Pollution and Climate-Induced Respiratory Health Effects: Do National Standards Adequately Provide a Safe Threshold for NO_x and O₃ in Urban New Jersey?

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Despite the decline in ambient concentrations of nitrogen oxides (NO_x) and ground-level ozone (O₃), adverse health effects have been observed at levels below federal standards. We examine the meteorological dependence of NO_x and O₃ for resultant respiratory health outcomes. The objectives were to characterize the temporal and seasonal variations in ambient concentrations of NO_x and O₃ as a plausible explanation for respiratory hospital admissions in the Meadowlands. The meteorological conditions and air pollutants that may be associated with respiratory hospital admissions were analyzed through path analytic models by using stepwise multiple regressions and bivariate correlations. Increases in O₃ and NO_x were associated with respiratory hospital admissions. The lowest hospital admissions occurred when NO_x concentrations were also at their lowest and corresponded to months with maximum photochemical activity. Hospital admissions in the fall, winter and spring were higher than in summer. There was a positive relationship between hospital admissions and personal exposure to NO₂ ($r=0.359$) and NO_x ($r=0.317$) over the short-term. With one unit increase in O₃, respiratory hospital admissions increased by 0.7 (95% CI: 0.254, 1.23) at 2 lag days and 0.5 (95% CI: 0.107, 0.969) with one unit increase in NO_x. Respiratory hospital admissions were not strongly distinguished by high O₃ concentrations in the summer except at lag intervals. Unlike O₃, NO_x did not show consistent seasonal behavior. This study provides evidence for additional plans for protection against respiratory illnesses, as well as an incentive for setting improved air quality standards in similar regions.

Keywords: Hospital Admissions, Meteorology, Nitrogen Oxides, Ozone, Seasonal Variations.

Introduction

Previous studies have shown that nitrogen oxides (NO_x) and ozone (O₃) pose major risks to public health (Weisel et al., 2002; 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), O₃ levels have repeatedly exceeded the national standard in many areas (Godish et al., 2014;

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Williams et al., 2009; USEPA, 2010). Lower levels of NO_x can result in higher O_3 levels, explained further below (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 et al., 2014; Brown et al., 2006); however, emissions from anthropogenic sources exceed natural sources (Godish et al., 2014). 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 et al., 2014; Tramuto et al., 2011; Washam, 2011).

Ground-level O_3 comes from two major sources: regionally from the upper stratosphere via convection movements (Godish et al., 2014), and locally by photochemical reactions (R1-R3). 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 (R4) (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 et al., 2014); 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 attacks causing premature mortality and reduced life expectancy (WHO, 2003; Bell, 2006; 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; Bell, 2006; Torres-Jardon and Keener, 2006; Orazzo et al., 2009; Ji et al., 2011). Not as many studies, however, have analyzed NO_x and O_3 (Thurston

GD et al., 1992; Godish et al., 2014) to characterize the underlying mechanisms of toxicity and respiratory impacts from the perspective of spatial and seasonal patterns (Tager IB et al., 2005; Ito et al., 2007; Tsai et al., 2008). In addition, many monitoring networks lack local information, particularly for traffic-generated pollutants known to vary on small scales (NJDEP, 2007; Kan et al., 2010; Giles et al., 2011).

The health-based O₃ standard has been exceeded in many states (Sillman, 1999; Godish et al., 2014) including New Jersey (Madsen and Mottola, 2003), where 19 and 29 unhealthy O₃ pollution days occurred in 2004 and 2005, respectively (NJDEP, 2006). In central and northern New Jersey, average O₃ levels greater than 0.06 ppm, caused emergency room (ER) visits 28% more frequently, demonstrating that O₃ 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 research assesses the impact of NO_x and O₃ on human health when influenced by local meteorology in the 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 O₃ concentrations and potential health effects not fully studied in New Jersey.

Materials and Methods

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 (Figure 1).

The prevailing winds blow from the northwest in winter. On summer days, winds usually blow 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 1. Situation of Study Site/Monitoring Station in Proximity to Highways and Pollutant Source Regions

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 (41°N, 74°W), ~ 1 km away from major highways from June 1, 2007 to May 31, 2008. Air was pumped into the instruments from inlets via two separate plastic tubes. 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).

Meteorological and Public Health Data

Meteorological data from the Lyndhurst site were supplemented with data from Weather Underground (www.underground.com). The primary meteorological parameters were temperature, wind speed, wind direction and barometric pressure. The daily counts of both ED and hospital admission data were obtained from the New Jersey Department of Health and Senior Services (NJDHSS). Health records based on International Classification of Diseases, Ninth Revision (ICD-9) codes, were approved by the Institutional Review Board (IRB). Since the data included non-confidential hospital record, individual identifiers were not necessary and permission 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 diagnostic codes for the following fourteen outcomes: 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 ER visits and hospital admission was determined by summing the daily count of each health condition considered a primary diagnosis in each category.

Data Analyses

Data analysis was conducted with SAS 9.2 and MINITAB 16 version software packages; SPSS (Version 18.0) was used for data input. 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 to estimate demonstrate 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 (Stieb, 2009). It demonstrated 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, etcetera (Brook et al., 2004; Dominici et al., 2004).

Results and Discussion

Seasonal Variations

Figure 2. Temporal Variations of Daily Average (a) O_3 and (b) NO_x Concentrations in One-year Period. Ozone Shows a Summer Maxima and Winter Minima with Pronounced Variations in Colder Months; NO_x Displays an Opposite Trend. Variations in NO_x Greater in Summer than in Winter

The average monthly/annual concentrations and variations of O_3 , NO_x , NO and NO_2 are presented in Table 1. 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 (Figure 2a). Conversely, NO_x concentrations were at their lowest during the summer months but did not display a strong seasonal pattern (Figure 2b), probably attributed to poor dependence on meteorological conditions (Khoder, 2009; Smith et al., 2011). Similarly NO and NO_2 consisted of a moderate winter peak and summer minima (Khoder, 2009; Bigi and Harrison, 2010; Smith et al., 2011).

Table 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

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 NO₂ 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 NO₂ while the photochemical formation of O₃ is inhibited by the lack of intense solar radiation (Sadanaga et al., 2008; Geddes et al., 2009).

The dependence of O₃ on NO_x is highly non-linear; O₃ 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 O₃ production in summer has made it difficult for local and regional regulatory agencies to meet a stringent 8-hr O₃ 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.

Seasonal Effect on Hospital Admissions

A greater effect of air pollution on respiratory hospital admissions is evident in cool and cold seasons (Fall, Winter and Spring) than in summer months (Figure 3), consistent with previous investigations. The statistical association between seasonal variations and respiratory admissions in Spring was $p < 0.001$, the season with greatest variability (1%).

Figure 3. Time Series of Hospital Admissions Show an Increasing Trend over the One-year Period (June 1, 2007 to May 31, 2008)

The seasons are defined as: fall (September to November), winter (December to February), Spring (March to May) and summer (June to August). 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. 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.

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).

Air Pollution Concentrations and Daily Hospital Admissions

There was a gradual increase in hospital admissions over the study period (Figure 3), corresponding to a small increase in the monthly mean NO_2 concentration (Figure 4).

The greatest effect of air pollution on respiratory hospital admissions is evident in the winter season (Figure 4a); a steady increase in hospital admissions was followed by a gradual decrease in spring (Figure 4b), fluctuations in summer (Figure 4c), and an increasing trend followed by a slight decrease in the fall (Figure 4d). 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; Clougherty et al., 2013). While the present study did not examine the age-related effect of air pollutants, 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. 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

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 (Stieb, 2009). 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 , when 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 in the multiple/forward stepwise 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 (Stieb 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 ($20\mu g/m^3$) in daily O_3 concentrations on the same day in the summer. A larger effect was found at lag 2, independent of other pollutants.

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 (Figure 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).

Figure 5. 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

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). 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.

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 for pollutants needed to be run by season, to provide 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.

Our 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 *Seasonal Variations* 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 (Figure 6a), when NO_x concentrations were highest and O_3 concentrations were lowest.

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 (Figure 6b). Conversely, the highest wind speeds occurred when O_3 concentrations declined to a minimum, generally from October to May, explaining more than 32% variability of the log value 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.

Figure 6. Meteorological Parameters: Wind Speed (a), Show an Increasing Trend and Temperature (b), Showing a Decreasing Trend from June 1, 2007 to May 31, 2008

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).

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 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 $y = \beta_0 + \beta_1 x_1 + \beta_2 x_2 + \beta_3 x_3 + \dots + \beta_p x_p + \varepsilon$ where y is the dependent variable (respiratory hospital admissions), x_i is the i th independent variable, β_i is the regression coefficient, β_0 is the intercept, p is the number of independent variables, and ε is the error with mean zero. In the final model, hospital admissions = 1310.926 + 165.798 weekday - 0.456 relative humidity - 29.758 barometric pressure + 0.742 O₃ lag 2 + 0.538 NO_x + 30.607 fall + 28.612 winter + 32.089 spring + 0.459 NO_x lag 5 (Table 2).

Table 2. 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

¹Variability of hospital admissions explained
²344 data pts without lag values
³295 data pts with lag values

After deleting all incomplete 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 equal variance and independence. The outlier detection identified 5 leverages and 8 large standardized residuals. When compared to other weekdays, there were 4 unusually low (Figure 7) and a few missing values of O₃, NO_x, NO and NO₂, that occurred on holidays (Figure 8).

Figure 7. Boxplots of Respiratory Hospital Admissions on Weekday and Weekends (1 = Weekday or if the Day is not Saturday/Sunday and Weekends (0 = Weekend or if the Day is Saturday/ Sunday)

Figure 8. Boxplots of O_3 , NO , NO_x and NO_2 Depicting the Distribution of Each Variable over the Study Period

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, the variables made up 344 data points (see Table 2) 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 (see Figure 9).

Figure 9. *Residual Diagnostic Distribution of Hospital Admissions. The Symmetrical Histogram Plot of Residuals Confirms the Normality Assumption*

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. 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. 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 2). Figure 10 shows the Residual Diagnostic Q-Q Plot for Hospital Admissions.

Figure 10. *Residual Diagnostic Q-Q Plot for Hospital Admissions. All Residuals are along a Straight Line*

There was a positive relationship between hospital admissions and personal exposure to NO₂ ($r=0.359$) and NO_x ($r=0.317$) over the short-term. With one unit increase in O₃, respiratory hospital admissions increased by 0.7 (95% CI: 0.254, 1.23) at 2 lag days and 0.5 (95% CI: 0.107, 0.969) with one unit increase in NO_x. Since O₃ had the least overall direct impact on respiratory hospital admissions, it was eliminated from the final model. Among the pollutants, NO₂ 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.

Conclusions

This study shows statistical associations between levels of individual or combined gaseous air pollutants and respiratory hospital admissions. 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 O₃ concentrations with adverse health outcomes, it presents evidence that is adequately consistent, coherent, and plausible. It demonstrates that at ambient levels, NO_x and O₃ may be enhanced by certain meteorological conditions with resultant health effects.

Personal exposure to respiratory health problems show association for O₃ 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 O₃ production efficiency which limits precursor pollutants, NO₂ seemed to have a greater overall effect on respiratory hospital admissions, a valuable contribution to current air pollution literature. Variations in O₃ occurred by season and specific time of day, with the highest concentrations in the summer and early afternoon, whereas traffic-related pollutants, NO and NO₂, 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.

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