temperature-dependence. This finding supports the proposal for improved air quality standards. In addition, more than one averaging period may be established for pollutants based on short-term variations, particularly for O_3. This would address the challenge faced by local and regional regulatory agencies to meet the current 8-hr O_3 standard and for states that appear to be in compliance because time periods during which people are at risk are masked by periods when O_3 production is negligible. By limiting the averaging period(s) to times when maximum pollutant concentrations are observed, health effects can be reduced through more stringent monitoring, assessment and warning systems.

Second, in both time series and statistical models, there was evidence of an association between respiratory hospital admissions and air pollutant levels, whether in particle or gas-phase. The positive and statistically significant relationship between hospital admissions and both logPM_{10} and $NO_{2,}$ r=0.5, (p=0.0001) may have future policy implications. Apparently, PM and O_3 concentrations were mainly dependent on meteorological factors whereas anthropogenic sources seemed to play a greater role in the elevation of NO_x concentrations. Although the production efficiency of O_3 is known to impose limits on its precursor pollutants (NO_x), a significant relationship was evident between hospital admissions and NO_2. This finding is a valuable contribution to current air quality literature since the impact of NO_2 may intensify as cities continue to expand and an over-dependence on motor vehicular transport becomes likely. In addition, the stronger association with NO_2 and respiratory health problems in the cold season is consistent with previous studies and is attributable to increased fossil fuel consumption.

Third, this study addressed the limitation acknowledged in national and regional studies that employ large-scale data collection methods from central monitoring stations, where spatial variability due to local meteorology and location has been overlooked. As mentioned in chapter three, previous studies have indicated that air pollution may vary within a single city as much as across different cities. At a single sample site, observations represent the unique qualities of both the local area and each sampling event. This is particularly true for PM measurement which is labor intensive and data from multiple sites may be cumbersome, presenting additional complexities. Although the health effects may be more distinguishable when the combined effects of pollutants are considered at a common location, however, it would be beneficial to analyze air pollution effects from multiple sites in future to compare the findings of the present study.

Fourth, the strong O_3 seasonal pattern may not always be indicative of a greater health risk on the same day of exposure, particularly in summer. In Newark, the most significant respiratory health effect of O_3 occurred at 1 lag day. With an increase of 0.01ppm in $O_{3,}$ there was a corresponding increase in hospital admissions by 2.16% (95% CI: 0.72, 3.59). Similar observations were made in the Meadowlands but a stronger association was observed at 2 lag days, where hospital admissions increased by 0.7 (95% CI: 0.25, 1.23) with one unit increase in O_3. These findings confirm previous research that employed several statistical methods to analyze these relationships in similar urban settings. On the other hand, consistent with the conclusion drawn in previous studies, due to the number of variables related to health outcomes, no single or combined model could completely take all factors that influence air quality and respiratory hospital admissions into account. The findings in this study may contribute to a better understanding of the basic mechanisms that are linked to air pollutant concentrations and their short-term health effects.

Finally, this research emphasized that a study of the linkages between meteorological conditions and air pollution on a local scale play a vital role in communicating complex physical characteristics of atmospheric modeling in a simplistic manner. These relationships must be understood to further elucidate pollution parameters and potential health risks to maximize the effectiveness of anti-pollution initiatives. The analysis of spatio-temporal effects in the study area may allow the design of a future health and weather risk-warning system. It has practical value and may be useful in complementing epidemiological studies that show statistical associations between air pollutants and hospital admissions for respiratory problems without establishing a cause-effect relationship.

5.2. Recommendations for future research

To expand this research, additional PM data analysis should be conducted to further investigate the seasonal variations and chemical composition of ambient PM in Newark. The present work was limited by a pilot study, although a subsequent one-year PM data set was collected at this site. Further analysis will provide stronger evidence of the association between meteorological parameters and public health.

Pollen data should be obtained to assess the chemical composition and toxicity of ambient air pollutants among high-risk populations through methods that adequately adjust for confounding factors. Such methods should distinguish concurrent respiratory responses to air pollutants from other outdoor exposures including fungal spores and additional meteorological factors. These important considerations will explain the etiological relationship between factors influencing ambient air pollutants and public health that remain unclear to the respiratory physician. In addition, further study of the complex interactive effects of weather, air pollution and other outdoor agents would provide greater insight into the relative contribution of each factor to hospital admissions.

Additional scientific investigations of this nature can establish improved standards for sulfates, the most dominant chemical component in ambient PM in New Jersey and other areas of the US North East Coast. These investigations will support an improvement of the National Ambient Air Quality Standards that seek to protect public health. Future studies of particle and gas-phase pollutants both in Newark and in the Meadowlands can inform policy makers regarding the need for more practical regulations that effectively reduce the risk to air pollution exposure throughout urban areas in New Jersey.

2.5 References

Abe, T., Tokuda, Y., Ohde, S., Ishimatsu, S., Nakamura, T., Birrer, R.B., 2009. The relationship of short-term air pollution and weather to ED visits for asthma in Japan. *Am. J. Emerg. Med*. 2,153-9.

Abdul-Wahab, S.A., Bakheit, C.S., Al-Alawi, S.M., 2005. Principal component and multiple regression analysis in modeling of ground-level ozone and factors affecting its concentrations; *Environmental Modelling & Software* 20, 1263-1271.

Ainslie, B., Steyn, D.G., 2007. Spatiotemporal trends in episodic ozone pollution in the lower Fraser Valley, British Columbia, in relation to mesoscale atmospheric circulation patterns and emissions. *American Meteorological Society* 46, 1631-1644.

Alam, K., Blaschke, T., Madl, P., Mukhtar, A., Hussain, M., Trautmann, T. and Rahman, S., 2011 'Aerosol size distribution and mass concentration measurements in various cities of Pakistan', *Journal of Environmental Monitoring*, Vol. 13, No. 7, pp.1944–1952.

Alexis, N., Barnes, C., Bernstein L., Bernstein, J.A., Nel A., Peden, D., et al., 2004. Health effects of air pollution. *Journal of Allergy and Clinical Immunology* 114, 1116-23.

Arabi, Y., Alshimemeri, A.**,** Taher, S., 2006. Weekend and weeknight admissions have the same outcome of weekday admissions to an intensive care unit with onsite intensivist coverage**.** *Critical Care Medicine* 34(3), 605-611.

Bell, C.M., Redelmeier, D.A., 2001. Mortality among patients admitted to hospitals on weekends as compared with weekdays. *N. Engl. J. Med.* 345, 663-668.

Bell, M.L., 2006. 'The use of ambient air quality modeling to estimate individual and population exposure for human health research: a case study of ozone in the Northern Georgia Region of the United States', *Environment International*, Vol. 32, N0.5,pp. 586–593.

Bell, M.L., Ebisu, K., Peng, R.D., Samet, J.M. and Dominici, F., 2009. 'Hospital admissions and chemical composition of fine particle air pollution', *American Journal of Respiratory and Critical Care Medicine*, Vol. 179, No. 12, pp.1115–1120.

Bernstein, J.A., Alexis, N., Barnes, C., Bernstein, I.L., Nel, A., Peden, D., Diaz-Sanchez, D., Tarlo, S.M., Williams, P.B., 2004. Health effects of air pollution. *Journal of Allergy and Clinical Immunology* 114, 1116-1123.

Berti G., Chiusolo M., Grechi D., Grosa, M., Rognoni, M., Tessari R., Pacelli B., Scarnato C., Mallone, S., Vigotti, M.A., Stafoggia, M., Primerano, R., Accetta, G., Dessì, M.P., Cernigliaro, A., De'Donato, F., Zanini, G., Forastiere, F., 2009. Environmental indicators in ten Italian cities (2001-2005): the air quality data for epidemiological surveillance]. *Epidemiology Prev*. 33(6 Suppl 1):13-26.

Bigi, A., Harrison, R.M., 2010. Analysis of the air pollution climate at a central urban background site, *Atmospheric Environment* 44, 2004-2012.

Blanchard, C.L., Tanenbaum, S.J., 2003. Differences between weekday and weekend air pollutant levels in southern California, *Journal of Air and Waste Management Association* 53(7), 816-28.

Blanchard, C.L., Tanenbaum, S.J., 2006. Weekday/Weekend differences in ambient air pollutant concentrations in Atlanta and the southeastern United States. *Journal of Air and Waste Management Association* 56, 271-84.

Brauer, M., Ainslie, B., Buzzelli, M., Henderson, S., Larson, T., Marshall, J. D., Nethery, E., Steyn, D., Su, J. (2008) 'Models of exposure for use in epidemiological studies of air pollution health impacts. In air pollution modeling and its application', *XIX (NATO Science for Peace and Security Series C: Environmental Security*, Borrego, C., Miranda, A.I. (Eds) Springer: Dordrecht, The Netherlands, pp.589–604.

Brook, R.D., Franklin, B., Cascio, W., Hong, Y., Howard, G., Lipsett, M., et al., 2004. Air pollution and cardiovascular disease: A statement for healthcare professionals from the expert panel on population and prevention science of the American Heart Association. *Journal of the American Heart Association* 109, 2655-2671.

Brown, S.S., Ryerson, T.B., Wollny, A.G., Brock, C.A., Peltier, R., Sullivan, A.P., Weber, R.J., Dube´, W.B., Trainer, M. Meagher, J.F., Fehsenfeld, F.C., Ravishankara, A.R., 2006. Variability in nocturnal nitrogen oxide processing and its role in regional air quality, *Science* 311, 67-70.

Brunekreef, B., Beelen, R., Hoek, G., Schouten, L., Bausch-Goldbohm, S., Fischer, P., Armstrong, B., Hughes, E., Jerrett, M. and van den Brandt, P. (2009) 'Effects of long-term exposure to traffic-related air pollution on respiratory and cardiovascular mortality in the Netherlands: the NLCS-AIR study', *Res. Rep. Health Eff. Inst.*, No. 139, pp.5–71.

Butler, T.J., Likens, G.E., Vermeylen, F.M. and Stunder, B.J.B., 2005. The impact of changing nitrogen oxide emissions on wet and dry deposition in the northeastern USA. *Atmospheric Environment* 39, 4851-4862.

Chan, T., Chen, M., Lin, I., Lee, C., Chiang, P., Wang, D. et al., 2009. Spatiotemporal analysis of air pollution and asthma patient visits in Taipei, Taiwan', *International Journal of Health Geographics* 8, 26.

Clapp, L.J., Jenkin, M.E., 2001. Analysis of relationship between ambient levels of O_3, NO_2 and NO as a function of NO_x in the UK. *Atmospheric Environment* 35, 6391-6405.

Dominici, F., McDermott, A., Hastie, T.J., 2004. Improved semi-parametric time series models of air pollution and mortality. *Journal of the American Statistical Association* 99(468), 938-948.

Dominici, F., Peng, R., Bell, M., Pham, L., McDermott, A., Zeger, S.L. et al., 2006. 'Fine particulate air pollution and hospital admission for cardio-vascular and respiratory diseases', *J. American Medical Association*, Vol. 295, No. 10, pp.1127–1134.

Ebelt, S., Brauer, M., Cyrys, J., Tuch, T., Kreyling, W.G., Wichmann, H.E. et al. 2001. Air quality in postunification Erfurt, East Germany: associating changes in pollutant concentrations with changes in emissions, *Environmental Health Perspectives*, Vol. 109, No. 4, pp.325–333.

Engler, C., Birmili, W., Spindler, G. and Wiedensohler, A., 2011. Analysis of exceedances in the daily PM_{10} mass concentration (50 μg m^{-3}) at a roadside station in Leipzig, Germany', *Atmospheric Chemistry and Physics*, Vol. 11, No. 5, pp.15831–15873.

Farrell, A., Carter, R., Raufer, R., 1999. The NOx Budget: market-based control of tropospheric ozone in the northeastern United States. *Resources and Energy Economics* 21, 103-21.

Forastiere, F., Stafoggia, M., Picciotto, S., Bellander, T., D'Ippoliti, D., Lanki, T. et al., 2005. 'A case-crossover analysis of out-of-hospital coronary deaths and air pollution in Rome, Italy', *American Journal of Respiratory and Critical Care Medicine*, Vol. 172, No. 12, pp.1549–1555.

Frumkin, H., 2002. Urban sprawl and public health', *Public Health Reports*, Vol. 117, 201–217.

Fusco, D., Forastiere, F., Michelozzi, P., Spadea, T., Ostro, B., Arca`, M., Perucci, C.A., 2001. Air pollution and hospital admissions for respiratory conditions in Rome, Italy, *European Respiratory Journal* 17, 1143–1150.

Geddes, J. A., Murphy, J.G., Wang, D.K., 2009. Long term changes in nitrogen oxides and volatile organic compounds in Toronto and the challenges facing local ozone control, *Atmospheric Environment* 43, 3407-3415.

Giles, L.V., Barn, P., Künzli, N., Romieu, I., Mittleman, M.A., van Eeden, S., Allen, R., Carlsten, C., Steib, D., Noonan, C., Smargiassi, A., Kaufman, J.D., Hajat, S., Kosatsky, T., Brauer, M., 2011. From good intentions to proven interventions: effectiveness of actions to reduce the health impacts of air pollution, *Environmental Health Perspectives* 119(1): 29-36

Godish, T., 2004. Air Quality 4th Ed. Lewis Publishers, New York.

Grineski, S.E., Staniswalis, J.G., Peng, Y., Atkinson-Palombo, C., 2010. Children's asthma hospitalizations and relative risk due to nitrogen dioxide (NO_2): Effect modification by race, ethnicity and insurance status. *Environ. Res.* 110(2), 178. doi:10.1016/j.envres.2009.10.012.

Gurjar, B.H., Nagpure, A.S., Singh, T.P. and Hanson, H. (Eds.) (2008) 'Air quality in megacities', *Encyclopedia of Earth, Environmental Information Coalition*, National Council for Science and the Environment, Washington, DC.

Gurjar, R., Butler, T.M., Lawrence, M.G., Lelieveld, J., 2008. Evaluation of emissions and air quality in megacities. *Atmospheric Environment* 42, 1593-1606.

Guttikunda, S.K., Gurjar, B.R., 2011. Role of meteorology in seasonality of air pollution in megacity Delhi, India. *Environ Monit Assess.* DOI 10.1007/s10661-011-2182-8.

Gvozdić, V., Brana, J., Malatesti, N., Puntarić, D., Vidosavljević, D., Roland, D., 2011. An analysis of the pollution problem in Slavonski Brod (eastern Croatia). *Coll Antropol*. 35, 1135-41.

Guerra, S.A., Lane, D.D., Marotz, G.A., Carter, R.E., Hohl, C.M. and Baldauf, R.W., 2006. 'Effects of wind direction on coarse and fine particulate matter concentrations in Southeast Kansas', *J. Air and Waste Management Association*, Vol. 56, No. 11, pp.1525–1531.

Hatzianastassiou, N, Katsoulis, B.D., Antakis, B., 2007. Extreme nitrogen oxide and ozone concentrations in Athens atmosphere in relation to meteorological conditions. *Environmental Monitoring Assessment* 1-3, 447-64.

Heuss, J.M., Kahlbaum, D.F., Wolff, G.T., 2003. Weekday/weekend ozone differences: What can we learn from them? *Journal of Air Waste Management Association* 53, 772-88.

Holman, C., 1999. 'Sources of air pollution', in Holgate, S., Samet, J., Koren, H. and Maynard, R. (Eds.): *Air Pollution and Health*, pp.115-148, Academic Press, London.

Hu, S., Fruin, S., Kozawa, K., Mara, S., Winer, A.M. and Paulson, S.E. (2009) 'Aircraft emission impacts in a neighborhood adjacent to a general aviation airport in Southern California', *Environmental Science and Technology*, Vol. 43, No. 21, pp.8039–8045.

Hussein, T., Karppinen, A., Kukkonen, J., Harkonen, J., Aalto, P.P., Hameri, K., et al., 2006. Meteorological dependence of size-fractionated number concentrations of urban aerosol particles. *Atmospheric Environment* 40, 1427-1440.

Ito, K., Thurston, G.D., Silverman, R.A., 2007. Characterization of PM2.5, gaseous pollutants, and meteorological interactions in the context of time-series health effects models. *J. Expo. Sci. Environ. Epidemiol.* 17 Suppl. 2, S45-60.

Jameson, P.F., Kalkstein, L.S. and Gergen, P.J. (1997) 'A synoptic evaluation of asthma hospital admissions in New York City', *American Journal of Respiratory and Critical Care Medicine*, Vol. 156, pp.1781–1788.

Jariwala, S.P., Kurada, S., Moday, H., Thanjan, A., Bastone, L., Khananashvili, M., Fodeman, J., Hudes, G., Rosenstreich D., 2011. Association between tree pollen counts and asthma ED visits in a high-density urban center. *J. Asthma* 48(5), 442-8.

Ji, M., Cohan, D.S., Bell, M.L., 2011. Meta-analysis of the association between short-term exposure to ambient ozone and respiratory hospital admissions. *Environ. Res. Lett.* 6, 11 pp, doi:10.1088/1748-9326/6/2/024006.

Kan, H., Chen, B., Zhao, N., London SJ, Song G, Chen G, et al. 2010. Part 1. A time-series study of ambient air pollution and daily mortality in Shanghai, China. *Res. Rep. Health Eff. Inst.* 154, 17-78.

Kelly, F., Anderson, H.R., Armstrong, B., Atkinson, R., Barratt B., Beevers, S., Derwent, D., Green, D., Mudway, I., Wilkinson, P., 2011. The impact of the congestion charging scheme on air quality in London. Part 1. Emissions modeling and analysis of air pollution measurements. *Resp Rep Health Eff Inst*. 155, 5-71.

Khoder, M.I., 2009. Diurnal, seasonal and weekdays-weekends variations of ground level ozone concentrations in an urban area in greater Cairo, *Environmental Monitoring Assessment* 149, 349-62.

Kim, J.J., Smorodinsky, S., Lipsett, M., Singer, B.C., Hodgson, A.T., Ostro. B., 2004. Traffic-related Air Pollution near Busy Roads, *American Journal of Respiratory and Critical Care Medicine* 170, 520-526.

Lai, L.W. and Cheng, W.L. (2009) 'Air quality influenced by urban heat island coupled with synoptic weather patterns', *Science of the Total Environment*, Vol. 407, No. 8, pp.2724–2733.

Lee, J.T., Kim, H., Song, H., Hong, Y.C., Cho, Y.S., Shin, S.Y. et al. (2002) 'Air pollution and asthma among children in Seoul, Korea', *Epidemiology*, Vol. 13, No. 4, pp.481– 484.

Linares, B., Guizar, J.M., Amador, N., Garcia, A, Miranda, V., Perez, J.R., Chapela, R., 2010. Impact of air pollution on pulmonary function and respiratory symptoms in children. Longitudinal repeated-measures study, *BMC Pulm Med.* 24, 10:62.

Liu, R.X., Kuang, J., Gong, Q., Hou, X.L., 2003. Principal component regression analysis with SPSS. Computer Methods and Programs in Biomedicine 71, 141-147

López-Villarrubia, E., Ballester, F., Iñiguez, C. and Peral, N., 2010. Air pollution and mortality in the Canary Islands: a time-series analysis, Environ *Health* 9:8.

Madsen, T., Mottola, D., 2003. The public health impact of air pollution in New Jersey. *The New Jersey Public Interest Research Group Law and Policy*, Trenton, N.J. 1-38.

McConnell, R., Islam, T., Shankardass, K., Jerrett, M., Lurmann, F., Gilliland, F., Gauderman, J., Avol, E., Künzli, N., Yao, L., Peters, J., Berhane, K., 2010. Childhood incident asthma and traffic-related air pollution at home and school. *Environ Health Perspectives* 118(7), 1021-6.

Munger, J.W., Fan, S.M., Bakwin, P.S., Goulden, M.L., Goldstein, A.H., Colman, A.S., Wofsy, S.C., 1998. Regional budgets for nitrogen oxides from continental sources: Variations of rates for oxidation and deposition with season and distance from source regions, *Journal of Geophysical Research* 103(7), 8355- 8368.

Mutius, E., Sherrill, D.L., Fritzsch, C., Martinez, F.D., Lebowitz, M.D., 1995. Air pollution and upper respiratory symptoms in children from East Germany, *European Respiratory Journal* 8, 723-728.

Namdeo, A., Tiwary, A., Farrow, E., 2011. Estimation of age-related vulnerability to air pollution: assessment of respiratory health at local scale. *Environmental International* 37(5), 829-37.

Nickmilder, M., de Burbure, C., Carbonnelle, S., Dumont,X., Bernard, A., Derouane A., 2007. Increase of exhaled nitric oxide in children exposed to low levels of ambient ozone. *Journal of Toxicological Environmental Health* 1:70 (3- 4):270-4.

NJDEP (New Jersey Department of Environmental Protection). 2006. Ozone: State of New Jersey Department of Environmental Protection [accessed March 21 2010] Available: http://www.state.nj.us/dep//airworkgroups/docs/ozone.pdf

NJDEP. 2006. 'Ozone: State of New Jersey Department of Environmental Protection', http://www.state.nj.us/dep/baqp/8hrsip/Chapter%203%20-a20Air%20Quality%20Trends.pdf

NJDEP. 2007. Ambient air monitoring plan. New Jersey Department of Environmental Protection, Bureau of Air Monitoring [accessed August 3 2010] Available: http://www.state.nj.us/dep/airmon/2007net.pdf

NJDHSS. 2008. New Jersey Department of Health and Senior Services (http://www.nj.gov/health/fhs/asthma/documents/asthma_strategic_plan2008-013.pdf)

Orazzo, F., Nespoli, L., Ito, K., Tassinari, D., Giardina, D., Funis, M., Cecchi, A., Trapani, C., Forgeschi, G., Vignini, M., Nosetti, L., Pigna, S., Zanobetti, A., 2009. Air Pollution, aeroallergens, and emergency room visits for acute respiratory diseases and gastroenteric disorders among young children in six Italian cities, *Environmental Health Perspectives* 117(11), 1780-1785.

Ostro, B.D., Tran, H., Levy, J.I., 2006. The health benefits of reduced tropospheric ozone in California. *Journal of Air Waste Management Association* 56, 1007-21.

Parish, D.D., 2006. Critical evaluation of US on-road vehicle emission inventories. Atmospheric Environment 40, 2288-2300.

Paschalidou, A.K., Kassomenos, P.A., 2004. Comparison of air pollutant concentrations between weekdays and weekends in Athens, Greece for various meteorological conditions. *Environmental Technology* 25(11), 1241-55.

Peel, J.L., Tolbert, P.E., Klein, M., Metzger, K.B., Flanders, W.D., Todd, K., Mulholland, J.A., Ryan, P.B., Frumkin, H., 2005. Ambient Air Pollution and Respiratory Emergency Department Visits. Epidemiology 16(2), 164-174.

Peters, A., Breitner, S., Cyrys, J., Stolzel, M., Pitz, M., Wolke, G. et al., 2009. 'The influence of improved air quality on mortality risks in Erfurt, Germany', *Resp. Rep. Health Eff. Inst.*, Vol. 137, pp.5-77, 79-90.

Pope, C.A., Burnett, R.T., Thun, M.J., Calle, E.E., Krewski, D., Ito, K. et al. (2002) 'Lung cancer, cardiopulmonary mortality and long term exposure to fine particulate air pollution, *Journal of the American Medical Association*, Vol. 287, No. 9, pp.1132–1141.

Pudasainee, D., Sapkota, B., Shrestha, M.L., Kaga, A., Kondo, A., Inoue, Y., 2006. Ground level ozone concentrations and its association with NOx and meteorological parameters in Kathmandu valley, Nepal, *Atmospheric Environment* 40, 8081-8087.

Qian, Z., Lin, H.M., Stewart, W.F., Kong, L., Xu, F., Zhou, D., Zhu, Z., Liang, S., Chen, W., Shah, N., Stetter, C., He, Q., 2010. Seasonal pattern of the acute mortality effects of air pollution, *J Air Waste Manag Assoc.* 60(4), 481-8.

Ren, C., Melly, S. and Schwartz, J. (2010) 'Modifiers of short-term effects of ozone on mortality in eastern Massachusetts – a case-crossover analysis at individual level', *Environ. Health*, Vol. 21, No. 9, p.3.

Ricciardolo, F. L. M., Ricciardolo, Sterk, P.J., Gaston, B., Folkerts, G., 2004. Nitric Oxide in Health and Disease of the Respiratory System. *American Physiological Society Review* 84, 731-765.

Rosenzweig, C., Soleckib, W.D., Parshalla, L., Chopping, M., Pope, G. and Goldberg, R. (2005) 'Characterizing the urban heat island in current and future climates in New Jersey', *Environmental Hazards*, Vol. 6, No. 1, pp.51–62.

Rossi, O.V., Kinnula, V.L., Tienari, J., Huhti, E., 1993. Association of severe asthma attacks with weather, pollen, and air pollutants, *Thorax* 48(3), 244-8.

Sadanaga, Y., Shibata, S., Hamana, M., Takenaka, N., Bandow, H., 2008. Weekday/weekend difference of ozone and its precursors in urban areas of Japan, focusing on nitrogen oxides and hydrocarbons, *Atmospheric Environment* 42, 4708-4723.

Samet, J.M., Utell, M.J., 1990. The risk of nitrogen dioxide: What have we learned from epidemiological and clinical studies. *Toxicology and Industrial Health, (USA)* 6:2, 247-262.

Schwartz, J. (2000) 'The distributed lag between air pollution and daily deaths', *Epidemiology*, Vol. 11, No. 3, pp.320–326.

Seinfeld, H.J., Pandis, N. S., 2006. Atmospheric chemistry and physics: From air pollution to climate change. Wiley-Interscience, 2nd Edition, New York.

Sillman, S., 1999. The relation between ozone, NOx and hydrocarbons in urban and polluted rural environments. Millenial Review series, *Atmospheric Environment* 33(12), 1821- 1845.

Smith, L.A., Mukerjee, S., Chung, K.C., Afghani, J., 2011. Spatial analysis and land use regression of VOCs and NO2 in Dallas, Texas during two seasons. *J Environ Monit.*13(4), 999-1007.

Song, F. and Gao, Y., 2009. 'Chemical characteristics of precipitation at metropolitan Newark in the US East Coast', *Atmospheric Environment*, Vol. 43, No. 32, pp.4903–4913.

Song, F., Shin, J.Y., Jusino-Atresino, R., Gao, Y., 2011. Relationships among the springtime ground-level NO_x, O_3 and NO_3 in the vicinity of highways in the US East Coast. *Atmospheric Pollution Research* 2, 374-383.

Stafoggia, M., Schwartz, J., Forastiere, F. and Perucci, C.A. (2008) 'Does temperature modify the association between air pollution and mortality? A multicity case-crossover analysis in Italy', *American Journal of Epidemiology*, Vol.167, No. 12, pp.1476–1485.

Stieb, D.M., Judek, S. and Burnett, R.T., 2002. 'Meta-analysis of time-series studies of air pollution and mortality: effects of gases and particles and the influence of cause of death, age and season', *J. Air and Waste Management Association*, Vol. 52, No. 4, pp.470–484.

Stieb, D.M., Szyszkowicz, M., Rowe, B.H., Lech, J.A., 2009. Air pollution and emergency department visits for cardiac and respiratory conditions: a multi-city time-series analysis. *Environmental Health* 10, 8-25.

Tager, I.B., Balmes, J., Lurmann, F., Ngo, L., Alcorn, S. and Kunzli, N., 2005. Chronic exposure to ambient ozone and lung function in young adults, *Epidemiology*, Vol. 16, No. 6, pp.751–759.

Thurston, G.D., Ito, K., Kinney, P.L., Lippmann, M., 1992. A mulit-year study of air pollution and respiratory hospital admissions in three New York State metropolitan areas: results for 1988 and 1989 summers. Journal of Exposure Analysis and Environmental epidemiology, 2(4), 429-450.

Torres-Jardon, R., Keener, T.C., 2006. Evaluation of ozone-nitrogen oxides-volatile organic compound sensitivity of Cincinnati, Ohio, *Journal of air and Waste Management Association* 56, 322-33.

Tsai, D.H., Wang, W.L., Wang, C.H., Chan, C.C., 2008. A study of ground-level ozone pollution, ozone precursors and subtropical meteorological conditions in central Taiwan, *Environmental Monitoring* 10(1), 109-18.

Tramuto, F., Cusimano, R., Cerame, G., Vultaggio, M., Calamusa, G., Maida, C.M., Vitale, F., 2011. Urban air pollution and emergency room admissions for respiratory symptoms: a casecrossover study in Palermo, Italy, *Environmental Health* 10(31) doi:10.1186/1476-069X-10-31

USEPA (United States Environmental Protection Agency). 2010. Ozone air quality standards http://www.epa.gov/glo/standards.html

USEPA (United States Environmental Protection Agency). 2006. Air quality criteria for ozone and related photochemical oxidants. Office of Research and Development, EPA/600/P-93/004aF, Vol. I-III.

US Environmental Protection Agency (USEPA) (2006) 'Air quality criteria for ozone and related photochemical oxidants', EPA/600/P-93/004aF, Vols. I–III, Office of Research and Development.

Vanderstraeten, P., Forton, M., Brasseur, O. and Offer, Z.Y., 2011. 'Black carbon instead of particle mass concentration as an indicator for the traffic related particles in the Brussels Capital Region', *Journal of Environmental Protection*, Vol. 2, No. 5, pp.525–532.

Varadarajan, C., 2007. Development of a source-meteorology-receptor (SMR) approach using fine particulate intermittent monitored concentration data for urban areas in Ohio', PhD thesis, University of Toledo, OH, available at http://etd.ohiolink.du/send- df.cgi/Varadarajan%20Charanya.pdf?toledo 1197655960 (accessed on 2 July 2010).

Vukovich, F.M. and Sherwell, J., 2002. Comparison of fine particles and the relationship between particle variations and meteorology at an urban site and a remote site in the Eastern United States, *J. Air and Waste Management Association*, Vol. 52, No. 5, pp.573–584.

Washam, C., 2011. Separating People from Pollution: Individual and Community Interventions to Mitigate Health Effects of Air Pollutants, *Environ. Health Perspect.* 119(1), A34.

Wei, X.L., Li, Y.S., Lam, K.S., Wang, A.Y., Wang, T. J., 2007. Impact of biogenic VOC emissions on a tropical cyclone-related ozone episode in the Pearl River Delta region, China, *Atmospheric Environment* 41, 7851-7864.

Weisel, C.P., Cody, R.P., Loiy, P.J., 1995. Relationship between Summertime Ambient Ozone Levels and Emergency Department Visits for Asthma in Central New Jersey. *Environmental Health Perspectives* 103, 97-102.

Weisel, C.P., Cody, R.P., Georgopoulos, P.G., Purushothaman, V., Weiss, S.H., Bielory, L., et al., 2002. Concepts in developing health-based indicators for ozone. *International Archives Occupational Environmental Health* 75(6):415-422.

WHO (World Health Organization), 2003. Health aspects of air pollution with particulate matter, ozone and nitrogen dioxide. Report on WHO working group Available: http://ec.europa.eu/environment/archives/cafe/activities/pdf/1st_report.pdf

White, J.E., 2010. AIRNow-International: the future of the United States real-time air quality reporting and forecasting program with GEOSS participation', available at http://www.earthzine.org/2010/01/25/airnow-international (accessed on 5 November 2010).

Williams, D.J., Potter, W.T., Clarkson, W.W., Sanders, D.A., Stevens, J.E., 2009. The impact of background ozone on compliance with revised National Ambient Air Quality Standards. *J Air Waste Manag. Assoc.* 59(1), 52-7.

Wilson, J.G., Kingham, S., Pearce, J. and Sturman, A.P. (2005) 'A review of intraurban variations in particulate air pollution: implications for epidemiological research', *Atmospheric Environment*, Vol. 39, No. 34, pp.6444–6462.

Yarwood, G., Stoeckenius, T.E., Heiken, J.G., Dunker, A.M., 2003. Modeling weekday/ weekend ozone differences in the Los Angeles region for 1997. Journal of Air Waste Management Association 53, 864-75.

Yerramilli, A., Dodla, V.B., Desamsetti, S., Challa, S.V., Young, J.H., Patrick, C., Baham, J. M., Hughes, R. L., Yerramilli, S., Tuluri, F., Hardy, M. G., Swanier, S. J., 2011. Air quality modeling for the urban Jackson, Mississippi Region Using a High Resolution WRF/Chem Model, Int. J. Environ. Res. Public Health 8(6), 2470-2490.

Zhao, Y. and Gao, Y. (2008a) 'Mass size distributions of water-soluble inorganic and organic ions in size segregated aerosols over metropolitan Newark in the US East Coast', *Atmospheric Environment*, Vol. 42, No. 18, pp.4063–4078.

Appendices

Appendix I. Computations for principal component analysis

A-1. Descriptive statistics for all variables

A-2. Stepwise Regression Model for (a) ozone using the original independent variables, (b) logOzone, (c) $logNO_2$

A-3. Pearson correlation matrix of independent variables

A-4. Rotated principal component loadings (Rotation method: varimax with Kaiser normalization)

A-5. Final calculations for partial regression model

A- 1: Descriptive statistics for all variables

	N	Minimum	Maximum	Mean	Std. Deviation
TEMP	344	16.00	88.00	55.00	16.77
HUM	344	25.00	90.00	61.52	13.98
PRESS	344	29.41	30.66	30.02	0.23
WIND	344	1.00	18.00	6.31	3.20
NO	344	0.22	39.89	5.70	6.03
NO2	344	3.29	37.37	15.64	7.20
OZONE	344	0.24	55.56	22.05	11.57
logOZONE	344	-0.61	1.74	1.26	0.32
logNO2	344	0.52	1.57	1.14	0.21

A-2a: Stepwise regression model for ozone using the original independent variables

Predictors	Constant	TEMP	HUM	PRESS	WIND	NO	NO2
Coefficient	241.792	0.388	-0.361	-7.364	0.422	-0.253	0.059
p-value	0.001	0.000	0.000	0.001	0.022	0.017	0.506
Adjusted R square		0.258	0.453	0.489	0.508	0.515	0.514

A- 2b: Stepwise regression model for log(ozone) using the original independent variables

Predictors	Constant	TEMP	HUM	PRESS	WIND	NO	NO2
Coefficient	6.480	0.009	-0.011	-0.172	0.015	-0.018	0.009
p-value	0.001	0.000	0.000	0.006	0.002	0.000	0.000
Adjusted R square		0.166	0.408	0.443	0.483	0.520	0.536

A- 2c: Stepwise regression model for log(NO2) using the original independent variables

Predictors	Constant	TEMP	HUM	PRESS	WIND	NO
Coefficient	-1.775	0.000	0.004	0.090	-0.019	0.015
p-value	0.165	0.555	0.000	0.030	0.000	0.000
Adjusted R square		0.003	0.151	0.228	0.412	0.530

A- 3: Pearson correlation matrix of independent variables

	TEMP	HUM	PRESS	WIND	NO	NO2
TEMP	1.000	.172	-.272	-.290	-.110	.073
HUM		1.000	-.303	-.244	.273	.330
PRESS			1.000	-.215	.140	.115
WIND				1.000	**-.475**	**-.526**
NO					1.000	**.683**
NO2						1.000

A- 4. Rotated principal component loadings (Rotation method: varimax with Kaiser normalization) Predictors: TEMP, HUM, PRESS, WIND, NO, NO2

	Principal Component					
	1	2	3	4	5	6
TEMP	.973	.070	-.141	-.070	-.148	.026
HUM	.073	.963	-.166	.112	-.100	.134
PRESS	-.143	-.163	.967	.053	-.116	.050
WIND	-.179	-.112	-.134	-.214	.918	-.224
NO	-.091	.127	.060	.902	-.219	.333
NO2	.037	.161	.058	.348	-.240	.889
Eigenvalue	1.015	1.013	1.007	1.000	0.993	0.972
% of Variance	16.916	16.882	16.778	16.671	16.548	16.206
Cumulative %	16.916	33.797	50.575	67.246	83.794	100.000

A-5. Final calculations for the partial regression model

Zscore(logOZONE)=.432PC1-.422PC2-.133PC3-.365PC4+.162PC5-.034PC6

C1=.973 Zscore(TEMP)+.073 Zscore(HUM)-.143 Zscore(PRESS)-.179 Zscore(WIND) -.091 Zscore(NO)+.037 Zscore(NO2)

C2=.07 Zscore(TEMP)+.963 Zscore(HUM)-.163 Zscore(PRESS)-.112 Zscore(WIND) +.127 Zscore(NO)+.161 Zscore(NO2)

C3= -.141 Zscore(TEMP)-.166 Zscore(HUM)+.967 Zscore(PRESS)-.134 Zscore(WIND) +.06 Zscore(NO)+.058 Zscore(NO2)

C4= -.07 Zscore(TEMP)+.112 Zscore(HUM)+.053 Zscore(PRESS)-.214 Zscore(WIND) +.902 Zscore(NO)+.348 Zscore(NO2)

C5= -.148 Zscore(TEMP)-.1 Zscore(HUM)-.116 Zscore(PRESS)+.918 Zscore(WIND) -.219 Zscore(NO)-.24 Zscore(NO2)

C6=.026 Zscore(TEMP)+.134 Zscore(HUM)+.05 Zscore(PRESS)-.224 Zscore(WIND) +.333 Zscore(NO)+.889 Zscore(NO2)

Zscore(logOZONE)=.41029 Zscore(TEMP)-.414408 Zscore(HUM)
-.16144 Zscore(PRESS)+.2222 Zscore(WIND)
-.476916 Zscore(NO)-.255798 Zscore(NO2)

Final General Partial Regression Model:

Log(OZONE) = 2.085584 + .007906 TEMP -.009584 HUM -.01629 PRESS +.022426 WIND -.025547 NO -.011489 NO2

$$Zscore(logNO2)=.046PC1+.511PC2+.306PC3+.115PC4-.409PC5$$

$$C1=.972\ Zscore(TEMP)+.074\ Zscore(HUM)-.142\ Zscore(PRESS)-.177\ Zscore(WIND) -.080\ Zscore(NO)$$

$$C2=-.07\ Zscore(TEMP)+.135\ Zscore(HUM)+.063\ Zscore(PRESS)-.254\ Zscore(WIND) +.956\ Zscore(NO)$$

$$C3= .073\ Zscore(TEMP)+.968\ Zscore(HUM)-.163\ Zscore(PRESS)-.120\ Zscore(WIND) +.139\ Zscore(NO)$$

$$C4= -.141\ Zscore(TEMP)-.162\ Zscore(HUM)+.967\ Zscore(PRESS)-.134\ Zscore(WIND) +.064\ Zscore(NO)$$

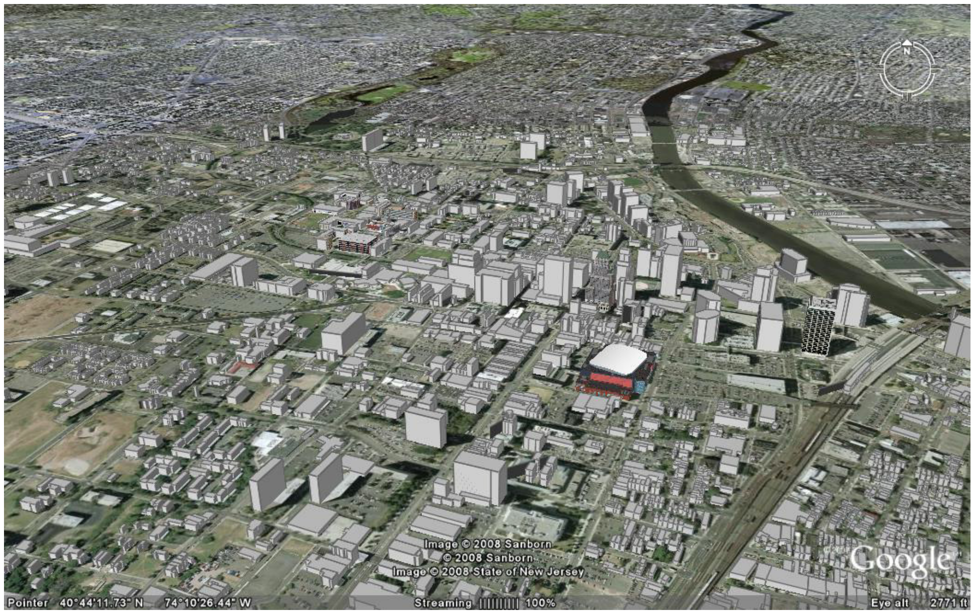
$$C5= -.157\ Zscore(TEMP)-.110\ Zscore(HUM)-.122\ Zscore(PRESS)+.934\ Zscore(WIND) -.236\ Zscore(NO)$$

$$Zscore(logNO2)= .077234\ Zscore(TEMP)+ .394957\ Zscore(HUM) +.136886\ Zscore(PRESS) -.57207\ Zscore(WIND) +.631254\ Zscore(NO)$$

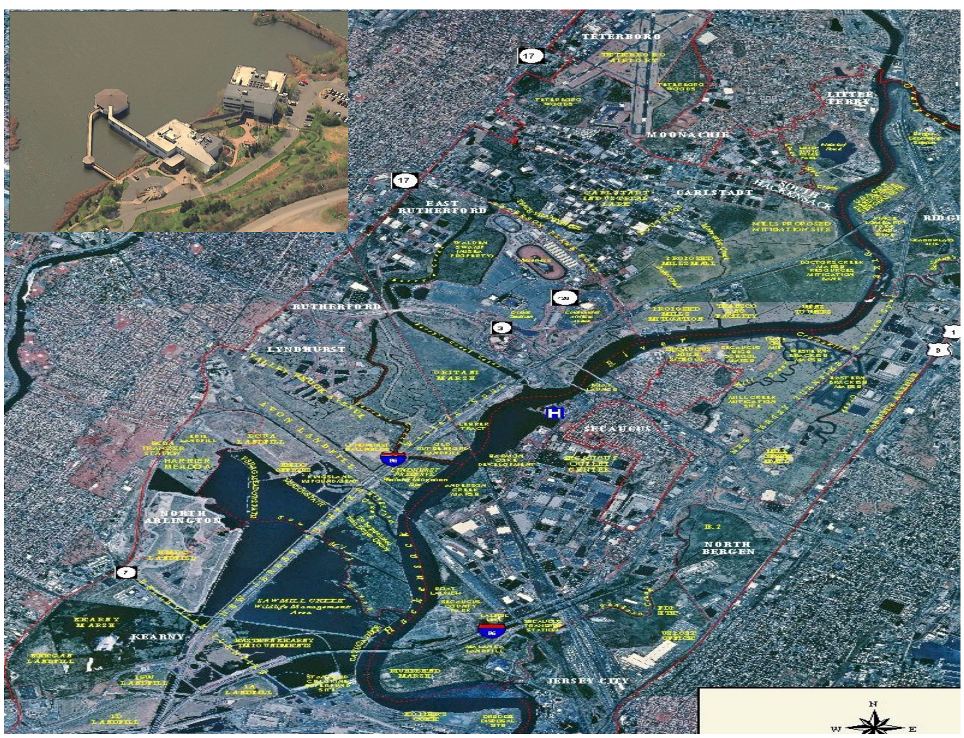
Final General Partial Regression Model:

$$Log(NO2) = .55519229 + .00098969\ TEMP + .006073206\ HUM + .009186\ PRESS - .03839\ WIND + .02248296\ NO$$

Appendix II: Satellite image of downtown Newark



Appendix III. Satellite image of the Meadowlands District (Inset: MERI)



Appendix IV. PM Sampling platform on roof of Bradley Hall, Rutgers-Newark

Appendix V. Formula used for mass concentrations of particulate matter

$C_f = \left(\frac{Mf}{Vf}\right)$

$C_c = \left(\frac{Mc}{Vt}\right) - \left(\frac{Vc}{Vt}\right) C_f$

$C_t = C_f + C_c$

Where: Cf = mass concentration [μg/m^3] of fine particle fraction

Cc = mass concentration [μg/m^3] of coarse particle fraction

Ct = mass concentration [μg/m^3] of PM_{10}

Mf = mass [μg] collected on fine particle fraction filter

Mc = mass [μg] collected on coarse particle fraction filter

Vf = volume [m^3] of air sampled through fine particle fraction filter

Vc = volume [m^3] of air sampled through coarse particle fraction filter

Vt = volume [m^3] of air sampled through fine and coarse particle fraction filter

Appendix VI. Correlation matrix of weather variables, pollutants and admissions

Bivariate Correlations

		Temp	Hum	Press	Wind	Ozone	NO	NOx	NO2	Admissions
Temp	Pearson Correlation	1	.159**	-.285**	-.270**	.530**	-.161**	-.077	.036	-.001
	Sig. (2-tailed)		.002	.000	.000	.000	.002	.148	.495	.990
	N	366	366	366	366	359	359	359	359	366
Hum	Pearson Correlation	.159**	1	-.303**	-.243**	-.339**	.188**	.249**	.263**	.021
	Sig. (2-tailed)	.002		.000	.000	.000	.000	.000	.000	.691
	N	366	366	366	366	359	359	359	359	366
Press	Pearson Correlation	-.285**	-.303**	1	-.200**	-.206**	.176**	.158**	.103	-.057
	Sig. (2-tailed)	.000	.000		.000	.000	.001	.003	.052	.280
	N	366	366	366	366	359	359	359	359	366
Wind	Pearson Correlation	-.270**	-.243**	-.200**	1	.152**	-.417**	-.521**	-.523**	-.017
	Sig. (2-tailed)	.000	.000	.000		.004	.000	.000	.000	.752
	N	366	366	366	366	359	359	359	359	366
Ozone	Pearson Correlation	.530**	-.339**	-.206**	.152**	1	-.382**	-.357**	-.250**	-.092
	Sig. (2-tailed)	.000	.000	.000	.004		.000	.000	.000	.083
	N	359	359	359	359	359	359	359	359	359
NO	Pearson Correlation	-.161**	.188**	.176**	-.417**	-.382**	1	.909**	.604**	.217**
	Sig. (2-tailed)	.002	.000	.001	.000	.000		.000	.000	.000
	N	359	359	359	359	359	359	359	359	359
NOx	Pearson Correlation	-.077	.249**	.158**	-.521**	-.357**	.909**	1	.882**	.317**
	Sig. (2-tailed)	.148	.000	.003	.000	.000	.000		.000	.000
	N	359	359	359	359	359	359	359	359	359
NO2	Pearson Correlation	.036	.263**	.103	-.523**	-.250**	.604**	.882**	1	.359**
	Sig. (2-tailed)	.495	.000	.052	.000	.000	.000	.000		.000
	N	359	359	359	359	359	359	359	359	359
Admissions	Pearson Correlation	-.001	.021	-.057	-.017	-.092	.217**	.317**	.359**	1
	Sig. (2-tailed)	.990	.691	.280	.752	.083	.000	.000	.000	
	N	366	366	366	366	359	359	359	359	366

**. Correlation is significant at the 0.01 level (2-tailed).

Curriculum Vitae
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1981-1986	Mahaicony Secondary School, Guyana.
1987-1990	Teacher's Certificate, Cyril Potter College of Education, Guyana.
1991-1995	B.A. Geography, University of Guyana.
1996-1998	M.Phil. Environment and Development, University of Cambridge, U.K.
1999-2000	Environmental Health Educator, Safe Space, NYC.
2000-2002	Program Coordinator, 21st Century Community Learning Centers (CCLC) Safe Space, Inc.
2002-2005	Director of Education, 21st (CCLC) Safe Space, NYC.
2004-2012	Adjunct Lecturer, The City University of New York.
2005-2009	Director of Health Access, Safe Space, NYC.
2007	Peer-reviewed article: Roberts-Semple, D., 2007. Protection or Survival? The conflict between conservation and exploitation of tropical rainforests, Forum on Public Policy, 3(4): 409-416.
2008-2012	Ph.D. Environmental Science, Rutgers University, Newark, New Jersey.
2009-2011	Graduate Teaching Fellow, Rutgers University, Newark, New Jersey.
2011-2012	Graduate Assistant, Rutgers University, Newark, New Jersey.
2012	Peer-reviewed article: Roberts-Semple, D., Gao, Y., 2012. Evaluation of air pollution, local meteorology and urban public health, *Int. J. Env. Technology and Management*, special issue, in press.
2012	Peer-reviewed article: Roberts-Semple, D., Song, F., Gao, Y., 2012. Seasonal characteristics of ambient nitrogen oxides and ground-level ozone in metropolitan northeastern New Jersey, *Atmospheric Pollution Research* 3, 247-257, doi: 10.5094/APR.2012.027.
2012	Peer-reviewed article: Roberts-Semple, D., Gao, Y., 2012, Impact of gas-phase air pollution on public health in metropolitan northeastern New Jersey, *Environmental Research*, in review.