eThesis: Linking Meteorology, Air Pollution, and Health in Melbourne, Australia

John Lanier Pearce, 2011

A PhD Thesis Submitted to the School of Geography and Environmental Science of MONASH UNIVERSITY in Partial Fulfillment of the Requirements for the Degree

DOCTOR OF PHILOSOPHY

by

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MELBOURNE, AUSTRALIA

2011

LINKING METEOROLOGY, AIR POLLUTION AND HEALTH IN MELBOURNE, AUSTRALIA

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Commencement Date: 27 March 2008

Confirmation Date: 31 October 2008

Mid-Candidature Date: 04 June 2010

Submission Date: 15 July 2011

ABSTRACT

The importance of meteorology in air pollution processes and its influence on human health is understood; however, these relationships and their interactions are expected to be different under a changing climate. Thus, a considerable challenge is presented to those charged with air quality management because alterations in meteorological conditions stemming from a changing climate will vary from one geographic region to the next. To further complicate matters, the influence of weather on air pollution and related health effects also varies geographically. Therefore, in order to better understand these consequences for any given region around the globe, regional scale studies are required. Such studies also elucidate the processes and how they may be similar and different between regions. The aim of this thesis is to assess the relationships between meteorology, air pollution and human health for Melbourne, Australia during the years 1999 to 2006 to provide insight into how this may alter under changing climate conditions. This is achieved using a novel cross-disciplinary approach that draws from the fields of atmospheric science, epidemiology, and statistics.

In the first part of the study, the influence of synoptic-scale circulation features on daily concentrations of ozone (O_3), particulate matter ≤ 10 μm (PM_{10}), and nitrogen dioxide (NO_2) were characterized by using a synoptic climatology developed using self-organizing maps (SOMs) and applied within the framework of a generalized additive model (GAM). Results demonstrated that large-scale circulation features were not a primary driver of local air quality during our study period. Nevertheless, differential effects were found between circulation features with a general trend of anticyclones being associated with significantly poorer air quality. In particular, NO_2 and O_3 were 20% higher than average when synoptic conditions resulted in a northeast

gradient wind over the region. For PM_{10}, maximum increases of up to 20% over normal concentrations occurred when a strong anticyclone was centered directly over the region.

The second part of the study again applied the framework of GAM and characterized the relationship between locally observed weather elements and daily pollutant concentrations. These findings demonstrated that local-scale meteorological conditions were a more important driver of air quality than synoptic-scale circulation. The key finding in this analysis was that when daily maximum temperatures exceeded 35 °C; O_3, PM_{10} and NO_2 concentrations were 150%, 150% and 120% higher than the average. Other elements such as winds, boundary layer height, and atmospheric moisture were also important; however, their influences were marginal when compared to temperature.

The final part of the study, an ecological epidemiological study, examined the statistical relationship between daily mortality and air pollution across different temperatures using case-crossover analysis (CCO) and GAM. Results showed that temperature behaved as an effect modifier in the air pollution-mortality relationship with each pollutant exhibiting stronger effects on mortality as ambient temperatures increased. These findings, expressed as the percentage change in mortality along with their 95% confidence intervals, were strongest when temperatures exceeded 22 °C as mortality increased 2.82% [0.84, 4.85] per 10 ppb increase in O_3, 3.14% [1.57, 4.75] per 10 µg increase in PM_{10}, and 5.05% [1.16, 9.10] per 10 ppb increase in NO_2.

In conclusion, findings from this thesis provide a direct link between weather, air quality, and human health. Moreover, the strength of weather – in particular temperature – as a driver in these relationships suggests that if current climate projections hold true and all else remains the same then air quality will decline. Finally, characterizing these relationships through the use of statistical analyses is of added benefit as results are a direct reflection of patterns in observed data and thus lack the bias present in deterministic modeling studies.

ACKNOWLEDGEMENTS

Over the past three years, I learnt a great deal in the process of my PhD candidature. Naturally, I could not have completed this endeavor on my own. This section is to recognize the individuals who helped me get this far.

First, I would like to acknowledge the people whom were directly involved in my PhD – Associate Professor Jason Beringer, Professor Nigel Tapper, and Professor Neville Nicholls. Gentlemen, you have each been exemplary supervisors and I could not have completed this PhD without your help. I am forever grateful for the time you have invested in me as I have grown tremendously as a researcher under your guidance. Your friendship and professional collaboration meant a great deal to me during this process. Secondly, a number of colleagues in Australia also provided material and support during this PhD. In this regard, I am indebted to Sean Walsh, Petteri Uotila, Margaret Loughalan, and Martine Dennekamp. Thirdly, to the Monash Research Graduate School and the Environmental Protection Authority, Victoria, for providing scholarship support and funding throughout my candidature.

Furthermore, to my mother-in-law, Anda Olsen, thank you for spending the countless hours reading and commenting on each draft of this manuscript and the research papers within. During your unofficial tenor as my editor, I have grown tremendously as a writer and thus am forever thankful.

In addition, on a more personal note, I would like to say thank you to my wife, Jama, for her support during the last three years. You have moved across the world with me, stayed with me during the first year when we lived in a shoebox and realized that Melbourne was actually quite cold, and continued to stay with me when it was so hard to leave a place that we had grown to love. I could not have done it without you.

Finally, to my newborn son, Lanier, during the last stages of my PhD you have provided moments of chaos and moments of pure joy. I would not change it for the world. Your birth was the final push I needed to finish this bloody thing!

GENERAL DECLARATION

Declaration for thesis based or partially based on conjointly published or unpublished work.

In accordance with Monash University Doctorate Regulation 17/ Doctor of Philosophy and Master of Philosophy (MPhil) regulations the following declarations are made:

I hereby declare that this thesis contains no material which has been accepted for the award of any other degree or diploma at any university or equivalent institution and that, to the best of my knowledge and belief, this thesis contains no material previously published or written by another person, except where due reference is made in the text of the thesis.

This thesis includes two original papers published in peer reviewed journals and one submitted unpublished publication. The core theme of the thesis is the influence of weather on air pollution and related health effects in Melbourne, Australia. The ideas, development and writing up of all the papers in the thesis were the principal responsibility of myself, the candidate, working within the School of Geography and Environmental Science under the supervision of Associate Professor Jason Beringer, Professor Nigel Tapper, and Professor Neville Nicholls.

The inclusion of co-authors reflects the fact that the work came from active collaboration between researchers and acknowledges input into team-based research.

In the case of Chapters 3, 4, and 5, my contribution to the work involved the following:

Thesis Chapter	Publication title	Publication status	Nature and extent of candidate's contribution
3	Investigating the influence of synoptic-scale meteorology on air quality using self-organizing maps and generalized additive modeling	Published	Formulation of research problem and the context of the research in the wider literature; data acquisition; direction for model approach; interpretation of results and writing.
4	Quantifying the influence of local meteorology on air quality using generalized additive models	Published	Same as above.
5	The influence of temperature on air pollution-mortality relationships in Melbourne, Australia	Submitted	Same as above.

Signed: [redacted]

Date: 6 July 2011

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CHAPTER 1
INTRODUCTION

INTRODUCTION

1.1 MOTIVATIONS AND RATIONALE

The importance of air quality on human health and the local environment has been known for centuries. In the Middle Ages, London air was so badly polluted by smoke that in 1301 Edward I passed a law banning coal burning in an attempt to curb smoke emissions (EPA 2010). Later, several health outcomes were linked to major air pollution episodes in the Meuse Valley in 1930, Donora, Pennsylvania, in 1948, and the London in 1952. Presently, the World Health Organization estimates that outdoor air pollution causes approximately two million premature deaths worldwide per year (WHO 2008). In the European Union, exposure to particulate matter alone is estimated to claim an average of 8.6 months from the life of every European (HEN 2011) and in the United States, air pollution is estimated to cost urban areas somewhere in the range of 71 to 277 billion dollars annually in health related costs (Muller and Mendelsohn 2007). For Australia, air pollution is estimated to be responsible for 2.3% of all deaths annually and in Sydney, it is responsible for almost 2000 hospitalizations costing nearly $4.7 billion per year (NSWDOH 2011). These are the costs in affluent developed nations with relatively clean air. The costs in developing countries are suspected to be far greater.

So how does air pollution affect us on an individual level? First, it is important to consider that the average adult breathes nearly 12 cubic meters of air every day and that children breathe even more air per kilogram of body weight (Doak 2003). This means that, on a daily basis, we inhale more far more air by volume than food or water. Unfortunately, everything we inhale has the potential to affect the body's circulatory, immune, and respiratory systems. In the case of air pollution, these effects are usually adverse. Epidemiological studies around the globe have shown that exposure to ambient air pollution can cause burning of the eyes, throat irritation, and breathing difficulties, and have linked exposure with cancer, damage to the immune, neurological, reproductive, and respiratory systems and even death (Samet JM 2000; Brunekreef and Holgate 2002; Dockery 2009). These findings have been largely linked to three common air pollutants - particulate matter (PM), ozone (O_3), and nitrogen dioxide (NO_2). Each of these is described briefly below.

Particulate matter (PM) is a general term used to describing air pollution consisting of a mixture of particles that can be solid, liquid, or both, suspended in the air and representing a complex mixture of organic and inorganic substances (Ebi and McGregor 2008). From a public health standpoint particulate matter is considered a risk factor to human health associated with air pollution as adverse health effects have been found for particles <10μm in aerodynamic diameter (Samet, Dominici et al. 2000). For regulatory purposes particulate matter ≤ 10 μm is referred to as PM_{10} and particulate matter ≤ 2.5 μm in diameter is termed $PM_{2.5.}$ Particle size is used as the differentiating factor associated with health risk as size plays an important role in determining the degree of penetration into the respiratory system of which a particle is capable. However, chemical composition has also been found to play a role in addition to size. Many scientific studies link breathing PM to a series of significant health problems, including: aggravated asthma, increases in respiratory symptoms like coughing and difficult or painful breathing, chronic bronchitis, decreased lung function, and premature death (Dockery 2009).

Ozone (O_3) is a gaseous molecule consisting of three oxygen atoms that is formed in the lower atmosphere by the photochemical reaction of precursors (volatile organic compounds (VOCs) and nitrous oxides (NO_x) in the presence of sunlight (EPA 2011). The adverse effects on the human body are believed to occur because ozone molecules directly injure the epithelial surfaces onto which they are absorbed. This reaction results in injury to the respiratory system that is similar to that of the common sunburn. This can lead to lung inflammation which can result in reduced lung function, increased airway reactivity, and increased respiratory symptoms associated with chronic disease (WHO 2006). Numerous epidemiological studies have shown ozone to have harmful effects on the respiratory system along with a positive association with mortality (Samet JM 2000; Kim, Lee et al. 2004; Zanobetti and Schwartz 2008).

Nitrogen dioxide (NO_2), a reddish brown toxic gas that is highly reactive in the atmosphere, has drawn increasing concern in recent years as a notable constituent in urban air quality. Typically, atmospheric NO_2 is derived from two sources: as a direct emission (considered to be a primary pollutant) or from chemical reactions in the atmosphere (secondary pollutant). Several studies have identified automotive traffic to be the main contributor to ambient levels of NO_2 in urban environments (Aldrin and Haff 2005). The adverse health effects of NO_2 stem from its solubility in water as it transforms to nitric acid upon contact with the mucus membranes lining the respiratory tract causing irritation and inflammation (Hesterberg, Bunn et al. 2009). Aggravated asthma, bronchitis, and reduced lung function have been linked to NO_2 exposure (Hesterberg, Bunn et al. 2009).

In summary, air pollution is a problem for all of us as the air we breathe is constantly being polluted by emissions from anthropogenic and natural sources. Moreover, inhalation of these air pollutants has been shown to adversely impact the human body's respiratory, cardiovascular, and immune systems leading to increases in morbidity and mortality. Fortunately, human populations can influence their exposure to these environmental air pollutants by individual choices (e.g. drive a motor vehicle or use public transport), however, governments typically have more overall control as many (but not all) regulate air pollution to protect public health. This alone makes air pollution research a topic of significant importance.

1.2 PROBLEM STATEMENT

Over the coming century, it is anticipated that human settlements will be confronted by significant environmental changes linked with changes in climate. These changes in turn are likely to impact human health in many ways – mostly adversely (McMichael, Neira et al. 2008). One issue that tends to be overshadowed in the consideration of these impacts is air quality. Research has shown that variables important to air quality such as temperature, precipitation, circulation patterns, cloud cover and humidity are likely to change in the upcoming decades (DPCD 2008, EPA 2009). These changes may adversely impact air quality by: a) impeding the dispersion rates of pollutants; b) enhancing the chemical environment for pollutant generation; c) increasing the strength of emissions from the biosphere, fires, dust and allergens; and d) influencing anthropogenic emissions through adaptive responses involving increased fuel combustion (Dawson, Racherla et al. 2009; Jacob and Winner 2009). These findings are of concern to air quality management as the impacts of global climate change on regional air quality and associated health effects are largely unknown. In response to these concerns, the development of methods to understand the possible impacts of climatic change on air quality has been identified as a priority area for future research (HEI 2007; EPA 2009).

To further complicate matters, current knowledge about air quality is not perfect, and therefore current models are not perfect. Thus if we can't answer the question of how meteorology has influenced air pollution and health in the present, then we can't answer a more important question: How will changes in climate influence air pollution and air pollution related health outcomes? These issues are a concern for cities around the globe because providing clean air to breathe is a vital component in public health.

1.3 RESEARCH OBJECTIVES

Considering this, this thesis focuses on how meteorology has influenced air pollution and air pollution related health outcomes within the last decade in order to elucidate how important climate driven changes in weather may be for future air quality.

Because of the complex nature of air pollution meteorology, different weather elements (e.g., air masses, temperature, humidity, wind, and boundary layer height) can behave differently across urban environments and therefore interact in locally characteristic and distinct manners with air pollutants. For example, meteorology at the synoptic-scale not only influences regional air quality through the passage of variable air masses over a particular region but it also governs many local-scale meteorological features important to air quality. Conversely, local-scale meteorological features may supersede certain synoptic-scale features resulting in unexpected air quality outcomes. As such, pollutant-atmosphere interactions need to be understood at various scales (e.g., local, regional, synoptic, and global) in order to develop ways to reduce the negative impacts of a changing climate over a particular region of interest. Additionally, it is important to consider the role of meteorology in determining the magnitude of air pollution related health effects. This thesis attempts to integrate these issues by proposing the following research questions.

Given that meteorology at the synoptic-scale is important in understanding local air quality, the first research question of this thesis is:

1. How do air pollution levels in Melbourne vary under the presence of different types of large-scale air masses over south-eastern Australia, and which air masses result in the poorest air quality?

Because meteorology at the local-scale is also important to air quality and subject to the influence of climate change, the second research question is:

2. How do air pollution levels respond to changes in individual meteorological elements at the local-scale, and which element is the most important driver of air quality?

With such data, we can determine the sensitivity of air quality to changes in meteorology across large spatial scales and to individual weather elements. The results of these works lay a foundation for assessing the risk and management of a changing climate on air quality.

Air quality management decisions are not just based on air pollutant levels but are also concerned with the impacts on the exposed population. Therefore, a thorough local understanding of air pollution-related health effects is critical in protecting public health. One factor that is important to consider is that changes in weather can influence a city's population behavior and hence air pollution exposures within the city. Acknowledgement of this behavior has resulted in the identification of certain elements of weather – in particular temperature, as effect modifiers in air pollution-human health relationships. To improve knowledge of this issue, as there are direct implications of climate change on this as well, the third and final research question is:

3. How does temperature influence the air pollution-mortality relationship in Melbourne?

Currently, there is no information for Melbourne on how temperature (presumably due to climate change and the urban heat island effect) could modify the magnitude of air pollution effects on health. Therefore, the overall aim of this thesis is to assess the relationships between meteorology, air pollution, and human health for Melbourne, Australia, in order to provide insight into how these relationships may be altered under climate change**.**

1.4 APPROACH AND METHODOLOGY

Achieving the aim of this thesis requires the implementation and linkage of three separate, yet tiered, research components. Conceptually, the completion of a component in the series informed decisions regarding the design of the next component. Each component addresses on of the three thesis questions and is presented as individual papers. The detailed approach and methodology is found in each paper. The first thesis question involved an examination of the relationship between synoptic-scale circulation patterns and air pollution that helped define and understand the importance of changes in large-scale circulation features on air quality. The objectives were; (1) determine the magnitude in which select air pollutants responded to large-scale atmospheric features, (2) identify which circulations were the most important in regards to poorer air quality, and (3) characterize the underlying local weather conditions under each circulation feature. Addressing these objectives in turn helped determine the meteorological elements assessed in the second thesis question – an examination of the relationship between local-scale meteorology and air quality.

This component was implemented in order to expand upon the first by taking a closer look at individual meteorological elements and their influence on air pollutant concentrations. The aim here was to identify the nature and magnitude in which each individual meteorological element affected local air pollution. Moreover, effort was made to determine the relative importance of each individual element on air quality. The final objective of the second component was to make clear which meteorological element had the most influence on air pollution. The third and final research component involved an epidemiological assessment of the impact of short-term air pollution exposures on mortality under different meteorological conditions. The aim was to identify if separate concentration response functions for the air pollution-mortality relationship existed under different ranges of temperature. The aim here was to provide a clear window into the role of meteorology on the air pollutant-human health relationship.

Overall, the approach and methodology presented here provides a means to results that provide a solid foundation into the nature and importance of meteorology as a driver of local air pollution and its role in determining health outcomes. This information is critical in understanding how changes in climate may affect levels of air pollution as well as their effects on the local population.

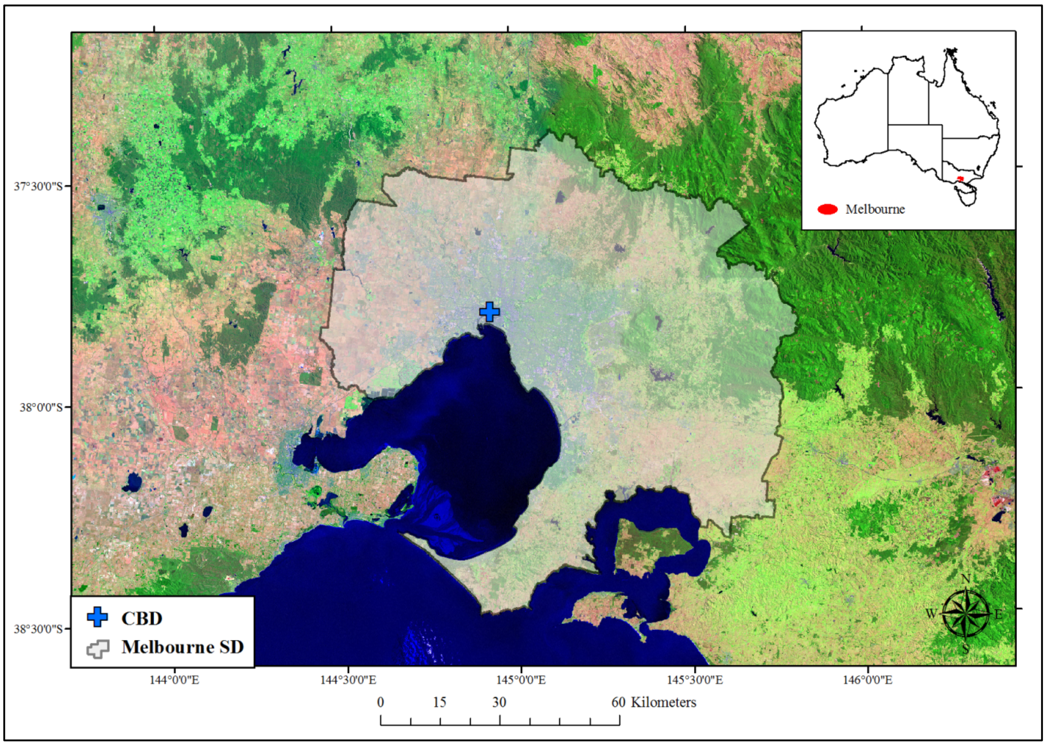
1.5 A CASE STUDY FOR MELBOURNE, AUSTRALIA

Melbourne, a city with a population of nearly four million, lies on Port Phillip Bay at the southern edge of Victoria between intercontinental Australia to the north and the Southern Ocean to the south (Figure 1.1). Founded in 1835, the city has a history of immigration with the largest period of growth occurring during the Victorian gold rush of the 1850s. Today, the city is often referred to as the ‘cultural capital of Australia’ and is consistently ranked one of the top cities in the world in which to live. This is based on criteria such as: business conditions, access to goods, safety, climate, quality of architecture, access to green space and nature, tolerance, and effectiveness of infrastructure. In this brief examination of Melbourne, I offer a sense of what the future of environmental risk holds by looking at trends in air quality, population, motor vehicles, and climate. The evidence presented in the summaries that follow emphasize the importance for research on air quality and subsequent health effects in the region.

Air quality in Melbourne can be described as relatively good when compared to other urban centers of similar size and despite increasing pressures from population growth and increases in the numbers of motor vehicles, air quality has remained steady over recent years (EPAVIC 2009). Of course, this may change as population increases and urban consolidation occurs. However, epidemiological research continues to identify that air pollution adversely affects the local population (Erbas, Kelly et al. 2005; Simpson, Williams et al. 2005; Dennekamp, Akram et al. 2010). These results are a sign that improvements can still be made. At present, the majority of pollutant emissions in the region are generated by motor vehicles followed by industry and the domestic/commercial sectors (EPAVIC 2009). Additional contributions are also being made by bushfires and airborne dust, which have been strongly associated with air pollution exceedances across the region (EPAVIC 2009). Certain meteorological conditions have also been shown to pose a threat to air quality. For example, periods of stability have been shown to result in the build-up of pollutants; the occurrence of a local circulation known as the ‘Melbourne eddy’ can recirculate pollutants around the bay; and strong windy conditions can transport dust from central Australia (Hurley, Manins et al. 2003; Chan, Cohen et al. 2008; EPAVIC 2009). These facts indicate that air quality in Melbourne is sensitive to meteorological drivers.

Melbourne’s population, which grew by 93,500 people in 2008-2009, continues to expand rapidly as it outpaced all other Australian capital cities during this time

(ABS 2010). This growth is anticipated to continue as population projections for the city estimate numbers in the range of six to eight million by 2056 (ABS 2008). This will likely result in dramatic changes across the current landscape as expansions to infrastructure will need to be made. Fortunately, present day Melbourne is a very dispersed, low-density city and therefore has the potential to accommodate smart growth (Barter, Kenworthy et al. 2003). However, aspects of this growth – particularly the ageing of the population, will present additional challenges for the city. At the time of the 2006 Census, children aged between 0 to 14 years comprised 18.6% of the population and persons aged 65 years and over accounted for 13.0% (ABS 2006a). By

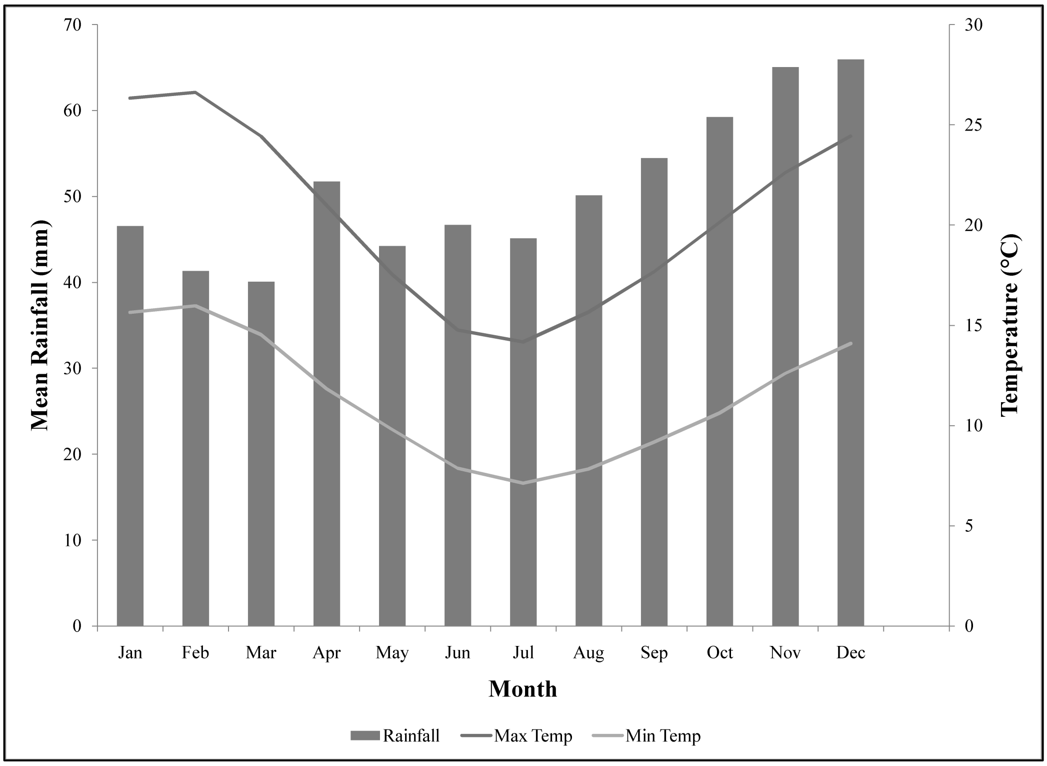


2036, the population of the elderly is anticipated to double (DPCD 2008). This is important as evidence indicates that the elderly require more medical care and are more sensitive to surrounding environmental conditions (Schwartz 2000; Stafoggia, Forastiere et al. 2006; Nicholls 2009).

Figure 1.1: A map of Melbourne, Australia showing the study region. CBD: Central Business District; SD: Statistical Division

Another aspect of population growth that can be detrimental to the health of the local environment is the coincident growth of motor vehicles. In Melbourne, it is known that motor vehicles are the largest emitter of air pollution (EPAVIC 2009) and any growth in this area is likely to increase the potential for poorer air quality. Between the years of 2005 to 2010, car ownership in the state of Victoria increased 12.6% ending with a fleet of 4.1 million registered vehicles (ABS 2011). Growth in the vehicle fleet has coincided with the reduction of public transport use across the city as methods of travel to work in Melbourne noted that 67% of commutes were performed using motor vehicles and 10% were accomplished using public transport (ABS 2006b). However, future plans for Melbourne aim to increase public transport ridership to 20% by 2020 (DSE 2002). Finally, annual distances travelled by Melbourne passenger vehicles continue to increase along with growth in the outer suburbs (DOT 2009). Current trends suggest (EPA2009) that emissions from motor vehicles will likely continue to increase in the future due to increased numbers of motor vehicles on the road making even longer trips.

The climate of Melbourne can best be described as having moderate temperatures and relatively low precipitation and falls under the Köppen climate classification *Cfb* - which is ‘oceanic’ (Figure 1.2). Nevertheless, periods of extreme heat do occur as a daily maximum of 46.4 °C was recorded on 7 February 2009 and the region averages over ten days per year above 35 °C (BOM 2010). This is due in part to the city's location at the pole-ward margin of a sub-tropical continent that results in the passage of very differing air masses over the region. Air masses originating from continental Australia tend to be hot and dry while air masses from the southern ocean tend to be cool and wet. While it is clear that solar radiation is the dominant driver of temperature in Melbourne (Nicholls, Uotila et al. 2010), other



factors such as the 'Melbourne eddy' and an urban heat island effect also play important roles.

Figure 1.2: Climatogram for Melbourne, Australia using Bureau of Meteorology data over the period of 1981 to 2010.

According to the United States National Oceanic and Atmospheric Administration (NOAA), the world warmed by approximately 0.7 °C in the 20th century with every year in this century being warmer than all but 1998 in the last (NOAA 2011). Moreover, 2010 tied with 2005 for the warmest year on record since 1880 (NOAA 2011). Climate scientists anticipate a future with higher rainfall variability, greater frequency of extreme events, sea-level rise, ocean acidification, and long-term shifts in temperature and precipitation (IPCC 2007; IPCC 2007b). These findings can be rather alarming as any of these changes can profoundly disrupt local environments that supply our basic needs.

In south-eastern Australia, attention on climate change has largely focused on rainfall patterns and temperature (SEACI 2010). In light of these concerns, various techniques have been used to estimate future climate in south-eastern Australia under scenarios presented by the output of global climate models. Statistical methods have been used to relate the large-scale model output to local climate variables and dynamical methods use the global model output to drive finer-scale models. All techniques indicate a warmer, drier climate in the future (SEACI 2010). Climatologists suggest that, apart from global climate trends, the driver of this transition in climate for south-eastern Australia is the observed intensification of high atmospheric pressure cells across the region, a scenario that can only be reproduced by models that include anthropogenic greenhouse gases (SEACI 2010). However, the precise magnitude, direction, and timing of the changes to come are still an area of uncertainty both regionally and globally. Due to this uncertainty, we need to be prepared for a range of future conditions that include the best and worst case scenarios. Therefore, further analyses on the expected impacts of such changes should be explored.

In summary, the increasing trends in population and vehicle ownership in Melbourne suggest that air pollution emissions will likely increase in the future. Furthermore, projected changes in climate indicate a future that will be warmer, drier, and more atmospherically stable. If these arguments are right, then policy makers need to differentiate the extent and magnitude in which these factors affect air quality and health in order to ensure better living environment in the future.

1.6 STRUCTURE OF THESIS

This thesis is a “thesis by publication” and the School of Geography and Environmental Science at Monash University determined the structure. In accordance with Faculty requirements, Chapter 1 presents a framing of the thesis research along with pertinent background information. Chapter 2 continues to expand the framing of the work by presenting a current literature review of the topics relevant to this thesis. Chapter 3 presents a manuscript accepted by *Atmospheric Environment* titled "Investigating the influence of synoptic-scale circulation on air quality using self-organizing maps and generalized additive modeling." Chapter 4 presents a manuscript accepted by *Atmospheric Environment* titled "Investigating the influence of local meteorology on air quality using generalized additive models." Chapter 5 presents a working manuscript titled "The influence of temperature on the air pollution-mortality relationship in Melbourne, Australia". Chapter 6 presents a conclusion chapter. Finally, a bibliography is presented followed by an appendix that includes the statistical coding used for Chapters 3 to 5.

CHAPTER 2

LITERATURE REVIEW

2.1 INTRODUCTION

The pathway of air pollution in the environment is directly influenced by chemical and physical atmospheric processes (Figure 2.1) thus making air pollution an area sensitive to weather and consequently changes in climate. To further complicate matters, air pollution-related health effects are also subject to change under a shifting climate as changes in weather may influence the risk associated with air pollution exposure. In south-eastern Australia, this is an area of growing concern because the future climate is expected to be warmer, drier, and more stagnant, due to overall global warming, a weaker global circulation, and a decreasing frequency of mid-latitude cyclones over the area. How will these projected changes influence air quality and related health outcomes over the region?

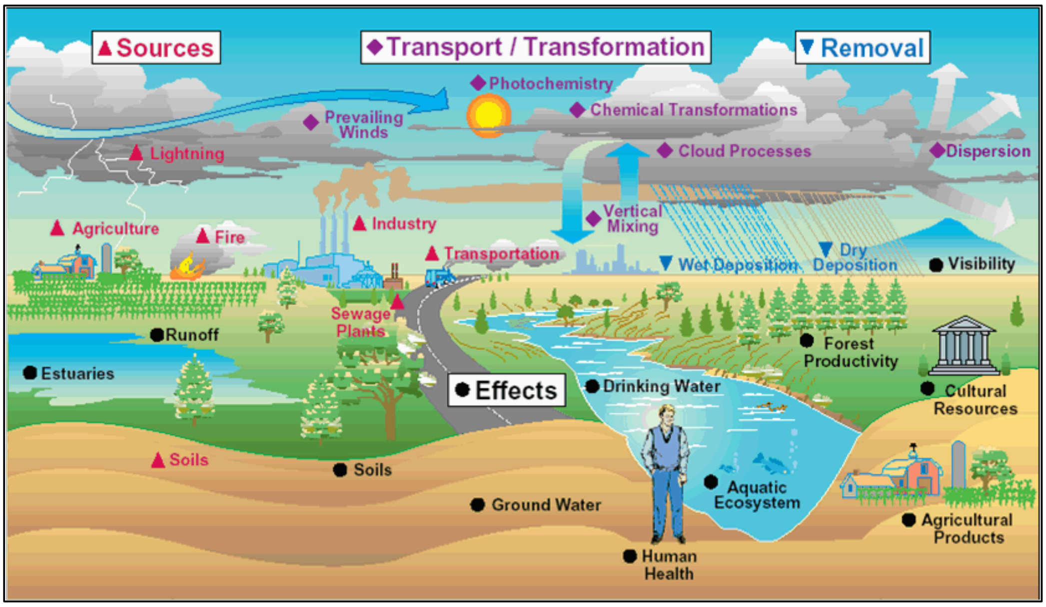


Figure 2.1: The interrelationships among air pollutants, emission sources, transport and transformation pathways, and environmental effects are complex. (Source: United States Environmental Protection Agency (EPA) report on national air quality trends found at http://www.epa.gov/airtrends/2007/report/highlights.pdf and accessed on 14 July 2011).

Answering this question is more complicated than it may seem as climatological influence on air quality and subsequent human health is complex (Figure 2.2). One can see that the specific type of climactic change, the direction of that change, and the magnitude of that change over a particular location is dependent upon many factors and thus the response of air quality will vary accordingly. A growing number of investigations have begun research on the anticipated 'climate effect' for air pollution using a variety of different approaches. Meaningful results have been provided by (a) performing statistical analysis to estimate the response of air pollution observations with meteorological variables, (b) using deterministic models to perform perturbation studies, and (c) using deterministic models to run simulations of future air quality. However, much is still unknown.

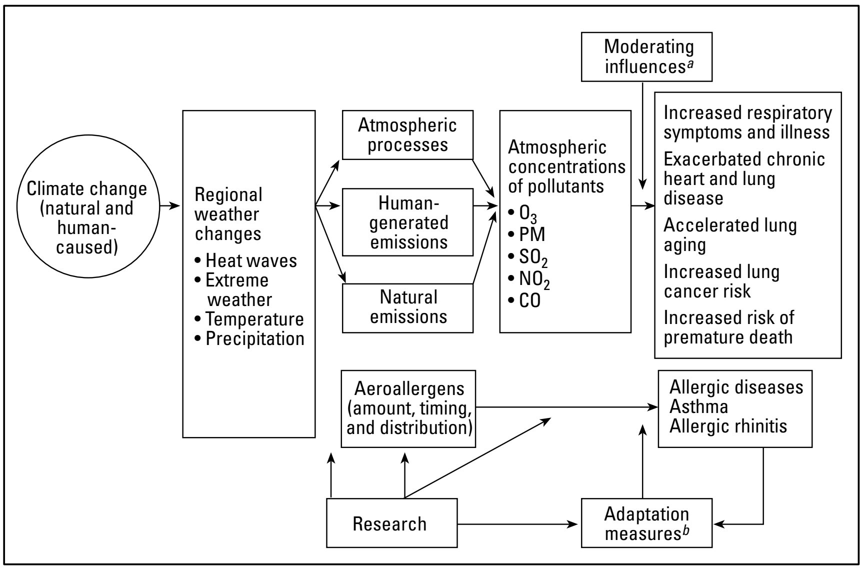


Figure 2.2. A simple diagram highlighting the potential effect of climate change on processes that influence air pollution and related health effects from Bernard et al. (2001). [a]Moderating influences include nonclimate factors that affect climate-related health outcomes, such as population growth and demographic change, standards of living, access to health care, improvements in health care, and public health infrastructure. [b]Adaptation measures include actions to reduce risks of adverse health outcomes, such as emission control programs, use of weather forecasts to predict air quality levels, development of air quality advisory systems, and public education.

In addition to the growing concern of changes in pollutant levels due to climate change are the impacts of this change on air pollution-related health effects. It is well known that air pollution has adverse effects on human populations. Unfortunately, climate change may alter these relationships by (a) affecting human exposure patterns;

(b) changing the mixture of pollutants in the environment; and (c) modifying the environment in which exposure occurs. Any of these changes could manipulate the nature in which air pollution affects local populations.

The structure of this literature review is as follows: First, a detailed discussion of the air pollutants of concern – ozone (O_3), particulate matter (PM), and nitrogen dioxide (NO_2) is provided. For reference, a brief discussion on global air quality regulation is also presented. In the second section of the review, present understanding of the role of meteorology on air pollution is provided. In this section, focus is given to meteorology at two scales deemed important to local air quality – *synoptic*-scale and *local*-scale. Moreover, presentation of studies that examine the sensitivity of air pollution to meteorological elements is also offered. In the third section of the review, focus is shifted to current understanding of how air pollution affects the health of urban populations around the globe. Within this section, a subsection on the potential of weather as an effect modifier of air pollution-health relationship is also presented. The next topic addressed is the impact of climate change on air quality. This section reviews studies covering global to regional scale predictions of climate impacts on air quality. The closing section of the review synthesizes each of the preceding sections, presents limitations and knowledge gaps, and provides discussion on how this thesis addressed some of these issues.

2.2 BACKGROUND ON POLLUTANTS OF CONCERN

Ozone

Ozone (O_3) is a gaseous molecule consisting of three oxygen atoms. Its formation in the lower atmosphere occurs by photochemical oxidation of carbon monoxide (CO), methane (CH_4), or non-methane volatile organic compounds (NMVOCs) by the hydroxyl radical (OH) in the presence sunlight (*hv*) and reactive nitrogen oxides ($NO_x = NO + NO_2$) (Hewitt and Jackson 2003).Similar reactions occur with CH_4 and VOCs but are far more complex. For a detailed discussion please see Hewitt and Jackson (2003).

Combustion is primarily responsible for VOCs, CO, and NO_x in the troposphere (Jacob and Winner 2009). Even so, vegetation is also an important VOC source. The atmospheric oxidation of water vapor produces most OH, which typically cycles in the atmosphere with other hydrogen oxides (HO_x). Outside the availability of sunlight, the production of ozone in the urban environment has been found to be limited by the supply of VOCs and NO_x (Jacob and Winner 2009). Ozone is primarily removed from the troposphere by photolysis in the presence of water vapor; however, uptake by vegetation can also make notable contributions. Surprisingly, wet deposition has little effect as ozone and its major precursors have low solubility in water. The atmospheric lifetime of ozone ranges from a few days in the boundary layer to weeks in the free troposphere.

Ozone and other photochemical oxidants injure the epithelial surfaces onto which they are adsorbed (Bernard, Samet et al. 2001). Experimental animal and *in vitro* studies have shown increased permeability and inflammation of airways; morphologic, biochemical, and functional changes; and decreased host defense functioning because of acute ozone exposure (Bernard, Samet et al. 2001). *In vitro* studies using very high concentrations of O_3 (> 500 ppb) suggest that O_3 has a low potential to cause mutagenic, cytogenic, or cellular transformation effects (Bernard, Samet et al. 2001). Thus, the health effects of concern relate primarily to lung inflammation, with clinical manifestations arising from direct effects on the lung and possibly indirect effects arising from systemic consequences of lung inflammation and mediator release.

Particulate Matter

Particulate matter (PM) is a general term describing air pollution consisting of a mixture of particles that can be solid, liquid, or both, suspended in the air and representing a complex mixture of organic and inorganic substances (Hewitt and Jackson 2003). The size and shape of the particles in the atmosphere typically range from a spherical diameter of 0.0001 μm to 100 μm, with categorizations being imposed to refer to the respective size as being coarse (2.5-10 μm), fine (0-2.5 μm), ultrafine (0.01-0.1 μm), and nucleation (< 10 nm). Principal components of PM include sulfate, nitrate, organic carbon, elemental carbon, soil dust, and sea salt (Chan, Cohen et al. 2008). The first four components are mostly present in fine particulate matter (Jacob and Winner 2009). Sulfate, nitrate, and organic carbon are produced within the atmosphere by oxidation of SO_2, NO_x, and NMVOCs; however, carbon particles are also emitted directly by combustion. Nitrate and organic carbon can exchange between the particle and gas phases, depending - in particular, on temperature (Dawson, Adams et al. 2007b). Unlike ozone, seasonality of PM is complex and location-dependent, thus, PM is typically viewed as an air quality problem year round. PM is efficiently scavenged by precipitation and this is its main atmospheric sink, resulting in atmospheric lifetimes of a few days in the boundary layer and a few weeks in the free troposphere (Jacob and Winner 2009). Again differing from ozone, background PM in the free troposphere is typically not a substantial contributor to surface air quality. Nevertheless, exceptions are plumes from large dust storms and forest fires, which can be transported on intercontinental scales.

The potential mechanisms linking inhaled particles to acute cardiopulmonary consequences are still uncertain; however, hypotheses have been offered concerning lung inflammation and cytokine release (Bernard, Samet et al. 2001). Similar to the response of ozone, it is believed that exposure activates stress signaling pathways from the epithelium to the lung microvessels - which may increase blood clotting. Additionally, increased concentration of fibrinogen and platelets, and sequestration of red blood cells in the lung mass have also been linked to particulate pollution (Bargagli, Olivieri et al. 2009). At the present, diesel particulates have become an area of increasing concern. This is because they have been shown to increase the synthesis of the allergic antibody IgE in animals and human beings, which likely increases sensitization to common allergens (Brown, Graham et al. 2007). Furthermore, many epidemiologic studies link breathing PM to a series of significant health problems, including: aggravated asthma, increases in respiratory symptoms like coughing and difficult or painful breathing, chronic bronchitis, decreased lung function, acute myocardial infarction, and premature death (Pope and Dockery 2006).

Nitrogen Dioxide

Nitrogen dioxide (NO_2) is a reddish brown toxic gas that is highly reactive in the atmosphere. It is tightly coupled with nitric oxide (NO) in the sunlit hours and the two rapidly interconvert with one another in the presence of ozone. Therefore, the term nitrous oxides (NO_x) is commonly used in discussions for both pollutants. For most anthropogenic sources, NO_x is emitted in the form of NO. For natural sources NO is emitted from soil processes and lightning discharge. During the high temperatures inside an internal combustion engine the heat/energy released initializes a reaction between nitrogen (N_2) and oxygen (O_2) to form nitric oxide (NO), which in the presence of O_3 is then oxidized in the air to form nitrogen dioxide (NO_2) (Hewitt and Jackson 2009). In the presence of sunlight NO_2 then transforms back into NO and O. It is important to note that these reactions cycle back and forth in a matter of seconds, giving NO and NO_2 extremely short atmospheric lifetimes. However, this is somewhat misleading as considering the two compounds together as NO_x gives a much longer lifetime of several hours. In the urban environment, NO_x is part of the complex mixture of primary and secondary pollutants associated with fossil fuel emissions as studies have identified automotive traffic to be the main contributor (Aldrin and Haff 2005). Removal of NO_x from the atmosphere occurs via dry deposition and through a reaction with OH to form HNO_3.

The adverse health effects of NO_2 stem from its solubility in water as it transforms to nitric acid upon contact with the mucus membranes lining the respiratory tract causing irritation and inflammation (Hesterberg, Bunn et al. 2009). Aggravated asthma, bronchitis, and reduced lung function have been linked to NO_2 exposure (Erbas, Kelly et al. 2005). However, characterizing direct effects of the pollutant on human populations has proven difficult. Therefore, the contribution of NO_2 to secondary particles and its role in the formation of O_3 may be more relevant to public health than any direct effects (Bernard, Samet et al. 2001). It is important to note that controversy exists over the direct health effects of NO_2, as some suggest an effect of NO_2 is an indicator of the typical urban pollutant mixture rather than implicating the gas itself (Vogel and Riemer 1999).Current Regulation and Management

As it is well known the pollutants discussed above are risks to human health, governing bodies around the world are challenged by developing strategies to reduce their impact on human health. This is largely accomplished via the imposition of regulatory standards as means to control their atmospheric concentrations. The World Health Organization (WHO) develops global air quality guidelines in an effort to challenge governments to improve air quality in their cities and thus protect human health. In the United States, the Environmental Protection Agency (EPA) is charged with managing air quality and is therefore responsible for regulating industrial emissions and imposing standards on the manufacturing of automobiles to insure that the air is safe. Across the European Union, the European Commission Environment (ECE) is responsible for similar tasks, and in Australia, these responsibilities fall on the National Environment Protection Council (NEPC). While much has been accomplished since the days before regulation, air pollution is still a problem around the globe. Current guidelines, regulations and standards are presented in Table 2.1.

Table 2.1: Current ambient air quality regulatory guidelines/standards from the World Health Organization (WHO), the United States Environmental Protection Agency (EPA), the European Union (ECE), and the Australian National Environment and Protection Council (NEPC).

		Averaging Period				
Pollutant	Regulatory Body	*1-h*	*4-h*	*8-h*	*24-h*	*1 year*
O_3 (ppb)	WHO	--	--	50	--	--
	EPA	120	--	75	--	--
	ECE	--	--	60	--	--
	NEPC	100	80	--	--	--
PM_{10} ($\mu g/m^3$)	WHO	--	--	--	50	20
	EPA	--	--	--	150	--
	ECE	--	--	--	50	40
	NEPC	--	--	--	50	--
$PM_{2.5}$ ($\mu g/m^3$)	WHO	--	--	--	25	10
	EPA	--	--	--	35	15
	ECE	--	--	--	--	25
	NEPC	--	--	--	25	8
NO_2 (ppb)	WHO	104	--	--	--	22
	EPA	100	--	--	--	53
	ECE	104	--	--	--	22
	NEPC	120	--	--	--	30

WHO: http://www.who.int/phe/health_topics/outdoorair_aqg/en/index.html;
EPA: http://www.epa.gov/air/criteria.html;
ECE: http://ec.europa.eu/environment/air/quality/standards.htm;
NEPC: http://www.ephc.gov.au/taxonomy/term/23

2.3 WEATHER AND AIR POLLUTION

Weather can be defined as the state of the atmosphere as it is in a particular place at a particular time or over a brief period, with special emphasis on short-term changes (American Meteorological Society 2011). A typical description of weather, as described by the field of meteorology, includes references to elements such as temperature, air pressure, humidity, visibility, clouds, wind, and precipitation over a period of minutes to days. Climate on the other hand, is a much broader concept and can be defined as the description of the variability of weather conditions over a particular region or latitude zone over a specific period of a month or greater. In the following, a review the effect of *weather* on air pollution is presented.

It is well known in the air quality community that concentrations of gases and particles are influenced by weather (Elminir 2005). Research has shown that there are two spatial scales of weather that are of great consequence – the *synoptic-scale* and the *local-scale*. Synoptic-scale systems encompass weather phenomena operating over large-scale regions (thousands of kilometers), which includes migrating cyclones and anticyclones, air masses, and fronts (American Meteorological Society 2011). Local-scale systems encompass weather over a particular region (e.g., city) and are typically discussed using daily measures of elements such as temperature, wind, pressure, precipitation, humidity, and the atmospheric boundary layer. At present, much is understood regarding the link between meteorology and air quality; however, more can still be learned.

Synoptic-scale air pollution meteorology

Synoptic-scale weather events are commonly described through assessments of pressure systems over a given area. Large-scale pressure variations are used because they correspond well with atmospheric circulation features (Huth, Beck et al. 2008). Circulation features at this scale are typically classified into two types: low-pressure systems and high-pressure systems (Sturman and Tapper 2006).

Low-pressure systems are often associated with cloudy skies, stormy weather, and fast surface winds. Theoretically, these types of pressure systems result in the dispersion of near surface pollution through horizontal and upward vertical transport (Hewitt and Jackson 2009). Additionally, clouds are believed to block sunlight that would otherwise drive photochemical reactions, reducing pollution further.

On the other hand, conditions under high-pressure systems are quite different; as they are characterized by relatively light surface winds, sinking air, and clear skies. These conditions are believed to be conducive to pollution build up because: sinking air can confine near surface pollution, slow near surface winds prevent horizontal dispersion of pollutants, and clear skies maximize sunlight available to drive photochemical reactions (Hewitt and Jackson 2009).

Influence of synoptic-scale circulation on air pollution

The influence of synoptic-scale meteorology on air pollutant concentrations has been an active area of study for several decades. Aims of the research have been to: (1) associate large-scale features with air pollution episodes, (2) improve air quality forecasts, and (3) gain insight into the processes affecting pollutant concentrations. They are useful for our purpose as an observational basis for understanding how sensitive air pollution is to changes at the large-scale – a potential outcome of climate change.

A notable study in the United States used synoptic climatology to demonstrate the relationships between atmospheric circulation and ozone over the Pittsburgh metropolitan area (Comrie and Yarnal 1992). Comrie and Yarnal (1992) found that high ozone events were associated with a slow moving anticyclone and that low ozone events occurred under cool, cloudy cyclonic conditions. In the United Kingdom, synoptic typing using principal component analysis and cluster analysis defined air mass types in order to relate them to several air pollutants across Birmingham (McGregor and Bamzelis 1995). McGregor and Bamzelis (1995) found that continental anticyclonic air masses were associated with high pollution events and that maritime cyclonic air masses were associated with low pollution concentrations. In China, similar results were found as an increasing air pollution index (API) was associated with high-pressure systems and a decreasing index was noted for low-pressure systems.

At present, only a few studies have looked at the associations between large-scale atmospheric processes and air pollution in south-eastern Australia. In Sydney, the relationship between synoptic climatology and ozone events in Sydney was assessed to identify if the pollution decreases seen in the late 1980's and early 1990's were a function of emission controls or synoptic situations (Leighton and Spark 1997). Correlations between anticyclonicity with moderate and high ozone days for winter and summer from 1978 to 1992 were found. More specifically, the authors noted that a northwesterly wind gradient (associated with the backside of a passing anticyclone) was the most common feature associated with medium and high levels of recorded air pollution and that anticyclones centered close to Sydney with a high degree of immobility resulted in significant pollution episodes.

In a more recent study, Hart, De Dear et al. (2006) investigated the meteorological features of ozone episodes in Sydney from 1992 to 2001. Eleven synoptic classes were identified using a combination of principal component analysis and cluster analysis for Sydney during the warm months (October-March) that were compared to exceedence in ozone. Over 90% of all episode days were associated with a high-pressure system being located in the middle to eastern Tasman Sea. This system was characterized as having light northwesterly gradient winds, an afternoon sea breeze, high afternoon temperatures, and a shallow mixing height along the coast and warming aloft during the day.

Local-scale air pollution meteorology

Local-scale meteorological effects on air pollution are typically described through the measured association or sensitivity of air pollutant concentrations to individual weather elements. These elements include temperature, winds, boundary layer characteristics, humidity, precipitation, and solar radiation.

Temperature, the representation of molecular kinetic energy, is important to air pollution meteorology for numerous reasons (Hewitt and Jackson 2009). First, surface temperatures are the primary driver of boundary layer base heights. This is important for air pollution because as high surface temperatures result in high boundary layer base heights thus leading to high mixing depths and consequently low pollution mixing ratios. Conversely, cold surface temperatures produce thin mixing depths and high pollution mixing ratios. Second, surface temperatures can have a strong influence on wind speeds. For example, a warm surface temperature enhances convection – vertical air circulation due to cool air sinking and warm air rising, which increases vertical mixing and leads to stronger surface winds. Faster near surface winds can result in greater dispersion of near surface pollutants or increase the re-suspension of loose soil dust and other aerosol particles from the ground (Jacobson 2002). On the other hand, cooler surface temperatures have the opposite effect; slowing down near surface winds and enhancing near surface pollution build up. Finally, surface temperatures are often correlated with levels of solar radiation – the primary driver of photochemistry.

In addition to controlling aspects of meteorology important to air pollution, temperature is also a primary driver of the rates of several processes important to air pollution formation. For example, near surface air temperatures can mechanistically drive air pollution by affecting the rates of chemical reactions, anthropogenic volatile organic compound (VOC) emissions, biogenic emissions, and aerosol thermodynamics (EPA 2009). More specifically, secondary pollutants - including ozone (O_3) and other photochemical oxidants and particulate sulfate, nitrate, ammonium, and secondary organic aerosols (SOA) - are formed in the ambient atmosphere via chemical reactions that are temperature sensitive (Ying and Kleeman 2003; Dawson, Racherla et al. 2009).

Moreover, motor vehicle related emissions have been found to increase due to enhanced evaporation of VOCs at higher temperatures and rates of biogenic gas emissions from vegetation have also been found to increase exponentially to increases in temperature (EPA 2009). Furthermore, temperature influences aerosol thermodynamics by affecting gas-to-particle partitioning as saturation vapor pressure - the capacity of air to hold vapors of a trace gas, increases with increasing temperature (Ying and Kleeman 2003).

Winds are important to air pollution meteorology because they can influence the build-up, transport, dispersion, or generation of air pollution. By definition, winds are the motion of air through the atmosphere that arises due to pressure gradients and variations in surface temperature (Jacobson 2002). They are notoriously complex as they are affected by large-scale pressure gradients, local pressure gradients, variable topography, and local turbulence. At the local-scale, important circulation features are often associated with a body of water (e.g., ocean, lake, and river). Take Melbourne for example, sea-breeze circulation features have been shown to form elevated layers of pollution by lifting and injecting polluted air into the inversion layer during its return flow to the ocean (Tory, Cope et al. 2004). An example of topographical importance is mountain breezes, which result from differences in friction and pressure and influence air pollution by forcing pollutants either up or down slopes or trapping them. Overall, high winds are typically associated with ventilated conditions and disperse air pollution near source areas. However, strong winds can also enhance the transport of polluted air and create important air quality events such as dust storms (Hewitt and Jackson 2009) and light winds are typically associated with stable conditions which have been found to result in increased pollutant concentrations (Ning and Sioutas 2010).

Another important meteorological determinant of air quality is characteristics of the planetary boundary layer (PBL). The PBL is the lowest layer in the atmosphere, usually taken up to a height of 1 to 2 kilometers, in which meteorological conditions are significantly affected by the Earth's surface (Australian Bureau of Meteorology 2010). As previously mentioned, the mixing conditions in the boundary layer are strongly linked to both synoptic-scale pressure systems and local temperature. The development of the mixing layer is an important controlling factor for air pollution episodes as it controls the vertical mixing of pollutants and hence dispersion in the upper atmosphere (Dawson, Racherla et al. 2009). In short, the higher the mixing depth, the more vertical mixing, and thus dispersion of air pollution that can take place. Another important feature of the boundary layer that specifically affects air pollution is capping inversions. A capping inversion is a statically stable layer at the top of the PBL that can be induced in many ways and results in the trapping of air at the surface therefore suppressing vertical mixing (American Meteorological Society 2011). Worst case scenarios occur when a strong inversion with high temperatures effectively creates a closed, heated reaction vessel that amplifies the photochemical production of secondary pollutants (EPA 2009).

Humidity corresponds to the concentration of water vapor in the atmosphere. The influence of humidity on air pollution is largely dependent on the nature of the pollutant's reaction with water. This has led to conflicting responses of air pollutants to increasing humidity. For example, ozone formation has been shown to be suppressed when humidity increases; however, formation of some secondary particle constituents has been shown to increase with humidity (Liao, Chen et al. 2006; EPA 2009).

Precipitation can be defined as any liquid or solid phase aqueous particles that originate in the atmosphere and fall to the Earth's surface (American Meteorological Society 2011). Common forms of precipitation include hail, rain, sleet, and snow. It is important to air pollution because of its role in the removal of pollutants from the atmosphere through wet deposition. In short, increasing precipitation leads to decreasing concentrations of pollutants in the atmosphere (EPA 2009).

The final meteorological element presented in this discussion is solar radiation. Solar radiation, the electromagnetic radiation emitted by the Sun, is primarily important to air pollutant production due to its role in photochemistry. A primary example is the positive response of ozone to increasing solar radiation due to the increase in photochemical oxidation rates (Dawson, Adams et al. 2007a). Outside of the seasonal variation over any given area, the day-to-day influence of solar radiation is strongly associated with changing cloud distributions. Therefore, increasing cloud cover has been shown to decrease photochemical oxidation rates therefore resulting in reductions of pollutants such as ozone (EPA 2009).

Sensitivity of air pollutants to local meteorological variables

Recent acknowledgement of the sensitivity of air quality to changes in climate has revitalized the importance of understanding air pollution-weather relationships. Traditionally, the purposes for this area of study have been: (1) to construct empirical/dynamical models for air quality forecasts, (2) to evaluate the effectiveness of emission controls, and (3) to further understanding of the processes affecting air pollution. However, in light of new concerns, a fourth purpose – to understand the sensitivity of air pollutants to changes in meteorology, has been introduced.

At present, two main approaches have been used to investigate the fourth purpose – the statistical modeling of air pollutants using meteorological variables and the perturbation of meteorological scenarios in dynamical models. The first approach uses observed values to estimate direct relationships without any assumptions. The second approach takes place within the constraints of modeling environments but is useful in identifying outcomes related to changes in specific processes.

In the United States, Camalier, Cox et al. (2007) used a generalized linear model (GLM) to characterize the relationship between ozone and meteorology for 39 major urban areas. The model was found to perform very well, yielding R^2 statistics as high as 0.80. The results provide strong evidence that ozone is generally increasing with increasing temperature and decreasing with increasing relative humidity. Moreover, examination of the spatial gradients of these responses identified that the effect of temperature on ozone was most pronounced in the north while the opposite was true of relative humidity. This is important because this finding indicates that the sensitivity to particular meteorological variables is variable across geography and climate.

A study focusing on particulate matter applied a multiple linear regression (MLR) to assess the correlation between $PM_{2.5}$ and its components with meteorological elements across the contiguous United States (Tai, Mickley et al. 2010). They found that up to 50% of the variability in $PM_{2.5}$ could be explained by temperature, relative humidity, precipitation, and circulation. Moreover, temperature was found to have positive associations with the sulfate, organic carbon (OC), and elemental carbon (EC) components of PM almost everywhere. However, heterogeneous trends were found for nitrate. As expected, precipitation exhibited a strong negative relationship with all components and periods of atmospheric stability/stagnation resulted in increases of PM.

In the United Kingdom, Carslaw, Beevers et al. (2007) use a generalized additive model (GAM) to explain daily concentrations of nitrogen oxides (NO_x), nitrogen dioxide (NO_2), carbon monoxide (CO), benzene, and 1, 3-butadiene using several meteorological and road traffic covariates. The results show that localized wind-flow patterns have a large influence on the model predictions (particularly NO_x and NO_2). Moreover, NO_x was shown to decline with increasing temperature; however, the opposite was reported for NO_2. Traffic was shown to be important for all pollutants.

A similar approach was taken in Oslo, Norway, to model air pollutant concentrations using measures of traffic volume and meteorological variables. Separate models were estimated for the concentration of PM_{10}, $PM_{2.5}$, the difference PM_{10}– $PM_{2.5}$, NO_2, and NO_x using the period 2001 to 2003. Results found that the most important predictor variables for air pollution were related to traffic volume and wind. Furthermore, relative humidity demonstrated a clear effect on the PM variables, but not on the NO variables. Other predictor variables, such as temperature, precipitation and snow cover on the ground were found to be of some importance for one or more of the pollutants, but their effects were less pronounced.

In form with the second approach discussed above, Dawson, Adams et al. (2007a; 2007b) examine the individual effects of various meteorological parameters on O_3 and $PM_{2.5}$ concentrations in the Eastern US using a chemical transport model in order to indicate that changes in climate could significantly affect O_3 and PM. The largest meteorological impact on ozone was attributed to temperature where an increase of 1 degree Kelvin resulted in a 0.34 ppb increase. Absolute humidity was also found to have a positive effect, where a 1% increase resulted in a 0.025 ppb increase in ozone. Rather small effects were seen for wind speed, mixing height, clouds, and optical depth. For PM, the study found that the strongest effects were seen due to temperature, wind speed, absolute humidity, mixing height, and precipitation. Wind speed, mixing height and precipitation affected all PM species while temperature displayed results that are more heterogeneous. For example, temperature increased average sulfate concentrations and decreased average nitrate and organic concentrations. Absolute humidity was shown to influence nitrate aerosol.

2.4 AIR POLLUTION-RELATED HEALTH EFFECTS

The effects of air pollution on human health have been a substantial research area for decades. Animal toxicology, human clinical exposure studies, field exposure assessment studies, and epidemiological investigations have all been used to derive data from the molecular scale to population level impacts of air pollution. Several adverse outcomes have been noted: premature death, hospitalization increases, and exacerbation of asthma are just a few. Moreover, other effects are biologic indications of responses that have uncertain outcomes. It is evidence provided by these studies that influences air pollution control policies around the world. Unfortunately, air pollution remains a problem and many questions regarding its impacts remain unanswered. One question that is of importance regarding climate change is if weather modifies the effect of air pollution exposure on human populations. This is a gap of considerable importance that has raised many arguments concerning our current understanding of air pollution-related health effects. In this section of the review, we focus on population level epidemiological studies that investigated the short-term (meaning an exposure-response window of five days or less) effects of O_3, PM, and NO_2 using time-series methodologies. We also place special emphasis on the Australian region. In regards to climate change, we consider studies that investigate the modification of health effects by weather. It is important to note that throughout this section risk estimates are presented with their 95% confidence intervals in [], where available.

O_3, PM, and NO_2

In the Northern Hemisphere, two of the most comprehensive and recognized population level epidemiological studies that focus on air pollution are the National Morbidity and Mortality Air Pollution Study (NMMAPS) conducted in the United States and the Air Pollution and Health: A European Approach (APHEA) study conducted in Europe (Katsouyanni, Zmirou et al. 1995; Samet JM 2000). Both of these studies estimate the short-term effects (typically a lag of up to five days is considered as 'short-term') of air pollution on the study populations using the framework of generalized additive Poisson regression models. The approaches aim to remove the confounding effects of seasonality and weather whilst estimating air pollution-related health effects. Findings from these studies have been the benchmark of the field and therefore we will briefly synthesize their results for our pollutants of interest on mortality.Finally, we should note that NMMAPs and APHEA are presented as they provide a critical foundation for time-series epidemiology and that multiple follow-up studies have been conducted (APHEA2 2001; Domincini et al. 2002; Ren et al. 2008; Samoli et al. 2006) ; however, for brevity they will not be discussed here.For PM_{10}, the NMMAP study noted a daily increase in mortality of 0.5% per 10 µg/m^3 increase in PM_{10} (Samet JM 2000). APHEA results for western European cities estimated that the increase in the daily number of deaths for all ages for a 10 µg/m^3 increase in PM_{10} was 2% [1%, 3%] (Katsouyanni, Touloumi et al. 1997). For O_3, APHEA noted that for a 50 µg/m^3 increase in the 1-hr maximum was associated with a 2.9% [1.0%, 4.9%] increase in mortality (Touloumi, Katsouyanni et al. 1997). Findings from NMMAPs show that for a 10 ppb increase in O_3 (during the summer) the posterior mean of the effect was 0.41 ppb [-0.20, 1.01]. For NO_2, APHEA noted a 0.30% [0.25%, 0.35%] increase in mortality per 10 µg/m³ increase in NO_2 (Samoli, Aga et al. 2006). However, no consistent pattern was seen in the NMMAPS data for NO_2.

As a follow up to the NMMAPS and APHEA studies, a collaborative effort between the two - Air Pollution and Health: A Combined European and North American Approach (APHENA), was conducted (Katsouyanni, Samet et al. 2009). Overall, the study noted that risk estimates from the US and Europe were consistent with previous findings for PM_{10} and O_3. Interestingly, Canadian cities were introduced into the analysis and the effects seen there were substantially higher. However, confidence intervals were significantly higher and it was suggested that population size was an issue.

Australia

In September of 2010, the Environment Protection and Heritage Council (EPHC) published results from its 'Expansion of the Multi-City Mortality and Morbidity Study' which examined the effects of air pollution on health in Australian and New Zealand cities during the years of 1998 to 2001 (Environment Protection and Heritage Council 2010). The study design was heavily influenced by the NMMAPS, APHEA, and APHENA protocols as the objective was to measure associations between daily air pollutant concentrations and daily hospital admissions/mortality counts. The cities considered in the study were Auckland, Brisbane, Canberra, Christchurch, Melbourne, Perth, and Sydney. Results found that O_3, PM_{10}, and NO_2 were all associated with increases in all cause mortality. However, the results for O_3 were only significant in the warmer months. Moreover, PM_{10} and NO_2 were both associated with increased hospital admissions for cardiovascular and respiratory disease. Ozone was found to significantly increase total respiratory and asthma related hospital admissions for children between the ages of 1 to 4 years in the warmer months. More specifically, a 1 ppb increase in the daily maximum 1-h average NO_2 resulted in a 0.2% [0.0%, 0.3%] increase in all cause mortality. A 1 ppb increase in daily maximum 8-h average O_3 during the warmer months resulted in a 0.1% [0.0%, 0.2%] increase in all cause mortality. For PM_{10}, a per interquartile range (IQR) increase in the 24-h average resulted in a 1.4% [0.2%, 2.6%] increase in all cause mortality. Comparison of results to international findings indicates that the effects of air pollution in Australia and New Zealand are similar to those around the world. Further comparison of results to previous studies conducted in Australia and New Zealand found that the results of the EPHC study are in general agreement with previous findings (Simpson, Williams et al. 2005). However, estimates for the effects of particulate matter were higher than many single-city studies conducted across the region (Simpson, Williams et al. 1997). This is most likely due to differences in controlling for bushfires (Morgan, Sheppeard et al. 2010). In Melbourne, there have been a few notable studies to focus on air pollution-related health effects (Simpson, Denison et al. 2000; Bennett, Simpson et al. 2007; Erbas, Chang et al. 2007; Dennekamp and Abramson 2011). However, none provide conflicting results with the EPHC study and therefore they will not be discussed in detail.

Weather Modification of Air Pollutant Health Effects

An area of growing interest in the air quality community is the *effect modification* (a.k.a., nonuniformity or heterogeneity of effect) of air pollution-related health effects by elements of weather. Simply put, effect modification occurs when exposure to a risk factor and the outcome varies due to the level of another variable. The weather element of most interest to air pollution researchers is temperature (Stafoggia, Schwartz et al. 2008). It is important to note that research has shown that changes in temperature – particularly at the extremes, can adversely affect human health (Basu 2009). Thus, temperature has historically been treated as a confounding factor in most air pollution-related health effects studies (APHEA2 2001). Temperature has historically been treated as a confounding factor because it is related to the exposure of interest – air pollution (Camalier, Cox et al. 2007; Jacob and Winner 2009; Pearce, Beringer et al. 2011), and it has also been associated with the outcome of interest – i.e. mortality/morbidity (McMichael, Wilkinson et al. 2008) However, due to a growing interest in climate related research, air pollution studies have begun to shift perspectives on the role of temperature (Roberts 2004; Ren and Tong 2006; Hu, Mengersen et al. 2008; Stafoggia, Schwartz et al. 2008). The plausibility behind this effect was first discussed when air pollution researchers began to find differential pollutant effects across seasons (Peng, Dominici et al. 2005). Typically, these were larger in the warmer months and it was inferred that weather was playing a role. The obvious candidate here is temperature.

Two recent studies in Europe and the United States found that temperature significantly modifies the effect of particulate matter on mortality (Roberts 2004; Stafoggia, Schwartz et al. 2008). Conversely, a study focusing on the effect of temperature found that the inclusion of air pollution in the models had very little effect on the risk estimates for temperature (Anderson and Bell 2009). However, interactions were not considered.

In Australia, results have also been found which indicate temperature behaves as an effect modifier in the air pollution-health relationship (Ren and Tong 2006). Even so, the authors' note that these effects are plausible for a number of reasons but that more research is needed as a clear pathway is yet to be defined. To a lesser extent, temperature has been observed to influence the ozone-mortality relationship; however, this was only noted by one study (Ren, Williams et al. 2009). This is most likely due to the temperature dependence of ozone.

2.5 CLIMATE CHANGE

Relatively speaking, issues regarding climate change impacts on air quality are a recent field of scientific investigation. Even so, it has been identified that climate change may influence air pollution and air pollution-related health effects by (a) changing weather and subsequently air pollution concentrations, (b) affecting natural and anthropogenic emission rates, (c) affecting background pollutant concentrations, and (d) impacting exposure patterns through changes in behavior due to adaptive responses (Bernard, Samet et al. 2001; Jacob and Winner 2009). Assessing the potential effect of climate change on air pollution has largely been based on the findings from observational studies that link weather to air pollution, model perturbation studies that manipulate weather elements in order to estimate an air pollution response, and model simulation studies that project future air quality by running air quality models using future climate scenarios (Jacob and Winner 2009). Estimation of the associated health effects is typically assessed by combining future air pollution scenarios (these are driven by future climate scenarios) with future population scenarios and modern day dose-response information. Assessments in this research focus on estimating the changes in global air pollution concentrations and/or regional/continental scale air pollution due solely to climate change over a period of the next 50 to 100 years.

In this section of the review, model simulation studies that have assessed the global impact on air pollution and studies that have assessed the regional/continental impacts of climate change are presented. Furthermore, works that have projected the future health consequences are also included. In brief, these studies are conducted as follows: (1) a future greenhouse gas emissions scenario is chosen and used to drive a global scale general circulation model (GCM), (2) output from the GCM is then used by a chemical transport model (CTM) to simulate the composition of the atmosphere on a global scale, (3) a regional climate model (RCM), using boundary conditions from the GCM/CTM, can then be used for finer-scale resolution over a region of interest, (4) the air pollution simulation is then done with a regional CTM using output from the RCM. Future health outcomes can then be assessed using these outputs. For more detailed information on study designs please see Jacob and Winner (2009). Also, please note that the IPCC emission scenarios are not explained here, for information see the Intergovernmental Panel on Climate Change emissions scenarios (IPCC 2000). The A2 scenario family, commonly used in modeling, is a collection of high emission scenarios. The pollutants of concern are, again, ozone, particulate matter, and nitrous oxides.

Global Projections

Using the A2 emissions scenario to project climate conditions for 2100, Liao, Chen et al. (2006) predict future ozone and particle concentrations driven by climate change by using emissions for 2000 and GCM projected 2100 climate conditions. In short, both present and future climate were simulated, with anthropogenic emissions held at present day levels to isolate the effects of climate change. Using this approach, the authors' found that between 2000 and 2100 climate change alone will reduce the global average tropospheric ozone burden -11.5%; however, this is not evenly distributed and increases are expected near heavily populated areas and areas of intense biomass burning. Moreover, when climate change is combined with anthropogenic emission changes (A2), a 43.8% increase is predicted in the ozone burden. For PM, the components of sulfate, nitrate, and black carbon, and primary organic aerosols are expected to decrease by 14%, 47%, 13%, and 9%, respectively. Nevertheless, secondary particles on the other hand are predicted to increase by 9%.

Racherla and Adams (2006), using a similar approach to Laio et al. (2006), investigated the shifts of global fine particulate matter and ozone concentrations to an A2 scenario for the 2050s where global average surface temperature increased 1.7 °C, humidity 0.9 g H_2O/kg air, and precipitation by 0.15 mm. Results found that the global burden of ozone decreased by 5% and its atmospheric lifetime reduced by 2.5 days. For PM, the global burden decreased by a range of 2 to 18% depending on species. To complicate matters, the model surface layer illustrated that there were regions of significant decreases and increases in the concentrations of fine particulate matter species and ozone indicating significant geographic variability.

As ensemble based assessments are the preferred, Dentener, Stevenson et al. (2006) used 26 state-of-the-art global atmospheric chemistry models and three different emissions scenarios to evaluate the effect of changing emissions and climate on global ground-level ozone for the year 2030. Results (and associated ± 1 standard deviations) indicate that by 2030, global average surface ozone is expected to increase 1.5 ± 1.2 ppb under the current air quality legislation around the world, and 4.3 ± 2.2 ppb under a relatively high IPCC (A2) emissions scenario. Results from a progressive scenario, were all currently feasible technologies are applied to reduce emissions, simulated a reduction in O_3 by 2.3 ±1.1 ppb.

Heald, Henze et al. (2008), using a coupled global atmosphere-land model driven by 2100 IPCC A1B scenario predictions linked to the Model of Emissions of Gases and Aerosols within the Community Land Model, predict the global mean secondary organic aerosol is predicted to increase by 36%, primarily due to increasing biogenic (26%) and anthropogenic (7%) emissions.

Regional

In the United States, Hogrefe, Biswas et al. (2004) simulated the effect of regional climate change alone on summer-averaged daily maximum 8-hour ozone for the 2020s, 2050s, and 2080s. Results showed increases of 2.7, 4.2, and 5.0 ppb when compared to the 1990s. For the 2050s, a separate analysis of various contributing factors was undertaken; results were +5.0 ppb from altered background ozone, +4.2 ppb from regional climate change, and +1.3 ppb from increased anthropogenic emissions. This evidence points to the importance of climate as a driver of ozone.

Another study in the United States modeled the effect of climate change only on ozone in 50 US cities for five summers in the 1990s and 2050s (Bell, Goldberg et al. 2007). The results show an increase in the summertime average daily maximum 1-hour ozone of 4.8 ppb (average of all cities), with a peak increase of 9.6 ppb. Notably, the study also estimated the effects of this increase on human health, with daily total mortality showing an increase of between 0.11 % and 0.27 %, from 1990 to 2050.

In Europe, Meleux, Solmon et al. (2007) predicted future ozone for the period 2070-2100, finding substantial increases (up to 25 per cent) in daily peak ozone for some regions. The authors noted the significance of biogenic isoprene emissions (an important ozone precursor), which will increase under higher temperatures.

Langner, Bergstrom et al. (2005) use two separate climate models to drive photochemical simulations for Europe, using the IS92a (business as usual) scenario, with results presented for the period 2050-2070. The simulations show an increase in surface ozone over southern and central Europe, and a decrease in northern Europe.

Forkel and Knoche (2006) examined the effect of climate change alone on ozone for southern Germany using simulations for the periods 1991-2000 and 2031-2039. Results indicate an increase of 2 to 6 ppb in the average daily maximum ozone, corresponding to a peak increase of approximately 10% in summer. Notably, biogenic emissions were varied with temperature (although vegetation cover was assumed to be the same) and anthropogenic emissions were held constant. The authors noted that the highest increase of the maximum ozone concentration seems to occur in regions where a high increase of the isoprene emissions coincides with high NO_x emission.

For PM, Tagaris, Manomaiphiboon et al. (2007) investigate the impacts of global change on overall concentrations and speciated components across the United States by comparing 2001 levels to projected levels in 2051. Their findings show that mean annual $PM_{2.5}$ concentrations across the United States are estimated to be 10% lower due to the estimated increase of precipitation in the future climate. Regionally, the eastern United States is projected to see the largest reductions due to emissions and meteorological changes. This is in direct contrast to the results seen by Racherla and Adams (2006) for the eastern United States, in which precipitation decreases were shown to increase PM.

A report by the Air Quality Expert Group (AQEG 2007) indicated that in the United Kingdom there is likely to be a decrease in emissions of most air quality pollutants and their precursors over the next 20 – 30 years. However, this is not because of a decrease in the use of fossil fuels, but because of improved technology. These improvements are expected to be driven by legislation. For example, the United Kingdom Government has pledged to cut carbon dioxide emissions by 60%, relative to 1990 levels, by 2060. If appropriate decisions are made, this is also likely to reduce emissions of air quality pollutants, since many air quality and climate change pollutants have the same sources.

Emissions from rapidly developing countries such as China and India are likely to increase, unless there are quite dramatic, and presently unanticipated, changes in technology. Not only will this lead to significant local and regional problems in these countries, but also to effects that are felt globally (AQEG 2007). Takemura, Nakajima et al. (2001) simulate future distribution, radiative forcing, and long range transport of aerosols in East Asia using Special Report Emissions Scenarios from the IPCC, a general circulation model, and a aerosol transport model for the next 50 years. Results suggest that carbonaceous aerosols will continue to increase over industrial and densely populated regions for the next 5 decades. Sulfate aerosols and wet deposition are also simulated to increase over East Asia in contrast to other areas of the globe.

2.6 SYNTHESIS AND RESEARCH GAPS

In this review, current knowledge on; three air pollutants – O_3, PM_{10} and NO_2, the influence of weather in the process of air pollution in the environment, the health effects of air pollution, and the potential affect of climate change in future air quality has been presented. For each last three topics, results from studies that were conducted at various locations and which used various approaches to the problems of interest were made accessible. In this closing section of the review focus is shifted from presentation to examine what those results mean and present the research needs and data gaps for the final three components – weather and air pollution, air pollution-related health effects, and climate change.

In the section on weather and air pollution two scales of meteorological phenomena were addressed – *synoptic*-scale events and *local*-scale elements. These scales were chosen for review because they have been shown to be both important to air quality and susceptible to climate change. The review of synoptic-scale air pollution meteorology identified that the major factors in which large-scale pressure systems influence air pollution are through vertical pollutant transfer, horizontal pollutant transfer, and cloud cover. Previous studies demonstrated that occurrences of anticyclonic circulation events resulted in increased pollutant concentrations due to the increased atmospheric stability and warmer temperatures and occurrences of cyclonic events resulted in the opposite due to the cooler, windier conditions. Similar findings were seen across variable topographies and climates, working to strengthen understanding of this process. While these findings do point to some common generalities regarding influences at the synoptic-scale, the response of air pollution to events at the synoptic-scale is more complicated than these simple associations. For example, the studies presented do not examine the influence of the entire range of synoptic states on air pollution but instead link high air pollution events to coincident conditions. Therefore, a critical research gap is to examine the influence of an entire range of synoptic states on air pollution. This would provide a better indicator of how important synoptic-scale weather is to air pollution. Moreover, as the previous research conducted in south-eastern Australia has only focused on ozone, the response of particulate matter and nitrogen dioxide over the region is unknown. These limitations are addressed in the first publication presented in this thesis as Chapter 3. In this Chapter, the influence of synoptic-scale circulation on O_3, PM_{10}, and NO_2 over the study region is evaluated. Additionally, this chapter expands upon previous synoptic-scale research by incorporating novel classification and statistical techniques that allow a comprehensive assessment over the study domain.

The response of air pollution to weather at the local-scale is more complex than events occurring at the synoptic-scale because of the heterogeneity of effects generated by local elements. It is understood that local weather elements play key roles in pollutant generation, transformation, transportation, and removal from the environment. Observational studies and modeling studies both indicate that temperature is of primary importance. This is most likely due to a broad range of factors as temperature drives aspects of air pollution meteorology and air pollution chemistry. Studies have demonstrated that increases in temperature typically result in higher pollutant concentrations – particularly for ozone. The results are less conclusive for PM as heterogeneous composition results in differential effects for individual meteorological elements and across geographic locations. To date, the majority of research focuses on ozone with an increasing trend in particle research and very little attention being given to NO_x. Winds, humidity, boundary layer conditions, and precipitation were all noted as being important; however, their importance was typically identified as being less significant than temperature.

Despite extensive research, critical gaps still exist in the understanding links between local meteorological elements and air pollution. Again, like the synoptic focused research the majority of understanding is based upon works conducted in the Northern Hemisphere. Thus, air pollution management in the Southern Hemisphere has to make the assumption that air pollution meteorology over their given region will behave in a similar fashion to the Northern Hemisphere. While this may be safe to assume in a very general fashion, the Southern Hemisphere, and in particular Australia, presents a very different climate and urban environment than its Northern Hemisphere counterparts making differential influences likely. For example, work by Camalier, Cox et al. (2007) identified variable influences of meteorological elements across the continental United States for ozone. This identifies the need for location specific research. These limitations are addressed in the second publication presented in this thesis as Chapter 4. In this chapter the influence of local weather elements on O_3, PM_{10}, and NO_2 over the study region is examined using a highly flexible nonlinear statistical model. Additionally, this Chapter expands upon previous research by evaluating pollutant responses at higher temperatures than previously evaluated.

The adverse effects of air pollution on human health have been well documented across variable populations and exposure conditions. There is clear evidence that increases in air pollution concentrations result in increased morbidity and mortality at the population level. However, the effects of air pollution on health are very complex, as there are many different sources and their individual effects vary from one to the other.

Unfortunately, many gaps remain in the understanding of how these pollutants influence health. One of these gaps is the role of weather, in particular temperature, as an effect modifier in the air pollution-health relationship. Although limited, the studies presented demonstrate the complexity associated with characterizing the influence weather has on air pollution related health effects. This is challenging because of the complexity of the temperature-health relationship – an association that exhibits nonlinearity at the extremes and complex lagged effects (Armstrong 2006), and the complexity of the air pollutant-temperature relationship (Camalier, Cox et al. 2007; Carslaw, Beevers et al. 2007). Spatial variability in these relationships further complicates matters and thus it is clear that more research is needed. Thus, the focus of the final research component of this thesis – Chapter 5, examines the role of temperature in the relationship between air pollution (O_3, PM_{10}, and NO_2) and mortality in Melbourne. This work will again provide a unique geographical perspective and characterize relationships using state-of-the-art statistical techniques.

The final section of the review focused on the impact of climate change on air quality. Overall, climate change on a global scale is anticipated to decrease global concentrations of particulate matter mostly due to an increase in atmospheric moisture. Regionally, areas that see increases in temperature, decreases in moisture, and increases in periods of stable air masses are likely to see increases in both ozone and particle concentrations. Additionally, the components that make up particulate matter are also expected to shift under a changing climate. Thus, particles present the most difficult research challenge in the context of climate change as there is still so much that is not understood about PM. The results for ozone are clearer as climate change has the potential to influence a number of meteorological variables in addition to temperature (ultraviolet radiation, wind speed, precipitation, atmospheric mixing and transport) that are important to ozone. Whether changes in these variables lead to increases, decreases, or no change in ozone concentrations in a given region will depend upon whether the effects of these individual changes on ozone act in concert or counteract each other. Additionally, changing patterns of atmospheric circulation at the hemispheric to global level are likely to be just as important as regional patterns for future local air quality (Takemura, Nakajima et al. 2001; Langmann and Graf 2003). Furthermore, spatial variation is expected, and depending on the spatial and temporal distribution of emissions, pollutant concentrations may remain unchanged, increase, or even decrease in some areas, highlighting the need for detailed modeling of individual regions.

The primary concern of climate change-air quality discussion is that temperatures will rise due to global warming and that wind speeds will decrease due to reduced global circulation. Moreover, specific areas need to also be concerned with decreasing precipitation – a significant mode of air pollution removal. Thus, to be prepared for climate change it is clear that air pollution studies need to be site-specific. This thesis provides an examination of present day data with statistical analysis in order to lay a foundation of how projected climate change may influence air quality over Melbourne, Australia. Of course, this thesis will not provide all the answers for the region and model simulations will certainly need to be conducted. Nevertheless, much can be inferred from the results presented in Chapters 3-5.

CHAPTER 3

INVESTIGATING THE INFLUENCE OF SYNOPTIC-SCALE METEOROLOGY ON AIR QUALITY USING SELF-ORGANIZING MAPS AND GENERALIZED ADDITIVE MODELING

Material in this chapter is reproduced from a research article published in Atmospheric Environment. References have been moved to a consolidated bibliography.

Declaration by candidate for Thesis Chapter 3

In the case of Chapter 3, the nature and extent of my contribution to the work was the following:

Nature of contribution	Extent of contribution (%)
Formulation of research problem and the context of the research in the wider literature; data acquisition and analysis; interpretation of results and writing.	95%

The following co-authors contributed to the work:

Name	Nature of contribution	Extent of contribution (%)
Jason Beringer	Formulation and revision of writing	1%
Neville Nicholls	Formulation and revision of writing	1%
Rob J Hyndman	Statistical guidance	1%
Petteri Uotila	Development of data	1%
Nigel J Tapper	Formulation and revision of writing	1%

Candidate's Signature	Date
[redacted]	6 July 2011

Declaration by co-authors

The undersigned hereby certify that:
(1) the above declaration correctly reflects the nature and extent of the candidate's contribution to this work, and the nature of the contribution of each of the co-authors.
(2) they meet the criteria for authorship in that they have participated in the conception, execution, or interpretation, of at least that part of the publication in their field of expertise;
(3) they take public responsibility for their part of the publication, except for the responsible author who accepts overall responsibility for the publication;
(4) there are no other authors of the publication according to these criteria;
(5) potential conflicts of interest have been disclosed to (a) granting bodies, (b) the editor or publisher of journals or other publications, and (c) the head of the responsible academic unit; and
(6) the original data are stored at the following location(s) and will be held for at least five years from the date indicated below:

Location(s): [1]School of Geography and Environmental Science, Monash University; [2]Department of Econometrics and Business Statistics, Monash University; [3]Marine & Atmospheric Research, CSIRO, Melbourne, Australia

Name	Signature	Date
Jason Beringer[1]	[redacted]	8 July 2011
Neville Nicholls[1]	[redacted]	18 July 2011
Rob J Hyndman[2]	[redacted]	18 July 2011
Petteri Uotila[3]	[redacted]	18 July 2011
Nigel J Tapper[1]	[redacted]	18 July 2011

TITLE: INVESTIGATING THE INFLUENCE OF SYNOPTIC-SCALE METEOROLOGY ON AIR QUALITY USING SELF-ORGANIZING MAPS AND GENERALIZED ADDITIVE MODELING.

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Keywords: air pollution, generalized additive models, self-organizing maps, and synoptic climatology.

Published in Atmospheric Environment 45 (2011) 128-136

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3.1 ABSTRACT

The influence of synoptic-scale circulations on air quality is an area of increasing interest to air quality management in regards to future climate change. This study presents an analysis where the range of expected synoptic-scale circulation patterns over the region of Melbourne, Australia are determined and linked to regional air quality. A self-organizing map (SOM) has been applied to daily mean sea level pressure (MSLP) reanalysis to obtain twenty large-scale synoptic patterns in the Australian region. A time series of the occurrence of the synoptic archetypes was then employed within the framework of a generalized additive model (GAM) to identify links between synoptic-scale circulation and observed changes air pollutant concentrations. The GAM estimated shifts in pollutant concentration under the occurrence of each synoptic type after controlling for long-term trends, seasonality, weekly emissions, spatial variation, and temporal persistence. Results found that the overall explanatory power of the synoptic archetypes in the models to be rather modest with 5.1% of the day-to-day variation in O_3, 4.7% in PM_{10}, and 7.1% in NO_2 being explained. This indicates that synoptic-scale circulation features are not the primary driver of day-to-day pollutant concentrations. Nonetheless, further analysis of the partial residual plots identified that despite a modest response at the aggregate level, individual synoptic categories had differential effects on air pollutants. In particular, when synoptic conditions result in a northeasterly gradient wind over the Melbourne area NO_2 and O_3 were 20% higher than average. For PM_{10} maximum increases of up to 20% occurred when a strong anticyclonic system was centered directly over the Melbourne area. In sum, the unified approach of SOM and GAM proved to be a complementary suite of tools capable of identifying the entire range of synoptic circulation patterns over a particular region and quantifying how they influence local air quality.

3.2 INTRODUCTION

Increased air pollutant concentrations in the urban environment do not typically result from sudden increases in emissions, but rather from meteorological conditions that impede dispersion in the atmosphere or result in increased pollutant generation (Cheng and Campbell 2007). A combination of meteorological variables important to these conditions includes temperature, winds, radiation, atmospheric moisture, and mixing depth (EPA 2009). Because synoptic-scale circulations are the envelope that governs all the above meteorological features synoptic climatological approaches have become a popular means for evaluating impacts of large-scale meteorological conditions on local environmental phenomena such as air pollution (Triantafyllou 2001; Cheng and Campbell 2007; Beaver and Palazoglu 2009). This has led the air quality community to recognize synoptic-scale circulations as an important driver of local air pollution (EPA 2009).

In this study, we wish to increase the understanding of the relationship between large-scale synoptic circulations and air pollution in Melbourne, Australia. The city of Melbourne, with a population of approximately 3.9 million (ABS 2010), is situated on Port Phillip Bay at the south-eastern edge of continental Australia in close proximity to the Southern Ocean at 37° 48’ 49” S and 144° 57 47” E (Figure 3.1). The climate of Melbourne can best be described as moderate oceanic and the city is famous for its changeable weather conditions (BOM 2010). This is due in part to the city's location at the pole-ward margin of a sub-tropical continent that results in the passage of very differing air masses over the region. The mid-latitude synoptic weather systems that affect the region produce persistent westerly winds between the subtropical anticyclonics to the north and the Southern Ocean lows to the south. This region is also dominated by fronts, which result from the interaction of subtropical and polar air masses. Although Melbourne’s air quality can be described as relatively good when compared to other urban centers of similar size, recent periods of anomalous environmental conditions present an interesting opportunity for analysis (Murphy and Timbal 2008). The overall objective of this research was to investigate how the occurrences of patterns within the expected range of synoptic circulation modes over Melbourne influence local air quality. In order to achieve this objective, a synoptic climatology was first developed for south-eastern Australia using self-organizing maps (SOMs) and those results were then linked to air pollution data using generalized additive modeling (GAM).

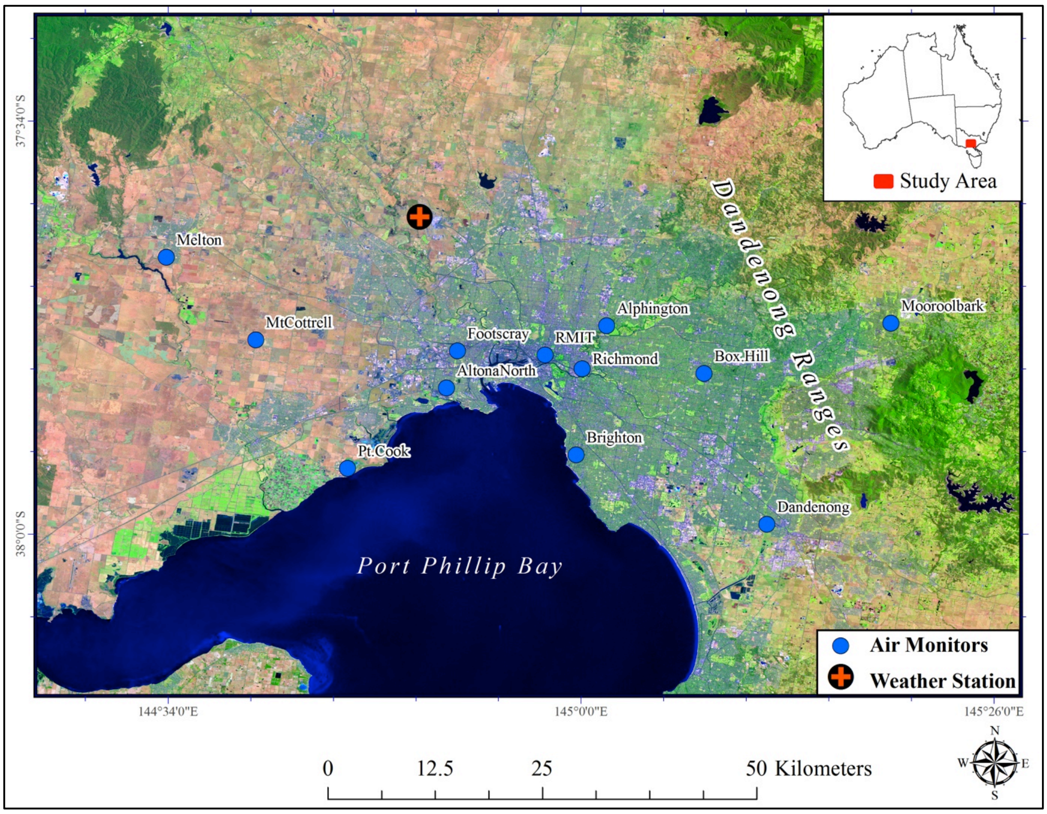

Figure 3.1: Map of local meteorological and air quality monitoring locations used in this study.

3.3 DATA

a. Large-scale Meteorological Data

Mean sea-level pressure (MSLP) data covering large tracts of space and time enables the investigation of large-scale pressure (synoptic) patterns over a particular region. MSLP is a commonly used proxy for atmospheric circulation because it is well known that it relates well to the spatial pattern of these processes (Huth, Beck et al. 2008). In this analysis, four-time daily gridded MSLP data from the ERA-Interim reanalysis (1989 to 2008) was used to determine the synoptic-scale circulation patterns influencing Melbourne. This reanalysis was produced by the European Centre for Medium-Range Weather Forecasts (ECMWF) and is discussed in more detail by Uppala et al. (Uppala and Dee 2008). MSLP fields were obtained for 10 a.m., 4 p.m., 10 p.m., and 4 a.m. local standard time (LST) for each day in the reanalysis period at a spatial resolution of 0.72° over the spatial domain of 35-44° S and 140-150° E. It is important to note that ERA40 and NCEP/NCAR reanalysis products were also trialed for this analysis. While these products produced climatologies of similar agreement ERAI was chosen as the enhanced spatial resolution of the data produced synoptic types that were more interpretable for the Melbourne region.

b. Local-scale Meteorological Data

The characterization of local weather conditions during the occurrence of varying synoptic states was made using daily automatic weather station observations for site number 086282 (Melbourne International Airport) for the period of 1999 to 2006. This site is located at 37° 40’ 12” S and 144° 49’ 48” E with an elevation of 113 m and was chosen because a comprehensive range of measures are collected on a consistent basis. Variables provided by Climate Information Services, National Climate Centre, Bureau of Meteorology included:

- Maximum daily temperature (°C)
- Mean sea level pressure (hPa)
- Global radiation (MJ/m^2)
- Water vapor pressure (hPa)
- Zonal *(u)* and meridional *(v)* wind components (km/hr)

- Precipitation (mm).

Additionally, boundary layer height (BLH) was taken from the ERA-Interim data using the location of 37° 30' 0" S and 145° 30' 0" E for 4 p.m. LST - the approximate time of maximum boundary layer depth.

c. Air Pollutant Monitoring Data

The Environmental Protection Authority Victoria (EPAV) provided local air pollution data taken from the Port Phillip Bay air monitoring network (Figure 3.1). Pollutants included ozone (O_3), particulate matter $\leq$ 10 µm (PM_{10}), and nitrogen dioxide (NO_2). O_3 and NO_2 concentrations are reported in parts per billion by volume (ppb) and were measured using pulsed fluorescence chemiluminescence and ultra violet absorption techniques. PM_{10} concentrations were measured using photospectrometry and are reported in micrograms per cubic meter (µg/m^3). This analysis uses the daily maximum value for 8-hr O_3, the 24-hr mean value of PM_{10}, and the daily maximum value for 1-hr NO_2 from all available monitoring locations over the period of 1999 to 2006 (Table 3.1). These timeframes were selected to parallel air quality objectives in the State Environment Protection Policy for ambient air quality (SEPP 1999). It is important to note that days on which significant air quality events (bushfires, factory emissions, etc.) deemed unrelated to meteorology occurred were omitted from the data. Furthermore, following guidance from EPAV, limits of detection were imposed at 5 ppb for O_3 and NO_2 and 5 µg/m^3 for PM_{10}. Essentially this removes data perceived as random noise from the data sets by omitting values below five. We should note that this removed approximately 1.6% of O_3, 0.2% of PM_{10}, and 3.4% of NO_2 – a very small portion of the data.

Table 3.1: Descriptive statistics of regional air monitoring data used in model development.

	O_3 (ppb)				PM_{10} (µg/m³)				NO_2 (ppb)			
Location	Mean	SD	Min	Max	Mean	SD	Min	Max	Mean	SD	Min	Max
Alphington	20.2	8.8	5.0	63.0	17.3	6.8	3.7	76.7	23.9	8.1	5.0	69.0
AltonaNorth	21.5	9.4	5.0	87.0	--	--	0.0	0.0	23.3	10.2	5.0	79.0
BoxHill	20.7	9.3	5.0	64.0	15.2	7.4	3.6	75.3	22.0	7.7	5.0	64.0
Brighton	22.8	8.9	5.0	75.0	15.2	6.5	3.0	75.9	21.7	9.5	5.0	74.0
Dandenong	23.0	9.1	5.0	73.0	17.6	8.0	3.8	51.8	21.9	8.4	5.0	65.0
Footscray	21.7	8.2	5.0	63.0	18.0	8.0	3.6	74.5	23.6	9.7	5.0	81.0
Melton	28.1	8.7	8.0	102.0	--	--	0.0	0.0	--	--	--	--
Mooroolbark	24.5	9.0	5.0	63.0	19.3	9.2	3.2	102.8	18.1	6.6	5.0	39.0
MtCottrell	26.5	8.7	6.0	72.0	--	--	0.0	0.0	--	--	--	--
PtCook	24.2	8.6	5.0	85.0	--	--	0.0	0.0	17.1	8.9	5.0	66.0
Richmond	--	--	--	--	17.2	7.8	3.9	110.5	24.8	8.9	5.0	76.0
RMIT	16.7	7.8	5.0	61.0	18.8	7.5	4.4	82.2	26.2	9.7	5.0	90.0
Overall	22.2	9.2	14.0	37.0	17.1	7.7	6.6	43.2	22.5	9.3	9.0	56.0

3.4 METHODS

a. Self-organizing Maps

A self-organizing map (SOM) is an artificial neural network (ANN) algorithm meant to be used for clustering, visualization, and abstraction (Kohonen 2001). Formally, the SOM can be described as a non-linear mapping of high dimensional input data onto the elements of a regular low-dimensional array based on their similarity in an ordered fashion (Kohonen 2001). The term 'self-organizing' refers to the SOMs training procedure in which iterative updates are made to the arrangement of the low-dimensional array based on the similarity/dissimilarity of the data from the reference vectors (Astel and Tsakouski 2007). The 'map' refers to the two-dimensional array that is typically used to visualize these vectors (e.g., synoptic circulation patterns) and their final arrangement (Hewitson and Crane 2002). SOMs have been used for a wide variety of environmental applications and have performed well when compared other similar techniques (Astel and Tsakouski 2007; Kalteh and Hiorth 2008).

In this analysis, we use SOMs to perform synoptic typing - an approach in which the atmospheric state is partitioned into broad categories in terms of the spatial patterns associated with atmospheric circulation. Synoptic typing or classification is used in order to provide insight into the influence of large-scale processes on local environmental conditions (Huth, Beck et al. 2008). Although there are several approaches to generating synoptic climatologies, we chose the method of SOM for its capabilities in identifying infrequent extreme events (Cassano, Uotila et al. 2006). This facility is of significant advantage to our air quality study as air pollution events – especially in Melbourne, tend to be rather infrequent which suggests that the forcing mechanism is also infrequent (NEPC 2005).

Technically, we apply the SOM algorithm to the MSLP data described previously using a 5 × 4 output dimension. It is important to note that the SOM algorithm, like *k*-means clustering, seeks to identify a user-defined number of synoptic types within the distribution of the input data. The use of a small dimension will result in a map that provides a broad generalization of the input data while a large dimension will result in a map with types that may be quite similar to adjacent types (Cassano, Uotila et al. 2006). We decided to base our SOM dimension (5 × 4) on previous works conducted in Australia as this arrangement has proven representative of the expected range of synoptic patterns over our region of interest (Hope, Drosdowsky et al. 2006; Alexander, Uotila et al. 2010; Nicholls, Uotila et al. 2010). Furthermore, this dimension was also deemed practical for use in our statistical analysis.

After the SOM algorithm is applied to the data and a resulting 'map' is produced a histogram of circulation can be developed. Applying the trained SOM to each data grid in the time series and assigning particular synoptic type to each period in the analysis achieves this. This results in a time series of synoptic charts that can be used to determine the frequency and duration of individual types within the data space (Hewitson and Crane 2002). This is a particularly attractive feature when trying to understand how events manifest from atmospheric circulation processes. Additionally, from this analysis it is also relatively simple to take all days mapping to a node and determine the mean meteorological conditions. In our case, this is done using the automatic weather station observations from Melbourne airport (Table 3.2). The software used to create the SOM is part of the SOM_PAK, which is available from http://www.cis.hut.fi/research/som-research. For more information regarding SOMs within the sub-discipline of 'synoptic climatology' please refer to Hewitson and Crane (2002).

Table 3.2: Mean values and standard deviations of observed meteorology under each synoptic type.

SOM	Temp (°C)		WVP (hPa)		*v* wind (km/hr)		*u* wind (km/hr)		Rad (MJ/m^2)		Precip (mm)		MSP (hPa)		BLH (m)	
	AVG	SD	AVG	SD	AVG	SD	AVG	SD	AVG	SD	AVG	SD	AVG	SD	AVG	SD
0	18.7	5.7	10.3	2.2	13.1	12.9	-10.9	8.9	12.8	7.5	3.3	10.2	1004	3	1616	614
1	19.4	5.6	9.9	2.2	11.4	14.3	-9.5	7.9	14.1	8.1	2.2	9.1	1010	2	1673	525
2	18.4	4.3	10.2	2.2	1.0	12.9	-12.2	6.7	14.9	7.8	1.6	2.9	1013	2	1524	373
3	17.8	4.3	10.3	2.3	-6.5	9.4	-11.6	6.7	15.3	8.0	2.1	4.5	1016	2	1455	345
10	23.9	7.8	13.1	3.1	7.7	15.1	-2.9	5.3	13.9	8.9	2.6	6.1	1006	2	1722	912
11	25.6	8.5	11.6	2.7	18.8	14.5	-0.4	3.2	16.7	10.4	0.6	1.9	1011	2	1904	921
12	20.7	6.1	10.3	2.1	12.2	15.2	-4.2	3.8	15.2	8.1	0.6	1.4	1015	2	1564	554
13	17.2	3.5	10.0	1.7	-0.5	12.6	-9.0	5.8	13.7	8.1	1.0	2.3	1019	2	1297	355
20	23.6	7.2	13.8	3.5	-0.6	13.3	-1.6	5.2	17.2	10.3	3.5	10.2	1010	2	1559	840
21	27.2	7.2	11.9	2.9	15.0	11.9	0.8	3.0	19.8	9.3	0.3	1.8	1014	1	2010	857
22	21.5	6.8	9.8	2.1	22.2	14.4	0.1	3.3	14.8	8.2	0.2	0.8	1017	1	1517	680
23	17.4	4.2	9.5	1.6	6.2	15.6	-5.2	5.8	13.1	7.6	0.9	1.8	1022	2	1205	336
30	22.0	5.8	13.5	3.2	-7.7	9.1	-2.6	4.6	17.6	9.9	1.7	4.6	1014	2	1381	643
31	25.9	6.8	12.3	2.8	4.0	12.0	0.4	3.3	22.0	8.9	0.3	1.4	1017	1	1848	759
32	21.2	6.2	10.3	2.2	12.4	14.8	0.0	3.2	16.2	9.1	0.6	2.4	1020	2	1370	645
33	15.9	3.2	9.3	1.5	5.1	13.6	-4.3	5.5	10.8	5.8	0.7	2.1	1026	2	1050	306
40	20.1	4.8	12.5	2.6	-13.6	7.8	-2.9	4.5	17.6	9.9	2.5	6.1	1017	2	1326	425
41	22.1	5.3	12.2	2.4	-10.0	9.8	-1.7	4.5	20.1	9.2	1.3	4.3	1019	2	1556	559
42	20.9	6.1	10.7	2.2	-1.7	12.4	-0.7	4.0	18.3	9.3	0.8	3.1	1022	1	1415	584
43	17.0	4.3	9.8	1.7	0.5	10.9	-1.9	3.7	12.3	6.8	0.4	1.2	1028	3	1064	401

b. Generalized Additive Models

Generalized additive models (GAMs) are regression models where smoothing splines can be used instead of linear coefficients for covariates (Hastie and Tibshirani 1990). This approach has been found particularly effective at handling the complexities of air pollution research (Aldrin and Haff 2005; Carslaw, Beevers et al. 2007). The additive model in the context of a concentration time series can be written in the form (Hastie and Tibshirani 1990):

$$log(y_i) = \beta_0 + \sum_{j=1}^{n} s_j(x_{ij}) + \varepsilon_i$$

(3.1)

where y_i is the ith air pollution concentration, β_0 is the overall mean of the response, $s_j(x_{ij})$ is the smooth function of ith value of covariate j, n is the total number of covariates, and ε_i is the ith residual with var(ε_i)= σ^2, which is assumed to be normally distributed. Smooth functions are developed through an integration of model selection and automatic smoothing parameter selection using penalized regression splines, which while optimizing the fit, make an effort to minimize the number of dimensions in the model (Wood 2006). The choice of the smoothing parameters is made through restricted maximum likelihood (REML) and confidence intervals are estimated using an unconditional Bayesian method (Wood 2006). This analysis was conducted using the *gam* modeling function in R environment for statistical computing (R Development Core Team 2010) with packages 'mgcv' (Wood 2006).

c. Model Development

The first step in the selection of individual models for O_3, PM_{10}, and NO_2 was to fit a preliminary base model. This was fit to each pollutant in order to control for the seasonality, persistence, spatial trend, and weekly emissions patterns that exist in these data. Following model (3.1) the preliminary model can be written as:

$$\log(y_i) = \beta_0 + s(time) + s(dow) + s\ (long,\ lat) + s\ (y_{i-1}) + \varepsilon_i \tag{3.2}$$

where *time* is a numeric vector ranging from 1 to 2922 (each day in the study period) included to account for long-term trends and seasonality, *dow* is a numeric vector ranging from 0 to 6 included to account for day-of-the-week, *long* and *lat* are the spatial coordinates of each monitor location included to account for spatial trend, and y_{i-1} is one day lag term included to account for short-term temporal persistence. It is important to note that the residual spatial variation is controlled by including a tensor product smooth, $s(long,\ lat)$, in the model and a smooth function of the preceding day's pollutant concentration, $s(y_{i-1})$, was included to control for temporal autocorrelation in residuals (Bivand et al. 2008). Additionally, since air pollution data are inherently cyclic, a predetermined smoothing parameter of k=32 (one knot (k) for each change of season) was used for the construction of the spline function for *time*. The motivation for this control is that function should represent a relatively symmetric cyclic pattern in the data. To check the adequacy of our methods for controlling for space-time effects, box-plots and time-series plots of residuals by monitor location were examined. No violations of assumptions were obvious in any pollutant.

Finally, the categorical predictor for synoptic-scale circulations (C) -- included to represent synoptic-scale circulation types – was added to the model. Following model (3.1) final models can be written as:

$$\log(y_i) = \beta_0 + s(time) + s(doy) + s(dow) + s\ (long,\ lat) + s(y_{i-1}) + C_p + \varepsilon, \tag{3.3}$$

where the term C_p represents the effect of the pth synoptic type and $p = 0, 1, \ldots, 19$.

d. Characterization of Synoptic Influence on Air Pollution

The explanatory power of model (3.3) was measured using the R^2 statistic. The aggregate impacts of synoptic circulations on each pollutant are assessed by taking the difference in the R^2 of model (3.2) and model (3.3). Individual relationships between particular synoptic types and each air pollutant are assessed using partial response plots presented as marginal effects.

3.5 RESULTS AND DISCUSSION

a. Self-Organizing Map

The SOM of MSLP provides a clear visualization of the atmospheric continuum affecting Melbourne by presenting twenty archetypes of synoptic states that characterize large-scale circulation over the region (Figure 3.2). Individual archetypes displayed in Figure 3.2 are referenced throughout the text using *xy* coordinates with archetype *00* being in the top left corner and archetype *43* being in the bottom right corner. The top left region of the map (*00*, *01*, and *10*) broadly infers a zonal flow with weak indications of troughs through central Victoria and up the east coast. These synoptic types are representative of low-pressure systems and likely exhibit relatively windy conditions due to the close nature of the isobars. The bottom left region of the map (*02*, *03*, *12*, and *13*) displays patterns that infer a more meridional (N to S) flow following front/trough passage, with lower pressure to the southeast and higher pressure to the northwest. Again, the isobars indicate that conditions under these regimes are also likely to be very windy. Patterns that are dominated by anticyclones (*33*, *42*, *and 43*) are located in the bottom right hand region of the map and likely experience relatively light winds. The top right hand region of the map (*30*, *31*, *40*, and *41*) suggests more zonal flow broadly from the east to northeast with relatively light winds. The top center of map (*20*, *21*, *and 22*) indicates weak meridional northerly flow with generally low pressure to the west and higher pressure to the east.

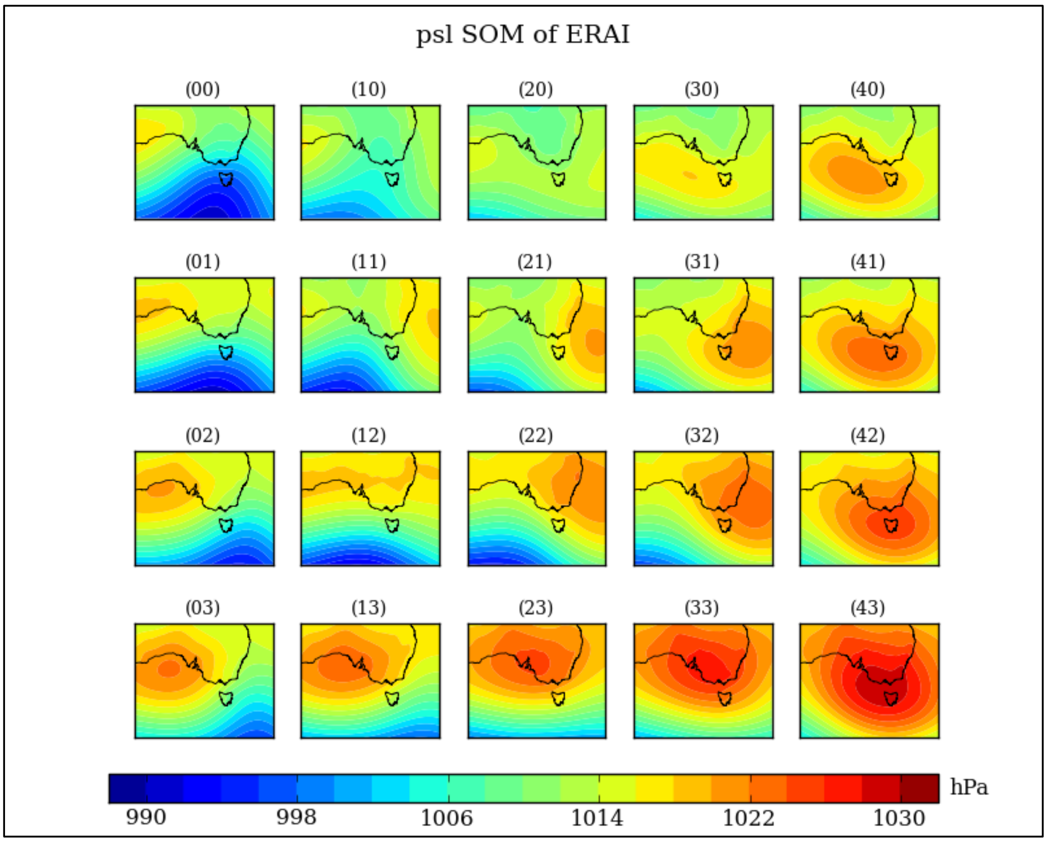

Figure 3.2: Synoptic types of annual MSLP pressure patterns generated using a 5×4 self-organizing map on ERAI reanalysis fields from 1989 to 2008. Individual nodes represent a classified synoptic type. Reference labels are provided above each node.

Frequency analysis of the archetypes across the data space found that the dominant modes of circulation are a cyclonic low-pressure circulation pattern (*00*) at 8.83%, and an anticyclonic circulation pattern (*43*) at 10.28% (Figure 3.3). In general, higher frequency modes are represented on the outer portions of the map while lesser frequent modes are presented closer towards the center. Notably, the frequency distribution across the map is quite varied from the expected 5%, with the frequency of occurrence showing an approximately 3:1 range from the most frequent to the least frequent archetype. Transitional states are represented by the less frequent archetypes.

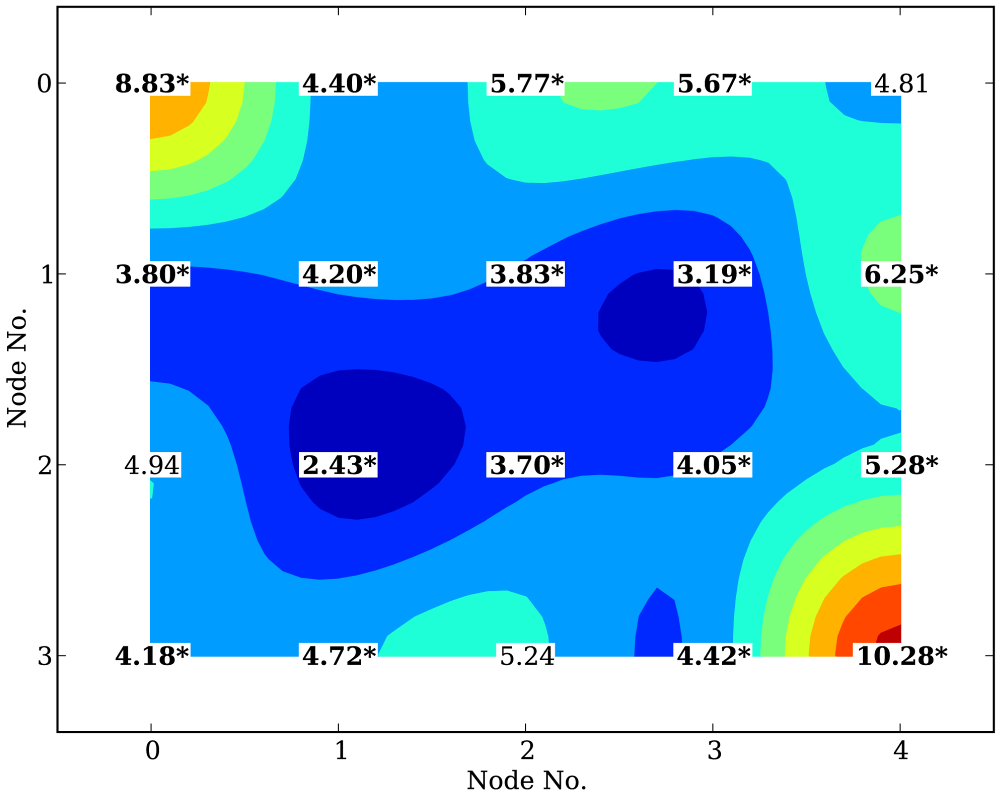

Figure 3.3: Histogram of circulation pattern frequencies (%) mapped to each SOM node over the period of 1989 to 2006. Frequencies significantly different from the expected 5% at the 95% confidence level are in boldface*.

Overall, using the SOM algorithm to classify MSLP patterns provides results that identify the dominant flow regimes for Melbourne to be westerly with low-pressure to the south over the Southern Ocean and high-pressure with light winds tending easterly. Transitory patterns are also identified and are placed closest to the dominant mode in which they are most similar. These pressure patterns can be used to infer regional scale wind speeds and direction along with mixing and would be expected to reflect the generally eastward movement of weather systems in this region. Table 3.2 helps provide an understanding of the local meteorology that is characteristically generated by each circulation mode.

b. Synoptic effects on air pollution

The GAM results found that circulation modes significantly influence ($F=122.6$, d.f.=19, $p<.0001$) O_3, PM_{10} ($F=69.42$, d.f.=19, $p<.0001$), and NO_2 (F=

133.4, d.f.=19, $p<.0001$) in Melbourne. Overall, model (3.3) explained 48.7% of O_3, 41.4% of PM_{10} and 36.7% of NO_2 using the covariates described in section 3.2. The addition of the categorical variable used to represent the SOM output was found to explain 5.1% of O_3, 4.7% of PM_{10}, and 7.1% of NO_2 variance in the final models, respectively. This suggests that synoptic-scale circulation is not the major driver of air pollution variations on a day-to-day basis across Melbourne. However, further analysis of the individual effects of each archetype clearly indicates that changes in circulation results in significant deviations for each pollutant in our study (Figures 3.4 and 3.5).

An overall picture of the range of effects for large-scale circulation on air pollution indicates that circulations being identified as similar by the SOM resulted in similar effects on each pollutant (Figure 3.4). This provides strong evidence that the increases in air pollution identified are the result of meteorological conditions. For ozone, concentrations were found to increase under modes located towards the right hand center of the map and decrease under modes located on the periphery – particularly the upper left corner. Particles and NO_2 show similar increases towards the right hand center of the map but also display increases towards the bottom right corner of the map (Figure 3.4). Decreases for these pollutants are also strongest for the upper left corner of the map.

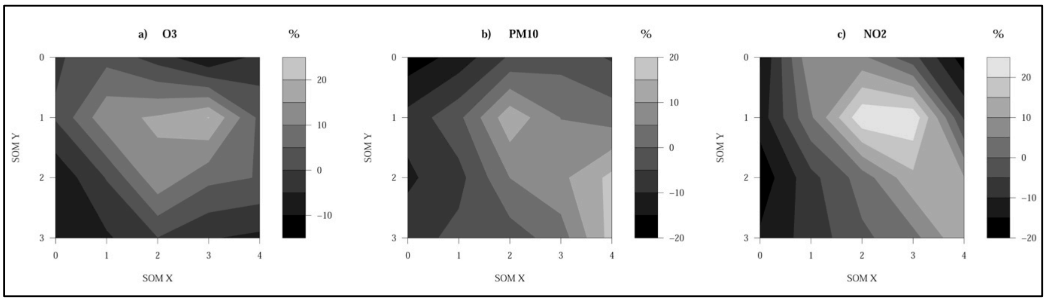

Figure 3.4: Contour plots of the estimated marginal effect (%) of circulation patterns on each pollutant mapped to each SOM node.

In regards to the right hand center of the map, a more specific analysis found that across all pollutants the largest significant positives were observed for either archetypes *31*or *21* (Figure 3.5). These archetypes are associated with an anticyclone being located to the east of south-eastern Australia in the Tasman Sea (Figure 3.2). The pressure gradients are suggestive of a light northeasterly gradient wind over Melbourne, which would oppose the inland penetration of bay and sea breezes. This has been found to impede the dispersion of local pollutants in Melbourne (Tory, Cope et al. 2004). Additionally, these synoptic types governed local meteorological conditions that can be characterized as having relatively high temperatures, high levels of radiation, northeasterly winds, low atmospheric moisture, and above average boundary layer height (Table 3.2). Similar findings on the relationship between associated local meteorology and synoptic-scale circulations that produced poor air quality were also noted in Sydney (Hart, De Dear et al. 2006).

Further assessment of occurrences when a large anticyclone was situated over or near the Melbourne area (Figure 3.2) identified that archetypes *43* and *42* were found to most significantly influence PM_{10} with notable increases also occurring for NO_2 (Figure 3.5). In regards to air shed composition, this suggests that Melbourne likely experiences more frequent periods of increased PM_{10} and increased NO_2 than periods with increased O_3 and NO_2 (Figure 3.3). This finding of increased pollution is in agreement with other works as synoptic circulation patterns that exhibit high anticyclonicity have been noted as important elsewhere (Leighton and Spark 1997; Triantafyllou 2001; Jacob and Winner 2009). Local conditions under these modes exhibited slightly cooler temperatures, light winds, and low boundary layer height, which indicate stable conditions (Table 3.2).

Circulation patterns that exhibited situations with a deep trough south of the Australian continent resulted in the largest decreases for all pollutants (Figure 3.4).

This is most evident for archetype *00*, *01*, and *02* that were found to govern periods under which pollutant concentrations were lowest in Melbourne (Figure 3.5). These synoptic types can be characterized by a strong cyclone in the Southern Ocean with the influence of low pressure extending north of the Melbourne region. The increased pressure gradients are suggestive of an approaching cold front from the south that likely exhibits a southwesterly gradient wind that clearly results in a cleansing of the Melbourne air shed (Figure 3.2). Local conditions exhibited cool temperatures, strong westerlies, and low radiation (Table 3.2).

Overall, the effect of synoptic-scale circulations on pollutant concentrations can be summarized as follows: anticyclones result in increased pollution when they are located over and to the east of Melbourne; cyclones to the south result in decreased pollution. The differences in concentrations under these conditions range by up to 30 and 40% for each pollutant. All other modes were found to influence pollution but their impacts were of lesser importance.

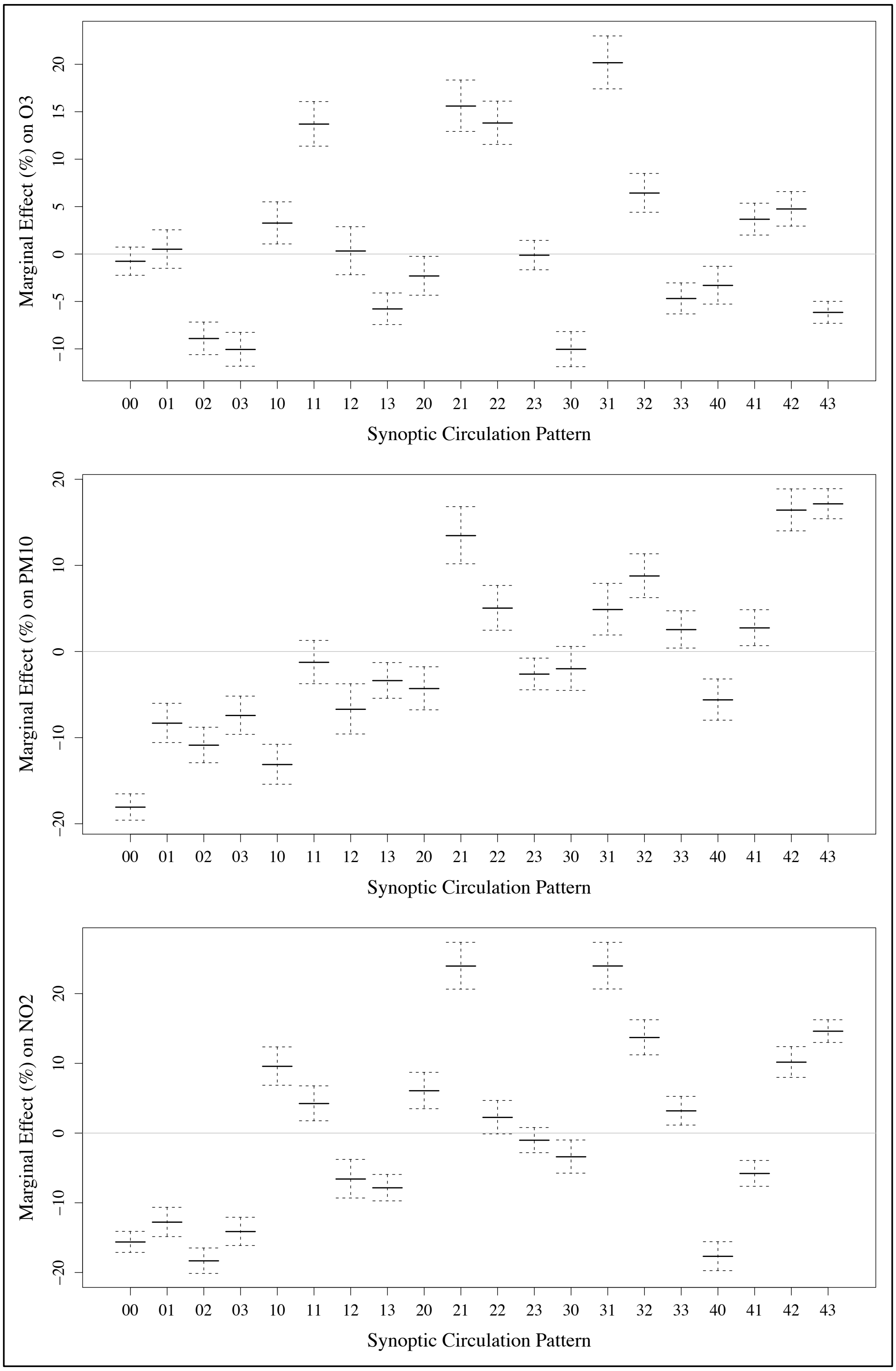

Figure 3.5: The estimated marginal effect (%) along with 95% confidence intervals for each synoptic type on pollutant concentrations.

c. Technical Approach

The use of the SOM technique proved an effective and reliable means for generating the synoptic climatology of our study region. The results proved to be a robust and straightforward medium to examine the relationships between the expected range of synoptic patterns for a region and local air pollution. We realize that the use of 20 synoptic patterns for an air pollution study is larger than normal and that further generalization is possible by considering the similarities between adjacent nodes and grouping those nodes within a specified minimum distance. However, the size of our SOM was beneficial as it allowed us to identify subtle changes in large-scale circulation patterns along with the variations in their effects on each air pollutant. This facilitated a stronger representation and understanding of how the entire atmospheric continuum influences air pollution across Melbourne. Further research suggestions for the use of SOMs in air quality research include: (1) investigating how patterns transition from one state to the next perhaps elucidating how 'sequences' influence air quality, (2) exploring how changes in long-term frequency affect air pollution perhaps leading to improved understanding of the impacts of potential climate change, and (3) evaluating the use of generated synoptic climatologies as a forecasting tool.

The major benefit of using GAM for this analysis was the ability to isolate the effect of synoptic circulations on air pollution by removing several confounding effects (our base model) inherent in the data. This is an important and often overlooked issue as air quality data are extremely complex and susceptible to bias (Thompson, Reynolds et al. 2001). In short, GAMs produced easily interpretable, robust results from a highly complex data set. Perhaps the most obvious limitation to the analysis is the lack of inclusion of delayed effects. Nonetheless, we decided not to include delayed effects in our models as pollution events in Melbourne are of extremely short duration rarely lasting more than a few hours at a time (NEPC 2005).

The importance of these effects would likely vary from one air shed to the next. Another limitation of the approach was the use of day-of-the-week as a surrogate for emissions patterns in the region. A more precise measure of emissions would likely improve models and resulted in a more effective separation of the effect from the synoptic estimates.

3.6 CONCLUSION

This study has demonstrated that combined use of SOM and GAM provided insight into the nature of circulation forcing on air quality that is not easily accessible through traditional methodologies. More specifically, using SOM allowed an assessment of the entire circulation continuum on air quality and GAM facilitated this assessment after removing significant confounding properties inherent to the data. Because of this approach these findings provide a robust observational foundation for the response of O_3, PM_{10} and NO_2 in Melbourne to synoptic-scale circulations. Overall, we found that the air pollutant response to changes in synoptic-scale circulations was somewhat modest -- identifying large-scale features as a secondary driver of local air quality. However, notable significant differences were found to occur in air pollution under specific synoptic types. The effects produced were found to be as much as 20% above mean values.

Overall, these results provide strong evidence to the behavior of air pollution under varying atmospheric circulation processes and general insight into the role of large-scale meteorology. For Melbourne, these findings suggest that if changes in synoptic behavior occur they may only have a minor effect on air quality. However, increases in the intensity and frequency of the synoptic conditions that were particularly degrading to air quality would most certainly have an adverse effect. This is an important finding regarding the potential effect of climate change on air quality. Unfortunately, this is only part of the story for regional air quality in Melbourne as more pronounced responses are expected to be driven by local-scale meteorological variables. In light of this knowledge gap, a study focusing on how local meteorological conditions affect air pollutant concentrations in Melbourne is planned.

3.7 ACKNOWLEDGEMENTS

The authors are grateful to Sean Walsh for important contributions to this study. This study was supported through research funds provided by the Environmental Protection Authority Victoria and Monash University. Neville Nicholls involvement was supported by the Australian Research Council through Discovery Project DP0877417

CHAPTER 4

QUANTIFYING THE INFLUENCE OF LOCAL METEOROLOGY ON AIR QUALITY USING GENERALIZED ADDITIVE MODELS

Material in this chapter is reproduced from a research article published in Atmospheric Environment. References have been moved to a consolidated bibliography.

Declaration by candidate for Thesis Chapter 4

In the case of Chapter 4, the nature and extent of my contribution to the work was the following:

Nature of contribution	Extent of contribution (%)
Formulation of research problem and the context of the research in the wider literature; data acquisition and analysis; interpretation of results and writing.	95%

The following co-authors contributed to the work:

Name	Nature of contribution	Extent of contribution (%)
Jason Beringer	Formulation and revision of writing	2%
Neville Nicholls	Formulation and revision of writing	1%
Rob J Hyndman	Statistical guidance	1%
Nigel J Tapper	Formulation and revision of writing	1%

Candidate's Signature	Date
[redacted]	6 July 2011

Declaration by co-authors

The undersigned hereby certify that:

(7) the above declaration correctly reflects the nature and extent of the candidate's contribution to this work, and the nature of the contribution of each of the co-authors.

(8) they meet the criteria for authorship in that they have participated in the conception, execution, or interpretation, of at least that part of the publication in their field of expertise;

(9) they take public responsibility for their part of the publication, except for the responsible author who accepts overall responsibility for the publication;

(10) there are no other authors of the publication according to these criteria;

(11) potential conflicts of interest have been disclosed to (a) granting bodies, (b) the editor or publisher of journals or other publications, and (c) the head of the responsible academic unit; and

(12) the original data are stored at the following location(s) and will be held for at least five years from the date indicated below:

Location(s): [1]School of Geography and Environmental Science, Monash University; [2]Department of Econometrics and Business Statistics, Monash University

Name	Signature	Date
Jason Beringer[1]	[redacted]	8 July 2011
Neville Nicholls[1]	[redacted]	18 July 2011
Rob J Hyndman[2]	[redacted]	18 July 2011
Nigel J Tapper[1]	[redacted]	18 July 2011

TITLE: QUANTIFYING THE INFLUENCE OF LOCAL METEOROLOGY ON AIR QUALITY USING GENERALIZED ADDITIVE MODELS

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Keywords: air pollution, climate change, generalized additive models, and meteorology.

Published in Atmospheric Environment 45 (2011) 1328-1336

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4.1 ABSTRACT

This paper presents the estimated response of three pollutants, ozone (O_3), particulate matter ≤ 10 µm (PM_{10}), and nitrogen dioxide (NO_2), to individual local meteorological variables in Melbourne, Australia, over the period of 1999 to 2006. The meteorological-pollutant relationships have been assessed after controlling for long-term trends, seasonality, weekly emissions, spatial variation, and temporal persistence using the framework of generalized additive models (GAMs). We found that the aggregate impact of local meteorology in the models explained 26.3% of the variance in O_3, 21.1% in PM_{10}, and 26.7% in NO_2. The marginal effects for individual variables showed that extremely high temperatures (45 °C) resulted in the strongest positive response for all pollutants with a 150% increase above the mean for O_3 and PM_{10} and a 120% for NO_2. Other variables (boundary layer height, winds, water vapor pressure, radiation, precipitation, and mean sea-level pressure) displayed some importance for one or more of the pollutants, but their impact in the models was less pronounced. Overall, this analysis presents a solid foundation for understanding the importance of local meteorology as a driver of regional air pollution in Melbourne in a framework that can be applied in other regions. This paper presents an improved display method where the effects across the range of the covariate on each pollutant were quantified on a percentage scale. Such presentation facilitates easy interpretation across covariates and models. Finally, our results provide a clear window into how potential climate change may affect air quality.

4.2 INTRODUCTION

It is well known that concentrations of gases and aerosol particles within local air sheds are affected by weather (Elminir 2005; Beaver and Palazoglu 2009). This understanding has led the air quality community to recognize that air pollution is an area sensitive to potential climate change. In an effort to provide those responsible for air quality management with potential 'what if' scenarios, a growing body of research on assessing the impacts of a changing climate on regional air quality has developed (Jacob and Winner 2009; EPA 2009). This increased scrutiny of air quality has highlighted that there are many aspects of air pollution that are still difficult to understand. One of these aspects is the estimation of the sensitivity of air pollutants to individual meteorological parameters. This has proven particularly challenging for several reasons (EPA 2009). First, meteorological parameters are inherently linked, resulting in strong interdependencies, for example, the dependency of boundary layer height on surface temperature or the link between surface temperature and radiation. These associations make separating the effects of individual parameters a highly complex task. Secondly, meteorological parameters can affect pollutants through direct physical mechanisms such as the relationship with radiation and ozone or indirectly through influences on other meteorological parameters such as the association between high temperatures and low wind speed (Jacob and Winner 2009). Thus, multiple approaches are necessary to understand the true nature of meteorological-pollutant relationships. To further complicate matters, the magnitude and nature of these effects can vary from one air shed to the next as well as across seasons, making site specific assessments necessary for understanding local responses (Dawson, Adams et al. 2007a; EPA 2009).

Statistical modeling is one approach that can be used for addressing the effects of meteorology on air pollution (Camalier, Cox et al. 2007). Statistical models are well suited for quantifying and visualizing the nature of pollutant response to individual meteorological parameters as they directly fit to the patterns that arise from the observed data (Schlink and Herbarth 2006). However, statistical techniques do not aim to fully describe the formation and accumulation of air pollutants in their chemical, physical, and meteorological processes (Schlink, Herbarth et al. 2006). In order to obtain a robust understanding for these aspects of air quality a combined approach including deterministic models is suggested (Jacob and Winner 2009). That being said, statistical modeling is a widely used, effective learning tool for a variety of air quality applications (Thompson, Reynolds et al. 2001; Schlink, Herbarth et al. 2006). Furthermore, non-linear statistical approaches have been shown to effectively describe the complex relationship between meteorological variables and air pollution (Thompson, Reynolds et al. 2001). Unfortunately, summarizing non-linear associations beyond a graphical display has often proved difficult and provided little information that is interpretable to the general public (Thompson, Reynolds et al. 2001). In the context of climate change impacts on air quality it has been suggested that statistical studies are most capable of providing insight into the potential impacts through development of observational foundations (Jacob and Winner 2009). These foundations provide a window into the possible extent of climate change impacts on air quality (Camalier, Cox et al. 2007).

This study aims to provide such an observational description for Melbourne, Australia. The city of Melbourne, with a population of approximately 3.9 million (ABS 2010), is situated on Port Phillip Bay at the south-eastern edge of the continent in close proximity to the Southern Ocean at 37° 48’ 49” S 144° 57 47” E (Figure 4.1). The climate can best be described as moderate oceanic, with occasional incursions of intense heat from Central Australia, and the city is famous for its highly changeable weather conditions (BOM 2009). Locals like to declare that Melbourne weather typically observes ‘four seasons in one day’. While Melbourne’s air pollutant levels are relatively low (Table 4.1) when compared to other urban centers of similar size, the city is subjected to a wide range of meteorological conditions that present an interesting opportunity for analysis (Murphy and Timbal 2008). With increasing population growth and urbanization in the Melbourne region there will be added pressures on air quality, which may result in less favorable conditions in the future. This will be superimposed upon the predicted effects of climate change.

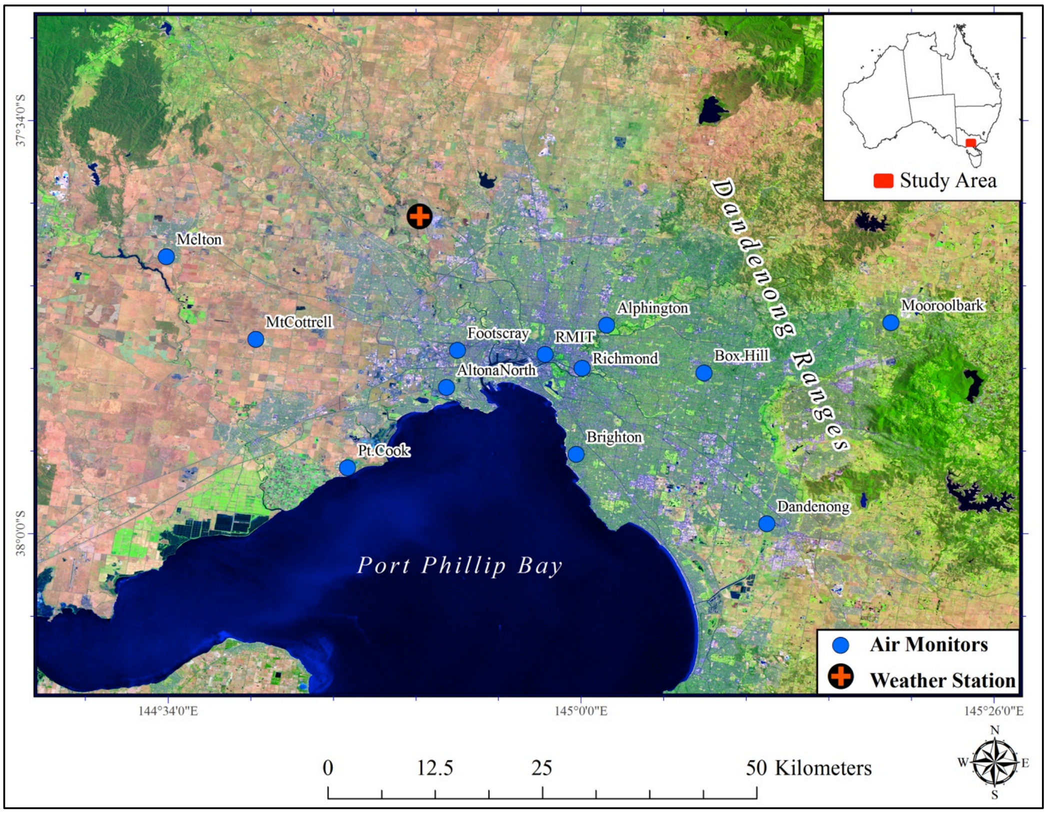

Figure 4.1: Map of air monitoring stations and meteorological station in study region.

The objective of this research is to quantify the magnitude in which regional air pollutants respond to local meteorology in Melbourne, Australia. This was achieved using the framework of generalized additive modeling (GAM) to estimate the response of ozone (O_3), particulate matter ≤ 10 μm (PM_{10}), and nitrogen dioxide (NO_2) to individual local meteorological variables. The meteorological-pollutant relationships have been assessed after controlling for long-term trends, seasonality, weekly emissions, spatial variation, and temporal persistence. The nature of the response of each pollutant to individual meteorological variables is presented using partial residual plots described on a percentage scale as marginal effects.

4.3 DATA

a. Local Meteorological Data

Links between air pollutants and local weather conditions were made using daily automatic weather station observations for site number 086282 (Melbourne International Airport) for the period of 1999 to 2006. This site is located at 37° 40' 12" S and 144° 49' 48" E with an elevation of 113 m and was chosen because a comprehensive range of measures are collected consistently over time. Variables provided by the Australian Bureau of Meteorology included:

- Maximum daily temperature (°C)
- Mean sea-level pressure (hPa)
- Global radiation (MJ/m^2)
- Water vapor pressure (hPa)
- Zonal (*u*) and meridional (*v*) wind components (km/hr)
- Precipitation (mm).

Additionally, boundary layer height (BLH) was taken from the ERA-Interim reanalysis using the location of 37° 30' 0" S and 145° 30' 0" E for 4 p.m. LST - the approximate time of maximum boundary layer depth. The ERA-Interim reanalysis is produced by the European Centre for Medium-Range Weather Forecasts (ECMWF) and is discussed in more detail by Uppala, Dee et al. (2008).

b. Air Pollutant Monitoring Data

Local air pollution data were provided by the Environmental Protection Authority Victoria taken from the Port Phillip Bay air monitoring network (Figure 4.1). Pollutants included ozone (O_3), particulate matter ≤ 10 μg (PM_{10}), and nitrogen dioxide (NO_2). O_3 and NO_2 concentrations are reported in parts per billion by volume (ppb) and were measured using pulsed fluorescence chemiluminescence and ultra violet absorption techniques. PM_{10} concentrations were measured using photospectrometry and are reported in micrograms per cubic meter ($\mu g/m^3$). This analysis uses the daily maximum value for 8-hr O_3, the 24-hr mean value of PM_{10}, and the daily maximum value for 1-hr NO_2 from all available monitoring locations over the period of 1999 to 2006 (Table 4.1). These timeframes were selected to parallel air quality objectives in the State Environment Protection Policy for ambient air quality (SEPP 1999). Additionally, days on which significant air quality events not associated with meteorology (i.e. bushfires) were known to have occurred and data below 5 ppb for O_3 and NO_2 and 3 $\mu g/m^3$for PM_{10} were removed. This removed 1.6% of O_3, 0.2% of PM_{10}, and 3.4% of NO_2, respectively.

Table 4.1 Summary of data used for model development.

Variable	Units	Mean	Median	Min	Max	SD	Definition	Source
O_3	ppb	21.8	21.0	4.0	102.0	9.5	Daily 8-hr Max	EPA
PM_{10}	$\mu g/m^3$	17.2	15.6	2.0	279.7	8.2	Daily Avg	EPA
NO_2	ppb	18.4	19.0	4.0	90.0	12.0	Daily 1-hr Max	EPA
Temperature	°C	20.0	18.9	8.6	44.6	6.3	Daily Max	BOM
Sea level pressure	hPa	1017.2	1017.4	991.5	1038.8	7.3	Daily Avg	BOM
Global radiation	MJ/m^2	15.3	13.3	0.1	36.6	8.8	Daily Sum	BOM
Vapor Pressure	hPa	10.8	10.3	4.6	24.5	2.6	Daily Avg	BOM
Zonal (u) wind	km/hr	-4.3	-2.7	-33.2	13.6	6.8	Daily Avg (N+,S-)	BOM
Meridonal (v) wind	km/hr	3.7	1.4	-30.6	51.9	15.6	Daily Avg (E+,W-)	BOM
Precipitation	mm	1.3	0.0	0.0	138.8	5.0	Daily Sum	BOM
Boundary Layer Height	m	1433.0	1343.0	195.0	4937.0	618.8	4 p.m. LST	ERA
Long-term Trend	Days	--	--	1.0	2922.0	--	values from 1:2922	--
Day of week	Days	--	--	0.0	6.0	--	values from 0:6	--
x coordinate	dd.ddd	144.97	--	144.57	145.33	--	Longitude	EPA
y coordinate	dd.ddd	-37.84	--	-37.99	-37.71	--	Latitude	EPA

4.4 METHODS

a. Generalized Additive Modeling

Generalized additive models (GAMs) are regression models where smoothing splines are used instead of linear coefficients for covariates (Hastie and Tibshirani 1990). This approach has been found particularly effective at handling the complex non-linearity associated with air pollution research (Dominici, McDermott et al. 2002; Schlink, Herbarth et al. 2006; Carslaw, Beevers et al. 2007). The additive model in the context of a concentration time series can be written in the form (Hastie and Tibshirani 1990):

$$log(y_i) = \beta_0 + \sum_{j=1}^{n} s_j(x_{ij}) + \varepsilon_i \tag{4.4.1}$$

where y_i is the ith air pollution concentration, β_0 is the overall mean of the response, $s_j(x_{ij})$ is the smooth function of ith value of covariate j, n is the total number of covariates, and ε_i is the ith residual with var(ε_i)= σ^2, which is assumed to be normally distributed. Smooth functions are developed through a combination of model selection and automatic smoothing parameter selection using penalized regression splines, which optimize the fit and make an effort to minimize the number of dimensions in the model (Wood 2006). Interaction terms, e.g. $s(x_1, x_2)$, can also be modeled as a thin-plate regression spline or a tensor product smooth. The choice of the smoothing parameters is made through restricted maximum likelihood (REML) and confidence intervals are estimated using an unconditional Bayesian method (Wood 2006). This analysis was conducted using the *gam* modeling function in the R environment for statistical computing (R Development Core Team 2009) with the package ‘mgcv’ (Wood 2006).

c. Model Development

The first step in the selection of individual models for O_3, PM_{10}, and NO_2 was to fit a preliminary base model. This was fit to each pollutant in order to control for the seasonality, persistence, spatial trend, and weekly emissions patterns that exist in these data. Following model (4.3.1) the preliminary model can be written as:

$$\log(y_i) = \beta_0 + s(time) + s(dow) + s(long,\ lat) + s(y_{i-1}) + \varepsilon_i \tag{4.4.2}$$

where *time* is a number between 1 and 2922 (representing each day in the study period) included to account for long-term trends and seasonality, *dow* is a number ranging from 0 to 6 included to account for day-of-the-week, *long* and *lat* are the spatial coordinates of each monitor location included to account for spatial trend, and y_{i-1} is a one day lag term included to account for short-term temporal persistence. It is important to note that the residual spatial variation is controlled by including a tensor product smooth, $s(long,\ lat)$, in the model and a smooth function of the preceding day's pollutant concentration, $s(y_{i-1})$, was included to control for autocorrelation in residuals. Additionally, since air pollution data are known to be seasonal, a predetermined smoothing parameter of k=32 (one knot (k) for each of the four seasons over the study period) was used for the construction of the spline function for *time*. The motivation for this control is that function should represent a relatively symmetric cyclic pattern in the data. To check the adequacy of our methods for controlling for space-time effects, box-plots and time-series plots of residuals by monitor location were examined. No violations of assumptions (i.e. residuals that were normally distributed and did not exhibit serial correlation) were obvious in any pollutant.

Final models were chosen using forward selection where each of the meteorological variables was added to the base model upon which Akaike's Information Criteria (AIC) was evaluated. A variable remained in the final model if the fit yielded a lower AIC. Following model (4.4.1), the final model for each pollutant that can be written as:

$$\log(y_i) = \beta_0 + s(time) + s(dow) + s\,(long, lat) + s\,(y_{i-1}) + s\,(temp) + s\,(u) + s(v) + s(wvp) + s\,(rad) + s\,(precip) + s\,(blh) + \varepsilon_i \quad (4.4.3)$$

where *temp* is daily maximum temperature, *u* is the zonal wind component, *v* is the meridional wind component, *wvp* is water vapor pressure, *rad* is radiation, *precip* is precipitation and *blh* is the 4 p.m. boundary layer height. It is important to note that exploratory analysis included covariates not listed in Table 4.1. This included using winds over shorter periods, various measures of radiation, temperature, and atmospheric moisture. None of these refinements made any significant improvements to the models.

e. Characterization of Meteorological Effects

The explanatory powers of the final models specified above were measured using the R^2 statistic. The aggregate impacts of local meteorology on each pollutant are assessed by the difference in the R^2 of model (4.4.2) and model (4.4.3). Individual relationships between particular meteorological variables and each air pollutant are assessed using partial response plots.

It is well known that representing the full relationship between the response and the predictor in multiple regression models is difficult due to high dimensionality (Faraway 2005). Therefore we opted to use partial response plots to reveal the marginal relationship between each meteorological variable and each air pollutant (Faraway 2005). A partial response plot shows the static effect (i.e. effects that are stable over time) of a particular meteorological variable on a particular pollutant whilst accounting for the effects of all other explanatory variables in the model (Camalier, Cox et al. 2007). This effect is described as the marginal relationship between the response and the predictor because it represents the relationship while the effects of all other predictors have been accounted for in the data (Faraway 2005). In our case, the *y*-axis of each partial response plot has been centered to the mean value of the response and adjusted to a percentage scale. These proportional values are the marginal effects (Harrell 2001). The marginal effect can be interpreted as the change in pollutant response from the mean as the covariate of interest is varied. In short, the partial regression plot allows us to focus on the relationship between one predictor and the response in isolation from the effects of other predictors in the model (Faraway 2005). Representing the marginal effects as proportions scaled to the mean make it easy to compare effects across covariates and pollutants. The displayed marginal effects are given by 100 * [exp($s(x)$)-1], where x is the meteorological variable of interest, and $s(x)$ is the corresponding smooth function in model (4.4.3).

4.5 RESULTS AND DISCUSSION

a. Ozone

Ground-level ozone is classified as a secondary pollutant because it forms in the atmosphere when emissions of precursors such as volatile organic compounds (VOCs) and nitrous oxides (NO_x) react with sunlight (WHO 2006). Concentrations have been linked to atmospheric conditions such as the availability of solar ultraviolet radiation capable of initiating photolysis reactions, air temperatures, and concentrations of chemical precursors (EPA 2009). Research conducted across many settings suggests that increasing O_3 pollution is most strongly linked with increases in temperature (Jacob and Winner 2009).

In Melbourne, model (4.4.3) explained 69.9% of the variance of log transformed O_3 with the components of model (4.4.2) accounting for 43.6% and the aggregate impact of meteorological variables accounting for 26.3%. The most significant meteorological variable for O_3 was temperature ($F=462.9$, $p<0.001$) with

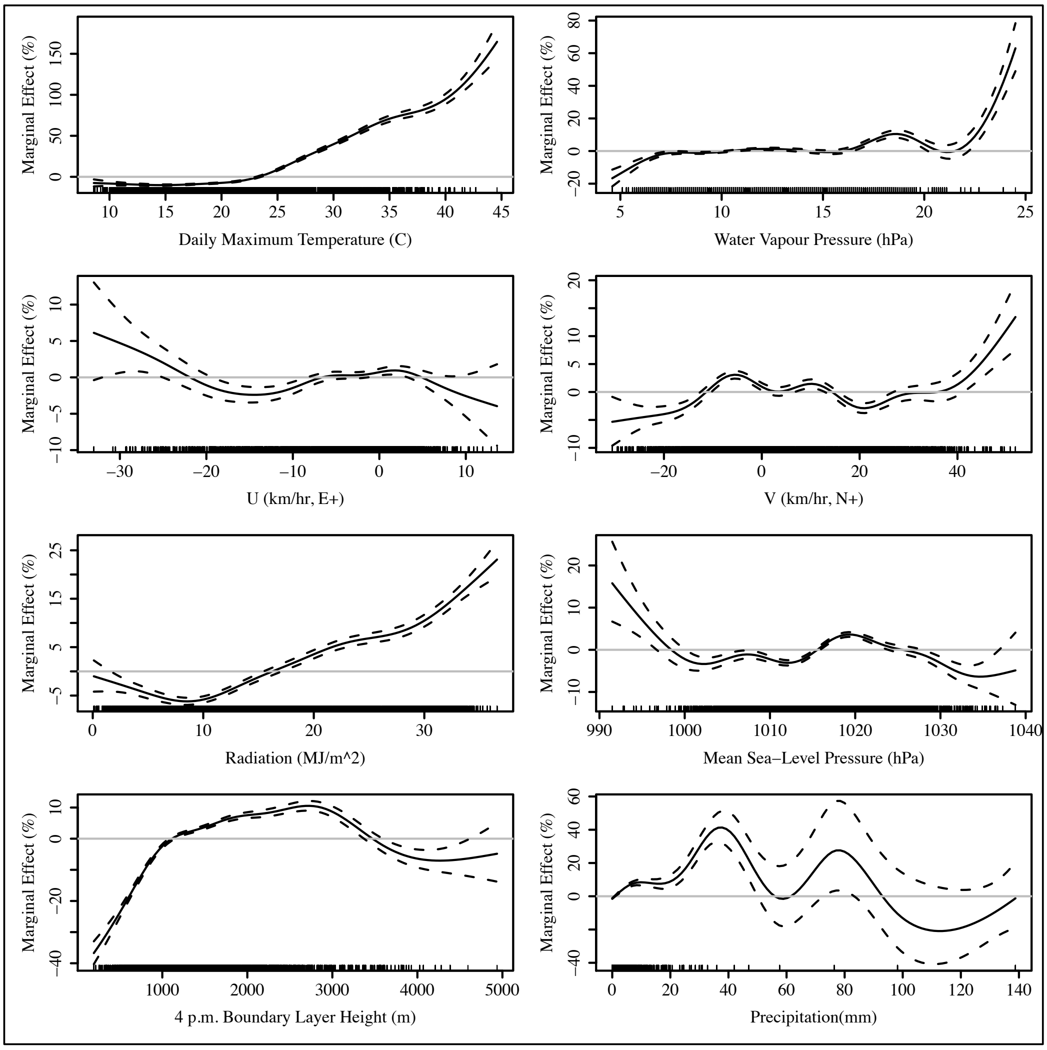

Figure 4.2: Partial response plots for O_3. The *y*-axis represents the marginal effect and the dashed lines are the 95% confidence intervals. The vertical lines adjacent to the *x*-axis represent the frequency of the data.

increased temperature being associated with increased ozone. This finding is most likely due to the role of temperature in the physical processes associated with ozone and its influence on local meteorology that affects air pollution. A partial residual plot (Figure 4.2) identified a positive non-linear relationship with marginal effects as great as 150%. This finding is in strong agreement with results from previous studies as increased temperatures have been shown to result in increased ozone in a variety of settings (Elminir 2005; Dawson, Adams et al. 2007a; Jacob and Winner 2009). A key finding here was that ozone concentrations were estimated to be 75-150% higher than average during the 92 days (3.5%) in the study period when daily maximum temperatures were in the range of 35 to 45 °C.

Water vapor pressure (F=27.5, p<0.001) was found to have little influence on ozone except when at the upper and lower extremes (Figure 4.2). Notably, increases of up to 80% were seen when water vapor pressure rose above 20 hPa. This positive response at high water vapor pressure is contradictory to findings that suggest that an increase in humidity suppresses ozone formation (EPA 2009). With the exception of the strong positive relationships at high water vapor pressures, similar results were noted by (Wise and Comrie 2005) for the dry climate of the southwestern United States. The estimated response for the zonal (u) wind component (F=5.6, p<0.001) in the model identified that increases up to 5% were expected when strong winds originated from the west and decreased with winds originating from the east. The response of ozone to meridional (v) wind (F=20.6, p<0.001) increased up to 15% under strong northerlies and decreased with winds from the south. The increase under northwest winds may be a result of inhibited local dispersion associated with a blocking of the bay breeze (Hurley, Manins et al. 2003). Weak winds have been associated with increased O_3 elsewhere (Dawson, Adams et al. 2007a). The effect of radiation (F=76.5, p<0.001) was found to be the strongest after values surpassed 20 MJ/m^2 as concentrations increased by as much as 25% (Figure 4.2). This relationship is consistent with the literature as radiation is a known driver in the photochemistry of ozone production (Dawson, Adams et al. 2007a). The response for mean sea-level pressure (MSLP) (F=27.5, p<0.001) found a slight increase in the marginal effects under low pressure (10%) and under moderate pressure (5%). The response was quite weak and is in relative agreement with other studies were MSLP has been found insignificant (Davis and Speckman 1999). The response of ozone to changes in boundary layer height (F=122.8, p<0.001) was found to be negative for heights below one kilometer where ozone decreased up to 40% (Figure 4.2). This negative effect is presumably due to an association with cold fronts that introduce clean air from the Southern Ocean into the Melbourne air shed. However, slight increases of up to 10% were shown between heights of one to three kilometers. The moderate relationship observed in Melbourne agrees well with findings from other empirical studies where the role of mixing depth has been shown to be rather limited (Jacob and Winner 2009). The response of ozone to precipitation ($F=38.5$, $p<0.001$) showed increases of up to 40% as precipitation levels were at or below 40 mm (Figure 4.2). After this threshold, confidence intervals increase in size and the relationship was generally negative. This is presumably due to wet deposition during heavy rainfall. The positive effect during light rainfall has been noted elsewhere and suggests that some degree of atmospheric moisture is beneficial to ozone production (Ordonez and Mathis 2005; Dawson, Adams et al. 2007a).

Overall, the strongest positive response for O_3 was found for high temperature with a maximum increase of 150%. Interestingly, this was followed by an 80% increase under extremely high water vapor pressure. More research is suggested to identify the mechanism behind this response. The strongest negative response occurred under low boundary layer heights where concentrations were found to decrease by as much as 40% below average.

b. Particulate Matter

Particulate matter consists of solid or liquid particles found in the air, including dust, pollens, soot, and aerosols from combustion activities (WHO 2006). Particles originate from a variety of mobile, stationary, and natural sources, and their chemical and physical compositions vary widely. Furthermore, PM can be emitted directly or can be formed in the atmosphere when gaseous pollutants such as SO_2 and NO_x undergo transformation to form secondary organic particles. This complexity has been highlighted in studies showing that the chemical and physical composition of PM varies depending on location, source, time of year, and meteorology (EPA 2009). A review of current research by Jacob and Winner (2009) found that observed correlations of PM concentrations with meteorological variables have been found to be inconsistent (direction depends on composition) and are generally weaker than for ozone. This indicates that the relationship with particulate matter is more complicated than with gaseous pollutants and that dependencies are likely to vary from one air shed to the next.

In Melbourne we found that model (4.4.3) explained approximately 57.8% of the variance of log transformed PM_{10} with the components of model (4.4.2) accounting for 36.7% and the aggregate impact of meteorological variables accounting for 21.1%. Daily maximum temperature ($F=265.6$, $p<0.001$) was identified as the most significant meteorological variable and increasing temperatures corresponded with increasing PM_{10} (Figure 4.3). The nature of the response was similar to the findings for ozone (particularly when a threshold of 35 °C was surpassed) as resulting concentrations were 100 to 150% higher than average. It is important to note that this finding contradicts results from model perturbation studies (Dawson, Adams et al. 2007b). However, some North American studies have stated that a positive response may be driven by increases in the sulfate component or black carbon of PM due to faster SO_2 oxidation (Jacob and Winner 2009). This seems unlikely to be the case in Melbourne as research has found that PM in Australian cities is of very low sulfur content (Chan, Cohen et al. 2008). More research is suggested in order to identify the mechanism behind this response. Water vapor pressure was also found to be quite significant ($F=143.4$, $p<0.001$) where increases as great as 30% were seen when values dropped below 10 hPa (Figure 4.3). This finding is similar to findings in other areas where crustal/soil dust is an important source of regional PM (Wise and Comrie

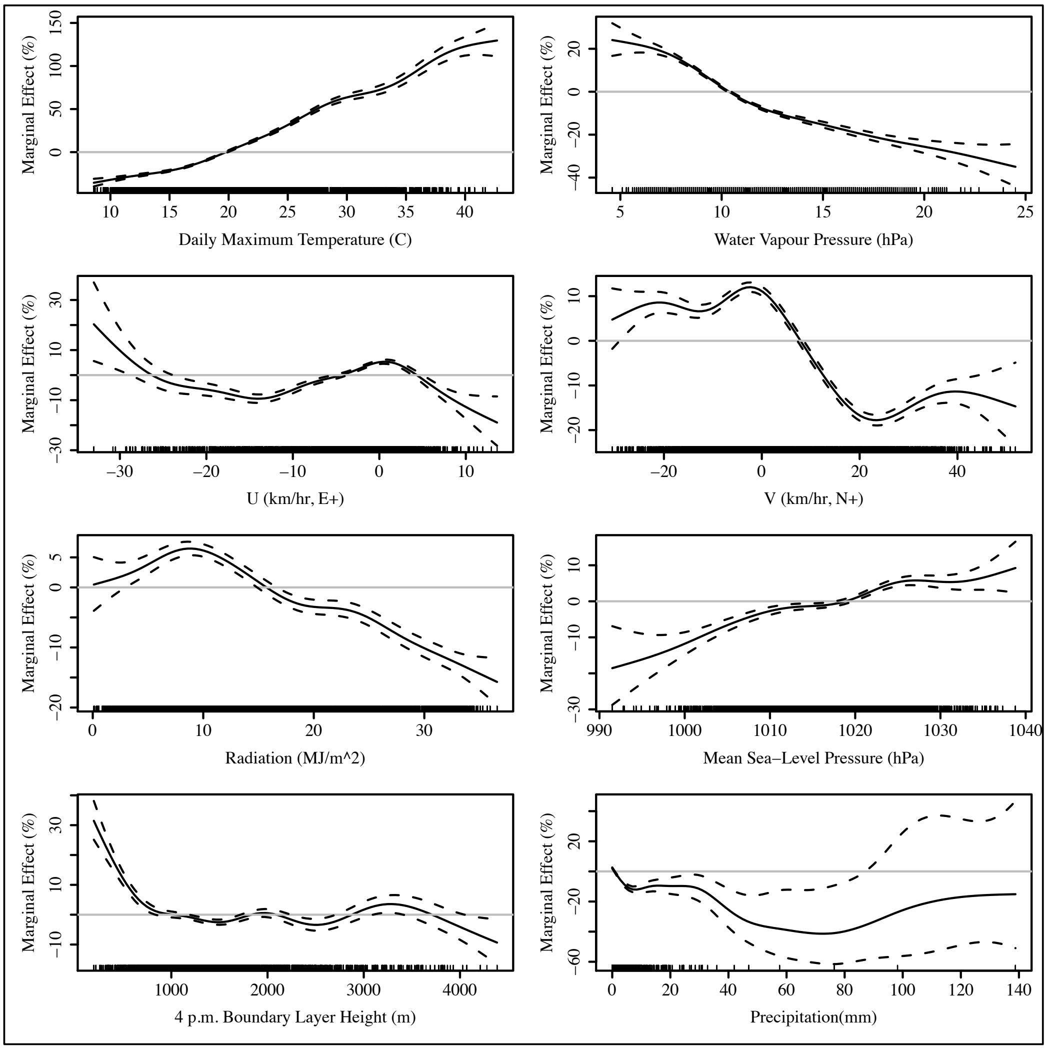

Figure 4.3: Partial response plots for PM_{10}. The *y*-axis represents the marginal effect and the dashed lines are the 95% confidence intervals. The vertical lines adjacent to the *x*-axis represent the frequency of the data.

2005). The response of PM_{10} to the zonal (*u*) wind component (F=34.1, $p<0.001$) indicated that under strong westerly winds concentrations increased by up to 20%. Meridional (v) wind (F=139.4, $p<0.001$) was also found to be quite significant with a 20% decrease occurring under strong northerly winds (Figure 4.3). The increase of PM_{10} under strong westerlies is most likely due to an increased contribution of regional dust and the decrease observed under strong northerlies is most likely the result of increased dispersion. Furthermore, slight increases also occurred under relatively light to stable winds showing that transport related PM can build up in the region. Other studies have noted the positive effect of stable conditions on PM in urban environments (Jacob and Winner 2009). Particles slightly increased (5%) under low levels of radiation (F=34.6, $p<0.001$) that suggests increases during periods of increased cloudiness and cooler months. The effect of mean sea-level pressure (F=19.9, $p<0.001$) shows that low pressures result in decreases up to 5% while increases of up to 10% were seen as pressures rose above 1020 hPa. This is most likely due to the strong association of high pressure with stability (EPA 2009). The nature of the response of PM_{10} to the 4 p.m. boundary layer height (F=22.6, $p<0.001$) showed a 30% increase for heights below one kilometer and a decrease above this height. Dawson, Adams et al. (2007b) also noted a similar response for low boundary layer heights stating that decreased dispersion was a likely factor. Increased precipitation (F=25.6, $p<0.001$) was found to have a negative effect on particle concentrations (Figure 4.3). This finding is in agreement with other work since the role of precipitation in wet deposition is well known (Dawson, Adams et al. 2007b; Jacob and Winner 2009).

Overall, the strongest positive response of PM_{10}, like O_3, was under high daily maximum temperatures as concentrations were up to 150% higher than average. The second largest increase (30%) was under low boundary layer heights. The largest decreases were associated with increased precipitation (60%) and increased water vapor pressure (40%). Relatively stable winds had a much lesser effect than anticipated indicating that dust is likely a major source of particles for the region.

c. Nitrogen Dioxide

Nitrogen dioxide is a reddish brown toxic gas that forms when nitric oxide emissions from automobiles and power plants react with oxygen in the atmosphere (WHO 2006). In the urban environment levels of NO_2 have been found to be strongly associated with emissions from vehicles and have also been found to contribute to the secondary formation of O_3 and fine particle pollution (EPA 2009). While less research has focused on the meteorological links for NO_2 it has been found that local dispersion and temperature play important roles (Carslaw, Beevers et al. 2007).

In this study, model (4.4.3) explained 56.3% of the variance of log transformed NO_2 with the components of model (4.4.2) accounting for 29.6% and the aggregate impact of meteorological variables in the model accounting for 26.7%. Increases in daily maximum temperature ($F=227.7$, $p<0.001$) were found to correspond with increases in NO_2 (Figure 4.4). Temperatures below 20 °C resulted in a 20% decrease and temperatures above 40°C resulted in a maximum increase of 120%.

This finding agrees with results from a single site in a multiple site study in Oslo, Norway, where a positive response was noted for temperatures across the range of 5 to 25 °C (Aldrin and Haff 2005). This may be partially explained by the influence of temperature on evaporative emission rates or the association between temperatures and other meteorological variables important to NO_2. Further research using deterministic models is suggested. The response of NO_2 to water vapor pressure ($F=77.7$, $p<0.001$) was similar in nature to the response for PM_{10} as increases up to 20% were shown for pressures below 10 hPa. As water vapor pressure increased above 10 hPa concentrations exhibited decreases. The small effect of relative humidity seen here was also noted by Aldrin and Haff (2005) and suggests that atmospheric moisture had relatively little influence on NO_2. The response of NO_2 to the zonal (u) wind ($F=150.7$, $p<0.001$) showed up to a 40% decrease under strong westerly winds and a slight increase under stable conditions (Figure 4.4). Meridional (v) winds ($F=589.1$, $p<0.001$) were found to be the most significant meteorological variable in the model with the response showing a 60% decrease under strong winds (Figure 4.4). An increase of up to 20% was shown for conditions that were stable. Stable conditions likely result in the buildup of local emissions within the Melbourne air shed as Carslaw, Beevers et al. (2007) also noted wind as the most significant meteorological predictor for traffic related NO_2. The response to radiation (F=50.8, p<0.001) exhibited a modest negative relationship where high levels resulted in a regional decrease of up to 20% (Figure 4). Low mean sea-level pressure (F=20.4, p<0.001) resulted in up to a 10% decrease while high pressure showed up to a 10% increase. This is most likely explained by increased stability during periods of high pressure. The response to the 4 p.m. boundary layer height (F=9.3, p<0.001) showed that concentrations decreased up to 30% as the boundary layer rose (Figure 4.4). Increased dilution within the boundary layer is the likely mechanism. A positive response to light precipitation (F=10.9, p<0.001) was identified although it should be interpreted cautiously as confidence intervals are rather large.

Overall, the strongest positive response for NO_2 occurred under high temperatures when concentrations increased by as much as 120%. This was followed by precipitation although confidence intervals are quite broad likely due to a low frequency of occurrence. The largest decrease in concentrations was shown for u and v wind components as strong winds resulted in a 60% decrease below the mean. Water vapor pressure also had a negative effect as increased values resulted in a decrease of up to 40%. The degree to which NO_2 responded to local meteorology – particularly temperature and wind, was greater than expected. The findings here suggest that local meteorology is of the same magnitude of importance for NO_2 as it is for O_3 and PM_{10} in the Melbourne air shed.

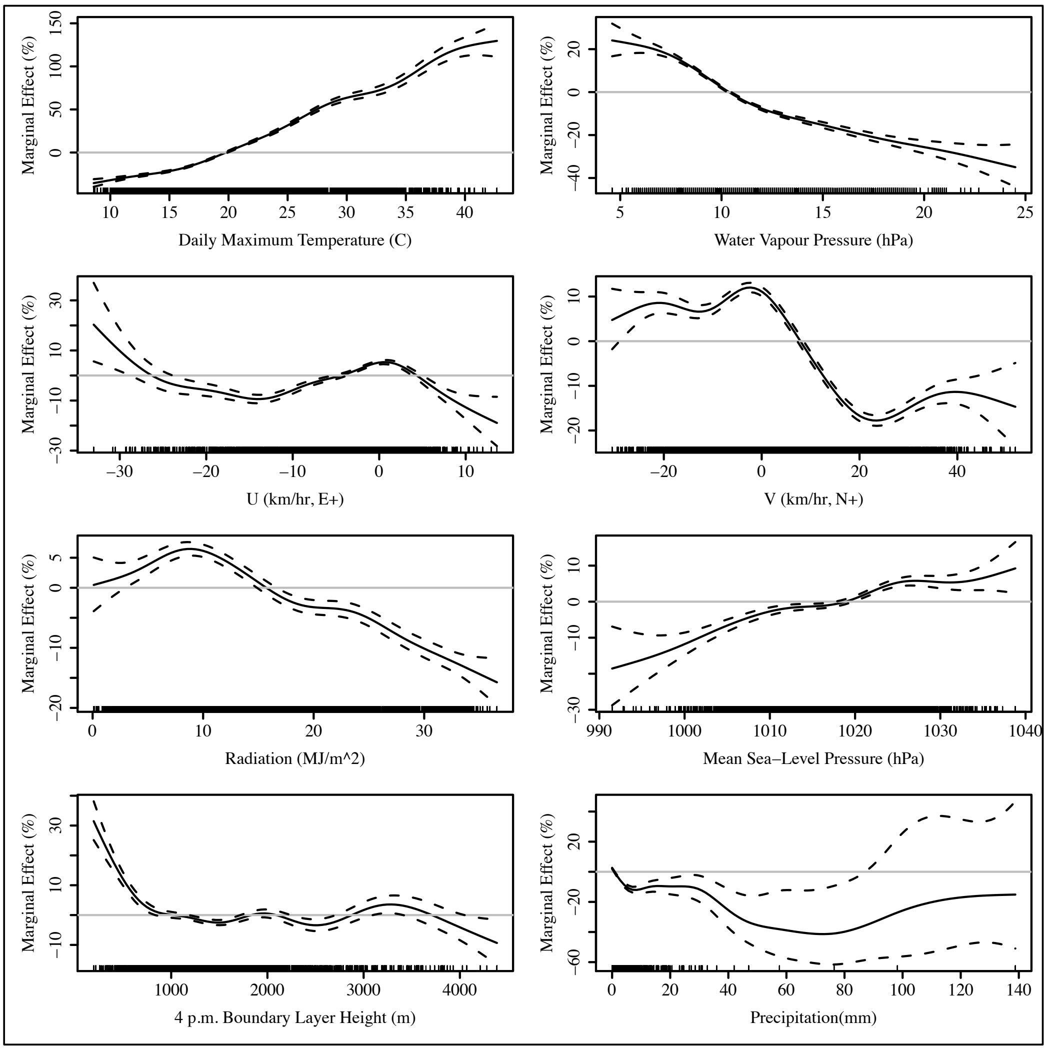

Figure 4.4: Partial response plots for NO_2. The *y*-axis represents the marginal effect and the dashed lines are the 95% confidence intervals. The vertical lines adjacent to the *x*-axis represent the frequency of the data.

d. Technical Approach

The use of GAM in combination with partial residual plots and marginal effects proved an effective and insightful way to characterize the relationships between individual meteorological variables representing local weather and air pollution. Complex non-linear dependencies were not only able to be visualized for each response, but their effects across the range of the covariate were also able to be quantified on a percentage scale. This quantification provides an expansion upon previous analysis by facilitating easy interpretation across covariates and models, which is especially important for communicating results to non-specialized audiences.

Although our approach did not consider the physical, meteorological and chemical processes in detail, the results produced were plausible and comparable to other studies. Furthermore, results produced are based on observational data eliminating the uncertainty associated with interpreting responses based on forecasts.

Perhaps the greatest limitation of the current work is the omission of interaction terms (ex. *s* (*temp, v*)) in the models. Not including these terms may have resulted in the underestimation of the overall impact of local meteorology on air pollution. However, it could also be stated that due to the interdependency of meteorological variables, interactions may be accounted for by a single dominant variable. In our case, this variable is most likely temperature. GAMs are quite capable of handling complex interactions and further research of models that include interactions is suggested. Minor improvements in this approach might include improved spatial resolution of all meteorological data as conditions in this paper are treated as being spatially uniform. However, this error is likely minimal as temperatures collected at the Melbourne Airport and each EPAV monitoring station during the study period indicated spatially homogeneous variation for temperature across the region as all correlation coefficients were above 0.96. Moreover, the statistical averages of daily maximum temperatures for each location were within 1 °C. Additionally, the inclusion of more sophisticated emissions data would likely improve model fit and therefore result in more accurate assessments of meteorological variables.

e. Potential Impacts of Climate Change

The Australian Greenhouse Office, using climate change projections developed by the Australian Bureau of Meteorology and the Australian Commonwealth Scientific and Research Organisation (CSIRO) for the city of Melbourne, anticipates a future air environment that exhibits increased temperatures, decreased moisture, and decreased wind speeds (DPCD, 2008). Notably, a projected increase in the number of days above 35 °C on the magnitude of 25% (~3 days) by 2030 and 50-100% (~7-14 days) by 2070 is also expected (DPCD, 2008). If such projections hold true, this study provides evidence from observational data that the air environment in Melbourne will become more conducive to poorer air quality under the current level of emissions. Our results confirm a statistically significant association between increasing pollutant concentrations and increasing temperatures. Therefore, it appears that increasing temperatures, particularly across the range of 35 to 45°C, will cause increases on the magnitude of 150% for O_3 and PM_{10} and 120% for NO_2, assuming everything else remains equal. Relationships with wind indicate that if increased periods of stability occur in the future then increases of 10 to 20% in PM_{10} and NO_2 are likely to occur and if increased winds from the northwest occur then increases up to 15% in O_3 will likely result. It is important to note that this finding is representative of the overall regional response to wind, not individual monitor's response to local winds. Findings for water vapor indicate that if the future climate brings increasingly drier conditions, then PM_{10} and NO_2 are likely to increase by as much as 25%. Our findings for radiation suggest that periods of increased cloudiness would likely result in slight increases of up to 5% for PM_{10} and NO_2 while the opposite can be said for O_3 which could see reductions up to 5%. If precipitation decreases in the future then increases will likely be seen for PM_{10} and NO_2. Changes in mean sea-level pressure (at the local scale) are not likely to significantly impact any pollutant. These findings provide an observational window into how climate change may affect local air quality in Melbourne through changes in local meteorology, but further research using synergistic processed base air quality models is suggested.

4.6 CONCLUSION

The overall objective of this study was to develop observational relationships between locally measured individual meteorological variables and select air pollutants in Melbourne, Australia. Moreover, a statistical methodology is presented for achieving this objective and results are presented in a manner where the complexities of those relationships are easily compared and understood. In Melbourne, we found that local meteorological conditions most strongly affect the daily variation associated with O_3 and NO_2 followed closely by PM_{10}. The strongest effects for O_3 were related to temperature, boundary layer height, and radiation. The most significant variables for PM_{10} were temperature, wind, water vapor pressure, and boundary layer height. Temperature also displayed the strongest influence on NO_2 which was followed by wind and water vapor pressure. The remaining variables displayed some effect for each air pollutant, but the responses for these were less pronounced. These results can be used to determine the relative importance of local weather as a driver of regional air pollution as well as the marginal effects of individual meteorological variables. Furthermore, by presenting the percent change in air pollutant response across the range of individual meteorological variables, a clear window into how potential climate change may affect air quality is provided. This window suggests that a significant ‘climate penalty’ may need to be taken into account in order to achieve future air quality objectives.

4.7 ACKNOWLEDGEMENTS

The authors are grateful to Sean Walsh and Petteri Uotila for their important contributions to the data used in this study. This study was supported through research funds provided by the Environmental Protection Authority Victoria and Monash University. Neville Nicholls involvement was supported by the Australian Research Council through Discovery Project DP0877417.

CHAPTER 5

THE INFLUENCE OF TEMPERATURE ON AIR POLLUTION-MORTALITY RELATIONSHIPS IN MELBOURNE, AUSTRALIA

Material in this chapter is reproduced from a working paper under preparation for submission. References have been moved to a consolidated bibliography.

Declaration by candidate for Thesis Chapter 5

In the case of Chapter 5, the nature and extent of my contribution to the work was the following:

Nature of contribution	Extent of contribution (%)
Formulation of research problem and the context of the research in the wider literature; data acquisition and analysis; interpretation of results and writing.	95%

The following co-authors contributed to the work:

Name	Nature of contribution	Extent of contribution (%)
Jason Beringer	Formulation and revision of writing	1%
Neville Nicholls	Formulation and revision of writing	1%
Rob J Hyndman	Statistical guidance	1%
Martine Dennekamp	Revision of writing	1%
Nigel J Tapper	Formulation and revision of writing	1%

Candidate's Signature	Date
[redacted]	6 July 2011

Declaration by co-authors

The undersigned hereby certify that:

(13) the above declaration correctly reflects the nature and extent of the candidate's contribution to this work, and the nature of the contribution of each of the co-authors.

(14) they meet the criteria for authorship in that they have participated in the conception, execution, or interpretation, of at least that part of the publication in their field of expertise;

(15) they take public responsibility for their part of the publication, except for the responsible author who accepts overall responsibility for the publication;

(16) there are no other authors of the publication according to these criteria;

(17) potential conflicts of interest have been disclosed to (a) granting bodies, (b) the editor or publisher of journals or other publications, and (c) the head of the responsible academic unit; and

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Location(s): [1]School of Geography and Environmental Science, Monash University; [2]Department of Econometrics and Business Statistics, Monash University; [3]Department of Epidemiology and Preventative Medicine, Monash University

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Jason Beringer[1]	[redacted]	8 July 2011
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Rob J Hyndman[2]	[redacted]	18 July 2011
Martine Dennekamp[3]	[redacted]	18 July 2011
Nigel J Tapper[1]	[redacted]	18 July 2011

TITLE: THE INFLUENCE OF TEMPERATURE ON AIR POLLUTION-MORTALITY RELATIONSHIPS IN MELBOURNE, AUSTRALIA

AUTHORS: John L. Pearce[a*], Jason Beringer[a], Martin Dennekamp[b], Neville Nicholls[a], Rob J. Hyndman[c], and Nigel J. Tapper[a]

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SHORT TITLE: Influence of temperature on air pollution-related mortality

ARTICLE TYPE: Research Article

KEY WORDS: air pollution, Australia, case-crossover, effect modification, generalized additive models, interaction, mortality, temperature

5.1 ABSTRACT

If climate projections hold true, rising temperatures will present a growing challenge to health authorities in urban environments. One area of concern for air quality is the lack of information on the response of health outcomes to changing air pollutant environments under increasing temperatures. Therefore, the objective of this research is to identify if temperature plays a role in the air pollution-mortality relationship in Melbourne, Australia. In order to achieve this objective we: (a) examine the main effects of each pollutant, (b) examine the effect of a warm/cool season on each pollutant-mortality relationship, (c) use a bivariate air pollutant-temperature response surface to estimate the joint effect on mortality, and (d) stratify the effect of each air pollutant on mortality by temperature to examine heterogeneity across specific temperature ranges. Main, seasonal, and strata effects were estimated using case-crossover analysis (CCO). Joint effects were estimated using generalized additive Poisson regression models. Ozone (O_3), particulate matter ≤ 10 µm (PM_{10}), and nitrogen dioxide (NO_2) were the pollutants assessed and both techniques controlled for long-term trends, seasonality, day-of-the-week, influenza, and weather. Results suggest that temperature modifies the effect of air pollution on mortality. This modification was most evident under high temperatures where air pollutant effects were the greatest. Additionally, this modification seemed to be nonlinear – particularly for PM_{10} and O_3 suggesting that the local population could be more sensitive to air pollution exposure under high temperatures. This should raise concern in the air quality arena, particularly in regards to projected climate change.

5.2 INTRODUCTION

The adverse short-term effects of high temperature and high air pollution exposures on urban populations have been well documented (Samet JM 2000; Simpson, Williams et al. 2005; Basu 2009). These exposures are believed to heighten the risk of mortality and morbidity through increased pressure on the cardiovascular, respiratory, and thermoregulatory systems of the human body. Population subgroups that have shown a particular sensitivity to these exposures are individuals with pre-existing conditions, the elderly (those 65 years of age and greater), and the very young (those 2 years of age or less).

Historically, the effects of air pollution and temperature on populations have been identified using statistical techniques that attempt to isolate individual pollutant and/or temperature effects by treating all other variables in the models as confounding factors (Dominici, McDermott et al. 2002). In short, this approach removes the variance in mortality that can be explained by the confounding factors before estimating the effect of the variable of interest. For example, most air pollution studies remove the effect of weather (e.g., temperature, and humidity) before estimating the effect of the air pollutant on a specified health outcome. This approach is formulated on the hypothesis that the effects of air pollution on health outcomes are stationary across time and are therefore not dependent on other environmental conditions. This methodology has produced relatively consistent results around the globe and has contributed greatly to understanding how populations respond to air pollution (Samet JM 2000; APHEA2 2001; Dominici F 2002).

Nevertheless, a growing body of evidence examining the role of temperature on the air pollution-mortality relationship has generated concerns regarding scenarios of increased temperature associated with changes in climate (Roberts 2004; Ren, Williams et al. 2006; Hu, Mengersen et al. 2008; Ren, Williams et al. 2008; Stafoggia, Schwartz et al. 2008). In general, findings from these studies suggest that temperature acts as an 'effect modifier' in the air pollution-mortality relationship. In short, this means that the effect of air pollution on mortality is different under differing states of temperature. While this relationship is deemed biologically plausible (Gordon 2003), attributing the effects to an interaction has proven difficult (Stafoggia, Schwartz et al. 2008). Considering this, more research in the area is warranted.

Here we examine the role of temperature on the air pollution-mortality relationship in Melbourne, Australia. The city of Melbourne, with a population of approximately 3.9 million, is situated on Port Phillip Bay at the south-eastern edge of continental Australia in close proximity to the Southern Ocean at 37° 48' 49" S and 144° 57'47" E (Figure 5.1). Despite experiencing low levels of air pollution (Table 5.1), previous studies have found a positive association between mortality and air pollution in Melbourne (Simpson, Denison et al. 2000; Simpson, Williams et al. 2005). Interestingly, these studies found that the relationships were strongest in the warmer months (October to March) and that the largest effects were associated with NO_2 and O_3. Moreover, Nicholls et al. (Nicholls, Skinner et al. 2008) (although not controlling for air pollution) found that the average daily mortality of the elderly was 15-17% greater when daily average temperatures were above 30 °C. While these findings indicate that temperature and air pollution both significantly affect mortality in Melbourne we do not know if interactions exist. This study aims to fill this gap by further examining the role of temperature on the air pollution-mortality relationship in Melbourne.

5.3 METHODS

Data. The data used in this study are concurrent daily time series of mortality, weather, influenza, and air pollution from Melbourne, Australia over the years 1999 to

2006. Daily mortality data were provided by the Australian Bureau of Statistics (www.abs.gov.au) and are the aggregate counts of non-accidental daily deaths of individuals aged 65 years and over across the Greater Melbourne (Figure 5.1).

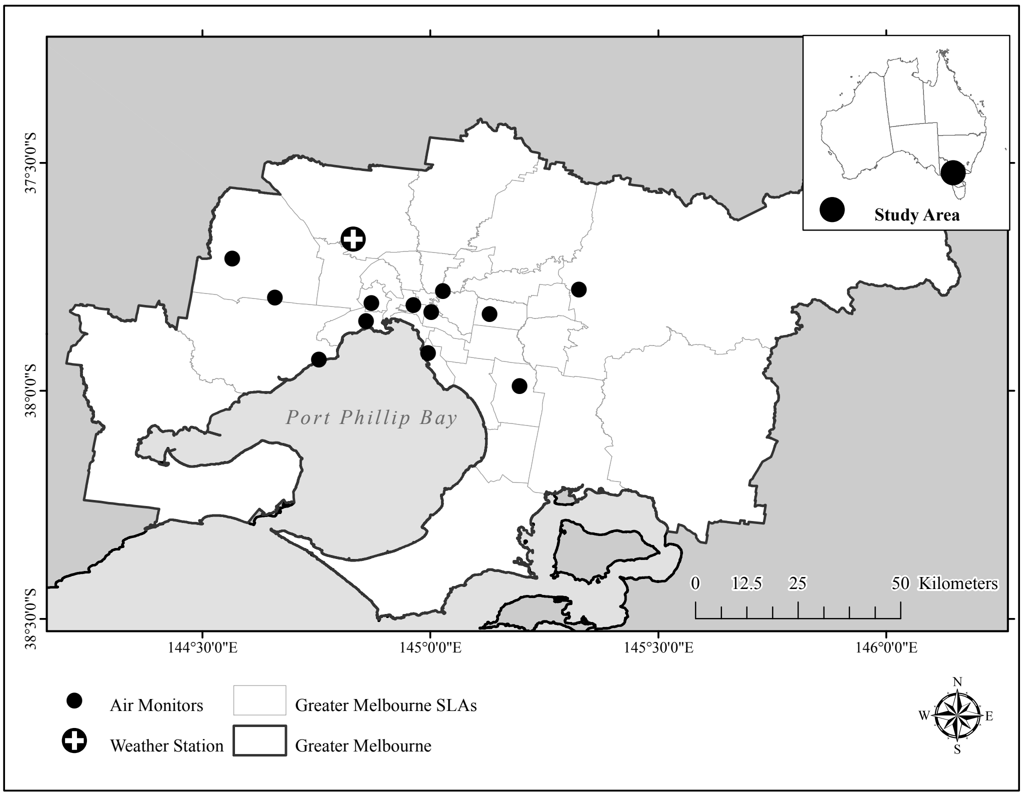

Figure 5.1: Map of study region.

Daily air pollution data were provided by the Environmental Protection Authority Victoria (www.epa.vic.gov.au) and consisted of the daily 8-h maximum O_3 (ppb) measured using chemiluminescence, 24-h average PM_{10} ($\mu g/m^3$) derived from hourly maximum values measured using tapered element oscillating microbalances (TEOM), and daily 1-h maximum NO_2 (ppb) concentrations measured using chemiluminescence taken from all monitors within the Port Phillip air monitoring network (Figure 5.1). An important aspect to consider when developing an air pollution exposure metric is the spatial heterogeneity that exists across the monitoring network. This becomes important when the presence/absence of a monitor on any given day presents bias in the metric due to spatial variability rather than true changes in exposure. To remove this bias, we developed our exposure metric for each pollutant as follows. First, the annual mean was computed for each monitoring location for each year, and subtracted from the daily observations from that monitor. This step removes the long-term spatial heterogeneity from the data set. After that, a 10% trimmed mean was calculated for each day using the daily deviances from each monitor. Using a 10% trimmed mean reduces the bias that can be present due to extreme values in the data set. The resulting values provide an exposure metric that represents the short-term variance in daily exposure without the influence of long-term spatial heterogeneity across the monitoring network and extreme values. Similar approaches have been used successfully elsewhere (Samet JM 2000; Schwartz 2000). Finally, if no observations were recorded for a day then it was dropped from the analysis, which occurred on less than five days for all pollutants.

Daily automatic weather station observations for maximum air temperature (°C), dew point temperature (°C), and mean sea-level pressure (hPa) were provided by the Bureau of Meteorology (www.bom.gov.au) for site number 086282 (Melbourne International Airport) (Figure 5.1). Finally, to take into account mortality associated with influenza outbreaks, this study used daily counts of influenza hospital admissions provided by the Department of Health Victoria (www.health.vic.gov.au).

Table 5.1. Descriptive statistics for daily mortality counts, pollutant, and meteorological measures during the years 1999 to 2006.

	Study Period					Warm Season (October to March)					Cool Season (April to September)				
	Mean	Median	Min	Max	SD	Mean	Median	Min	Max	SD	Mean	Median	Min	Max	SD
Deaths (n/day)[+]	47.9	47.0	14.0	80.0	8.2	45.5	45.0	21.0	68.0	7.4	50.2	50.0	14.0	80.0	8.2
PM_{10} (µg/ m^3)*	17.3	16.0	4.0	111.0	7.6	18.8	17.0	5.0	111.0	8.2	15.8	14.0	4.0	65.0	6.7
O_3 (ppb)**	22.2	21.0	5.0	92.0	8.8	25.7	23.0	9.0	92.0	10.1	18.8	19.0	5.0	38.0	5.3
NO_2 (ppb)***	22.4	22.0	5.0	75.0	8.0	19.7	18.0	7.0	75.0	8.2	25.0	25.0	5.0	61.0	6.8
MXT (°C)	20.2	19.0	9.0	45.0	6.4	24.1	23.0	12.0	45.0	6.0	16.3	15.0	9.0	35.0	3.8
DPT (°C)	7.8	7.0	-4.0	21.0	3.5	9.3	9.0	-4.0	21.0	3.6	6.3	6.0	-3.0	15.0	2.6
MSLP (hPa)	1017.2	1017.0	992.0	1039.0	7.3	1014.8	1015.0	993.0	1034.0	5.8	1019.6	1021.0	992.0	1039.0	7.8

Abbreviations: MXT, daily maximum temperature; DPT, dewpoint temperature; MSLP, mean sea-level pressure
*24-h average, **8-h maximum, ***1-h maximum, [+]Deaths for persons aged 65 years and above

Statistical analysis. The aim of the statistical analysis was to identify if temperature acts as an effect modifier in the air pollution-mortality relationship. In order to achieve this objective we: (a) examined the main effects of each pollutant, (b) examined the effect of a warm/cool season on each pollutant-mortality relationship, (c) used a bivariate air pollutant-temperature response surface to estimate the joint effect on mortality, and (d) stratified the effect of each air pollutant on mortality by temperature to examine heterogeneity across specific temperature ranges. Estimating main and seasonal effects provides a baseline from which comparisons can be made. Joint effects allow the most flexibility as no assumptions regarding the data are made and strata effects provide quantitative estimates that can be compared with other models. Similar approaches have been used in other investigations focused on the interaction between air pollution and temperature on mortality (Roberts 2004; Ren, Williams et al. 2006; Stafoggia, Schwartz et al. 2008).

Main and Seasonal effects. Case-crossover analysis (CCO) was applied to estimate the individual effects of air pollution on mortality and the effect of season on this relationship. This approach, first introduced by Maclure (1991), was chosen for this analysis as it has proven to be a viable alternative to time series methods for estimating the acute effects associated with short term air pollution exposures (Lu and Zeger 2007; Stafoggia, Schwartz et al. 2008; Dennekamp, Akram et al. 2010). We used a time stratified design with a stratum length of 28 days. Control days were also matched within each 28 day period by day-of-the-week. In short, this compares any given case day to control days occurring within a 28 day window on the same day-of-the-week. For example, mortality counts on Tuesday, 15 January 2002 would be compared to control days on the 1st, 8th, 22nd, and 29th of January 2002. By comparing cases to controls within this designated window, the method reduces confounding associated with long-term trends, seasonality, day-of-the-week, and weather.

To generate effect estimates, the CCO fits a Cox proportional hazards regression (also known as conditional logistic regression) to each pollutant where time dependent variables were incorporated using the counting process formulation of Anderson and Gill (Anderson and Gill 1982). Daily mortality was the outcome variable and the indicator of case/control day with each air pollutant being the exposure variable of interest. Further variables included linear terms for daily maximum temperature (lag 0-1), dew point temperature (lag 0-1), mean sea-level pressure (lag 0-1), and daily counts of influenza hospital admissions. These variables were included to further control for confounding of weather and influenza epidemics beyond the controlling effect of our 28 day window. The effect of the warm/cool season was assessed by including a season stratum variable in which the 'WARM' season for Melbourne included the months of October to March and the 'COOL' season included the months of April to September.

This analysis was conducted using the *casecross ( )* modeling function in the R environment for statistical computing within the 'season' package (Barnett and Dobson 2010; R Development Core Team 2010). All effects on mortality are presented in terms of percent risk (%) and can be interpreted as the percent change in daily mortality per 10-unit exposure. They are estimated by 100*(exp(β*10)-1), where β is the linear coefficient for an air pollutant.

Joint effects. The smooth air pollutant-temperature response surfaces were modeled using tensor product smooth functions within a generalized additive Poisson regression model (Wood 2006). As the time stratified CCO has been described as a special case of the time-series log linear model, this approach was deemed comparable with the CCO results (Lu and Zeger 2007; Stafoggia, Schwartz et al. 2008). The technique models the joint effect of air pollution and temperature on mortality as a continuous function of both variables with a strict nesting of the main effects (Wood 2006). More specifically, if Y_t is the total daily mortality count on day t, then Y_t's are Poisson distributed with expectation μ_t and with possible overdispersion θ. The general form can be written as:

$Y_t \sim \text{Poisson}(\mu_t)$

$\text{Var}(Y_t) = \theta\mu_t$

$$\log(\mu_t) = \alpha + FLU + s_1(time) + \gamma DOW_t + s_2(DPT_{t-i}) + s_3(MSLP_{t-i}) + te(AT_{t-i}, AP_{t-i})$$

where i refers to number of lag days; s(·) denotes the penalized regression splines; te(·) denotes the tensor product smooth; α is the overall mean; and γ is a vector of coefficients. Tensor product smooth's are deemed advantageous here as they do not assume isotropy like regression splines and have proven an effective means for scaling predictors relative to one another when both are operators in the same smooth but are measured in fundamentally different units (Wood 2006). The variables *DOW*, *FLU*, *DPT*, *MSLP*, *AT*, and *AP* refer to day-of-the-week, daily influenza counts, dew point temperature, mean sea-level pressure, air temperature, and the exposure metric for the air pollutant of interest, respectively.

In accordance with the literature, long-term trend and seasonality are accounted for using an estimated smooth function, *time*, on a numeric vector of calendar days with a scaling factor of seven degrees of freedom (df) per year (Samet JM 2000; APHEA2 2001; Dominici, McDermott et al. 2002). This essentially detrends the data in a manner that removes temporal signals larger than two months from the data. This analysis was conducted using the *gam ( )* modeling function in the R statistical environment with the *mgcv* package (Wood 2006).

Strata effects. To evaluate the effect of air pollution on mortality for different temperature strata we used a modified version of the CCO model to investigate the main and seasonal effects. This modification was the replacement of the main effect of temperature by the inclusion of a three level strata variable designed to partition the data by ranges of temperature corresponding to relatively low, mid, and upper ranges experienced across Melbourne. Previous studies have typically used some measure of percentiles as strata criteria (Roberts 2004; Ren, Williams et al. 2006; Stafoggia, Forastiere et al. 2006). This approach has been noted to be less than ideal as the choice of temperature ranges is somewhat arbitrary (Roberts 2004).

In order to improve upon this, we first investigated the sensitivity of the stratification model to the choice of cut points in order to identify a specific temperature at which the model performance is paramount. Simply put, this was achieved by iteratively running our stratification model while gradually shifting the second cutoff point of maximum temperature from 20 °C to 40 °C. To define the optimal temperature ranges for each pollutant we examined Akaike's information criteria (AIC) for each model fit across the range of cut point values. This approach, in addition to reporting quantitative air pollutant effect estimates for each temperature stratum, allows us to report the upper temperature ranges in which estimated effects are most effective in explaining mortality.

5.4 RESULTS

The average number of all-cause-non-accidental daily deaths in the elderly population for Melbourne during the study period was 48 persons per day with a minimum of 14 and a maximum of 80 (Table 5.1). Mortality counts were higher in the cooler months but short-term fluctuations were obvious for all seasons (Figure 5.2). A slight positive long-term trend was also visually apparent in the data and is consistent with trends in population growth (ABS 2010).

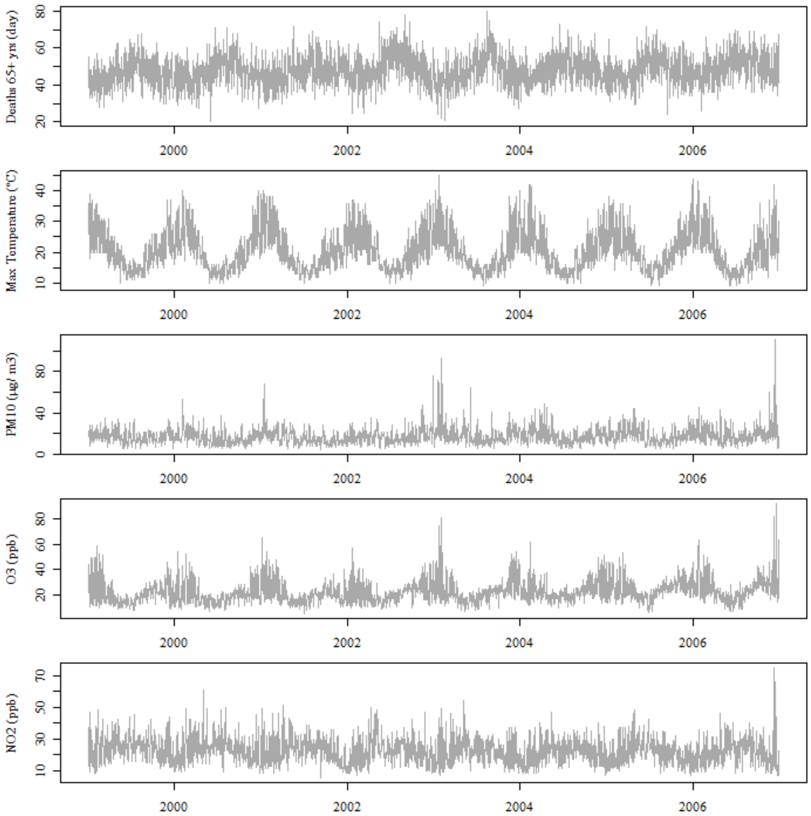

Figure 5.2: Time series of mortality, temperature, and air pollutants during the study period.

Air pollution concentrations during the study period were low with average concentrations of 22 ppb for 8 hr O_3, 17 μg/m^3 for 24 hr PM_{10}, and 22 ppb for 1 hr NO_2 (Table 5.1). However, exceedances of air quality standards (NEPC 2010) occurred on 108 days for PM_{10}, five days for O_3, and zero days for NO_2, respectively. The behavior of PM_{10} did not present an obvious seasonal pattern (Figure 5.2). Even so, the largest peaks did occur during the warmer months and are most likely the result of bushfires and dust storms. The pattern of O_3 was largely similar to temperature with peaks in the warmer months emphasizing the dependency of ozone formation on temperature (Figure 5.2). The pattern of NO_2 displayed moderate increases in the cooler months.

Temperature displayed a strong cyclic pattern with peaks typically occurring during the months of December through March. Pearson correlations showed that during the warm season air pollution and temperature were strongly correlated yet, during the cool season the correlation was substantially weakened – particularly for O_3 and NO_2 (Table 5.2).

Table 5.2: Pearson correlations among the daily pollutant measures stratified by season.

	PM_{10}	O_3	NO_2	MXT
PM_{10}	--	0.49	0.43	0.54
O_3	-0.02	--	0.59	0.71
NO_2	0.59	-0.23	--	0.51
MXT	0.46	0.29	0.22	--

MXT: daily maximum temperature
Warm season (Oct to Mar) is above diagonal; Cool season (Apr to Sep) is below

Main and Seasonal Effects

Results from CCO focusing on the main effects of each pollutant showed that a 10 unit increase during our study period resulted in no significant effects for all pollutants (Figure 5.3). However, seasonal CCO models found that effects (along with their 95% confidence intervals) were significant in the 'warm' season as mortality increased 2.21% [0.48%, 3.98%] per 10 ppb increase in O_3 and 1.61 [0.35%, 2.89%] per 10 μg increase in PM_{10}. NO_2 did not exhibit a significant effect in the 'warm' season. Moreover, effects during the 'cool' season were found to be insignificant for all pollutants (Figure 5.3). It is also important to note that in all pollutant models DPT, FLU, and MSLP were found to be insignificant.

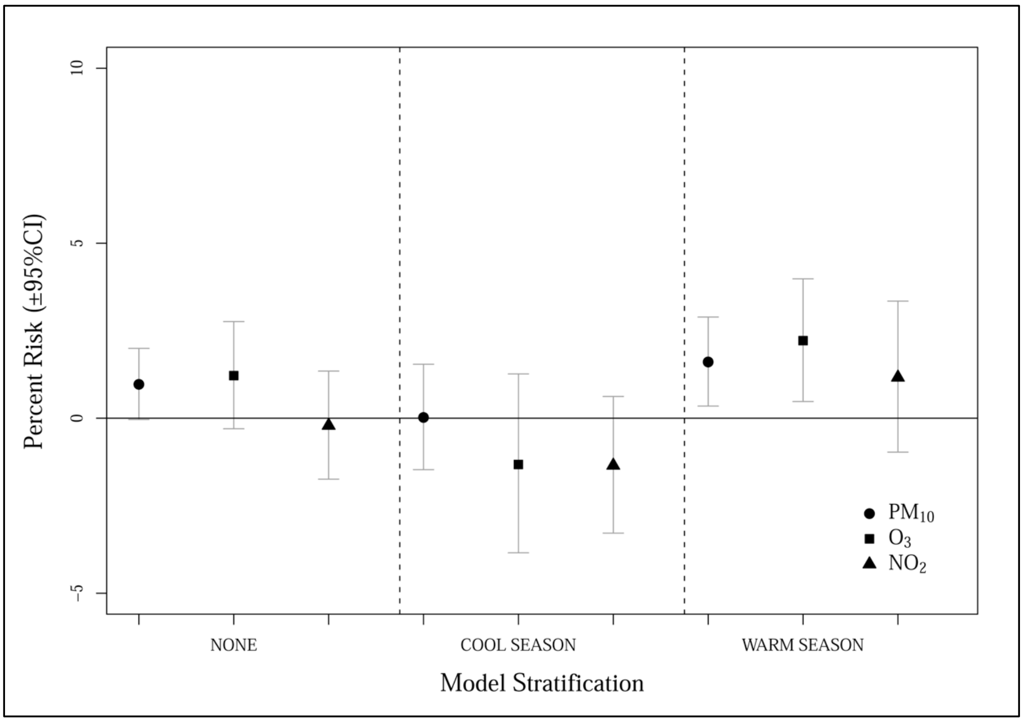

Figure 5.3: CCO estimated increases (and 95% confidence intervals) in daily mortality (65+ years) corresponding to a 10-unit increase in pollutant concentration for models without stratification (a.k.a. main effects) and with stratification by season.

Joint effects

The results of the Poisson regression suggest that the effects of air pollution on mortality vary over the range of temperature – with effects being largest in the upper ranges of temperature (Figure 5.4). The most prominent differences occur for PM_{10} and O_3 as considerable increases are found at high temperatures. Furthermore, these results demonstrate that the joint effect is nonlinear, which implies a synergistic relationship between temperature and air pollution. The relationship for NO_2 however, is far less pronounced and seems to exhibit a more linear-like trend. Similar to the previous CCO models, DPT and MSLP were again found to be insignificant. However, the covariate for FLU was found to be significant for all pollutants. The difference in the significance for FLU across techniques is likely due to differing approaches to handling temporality in the data.

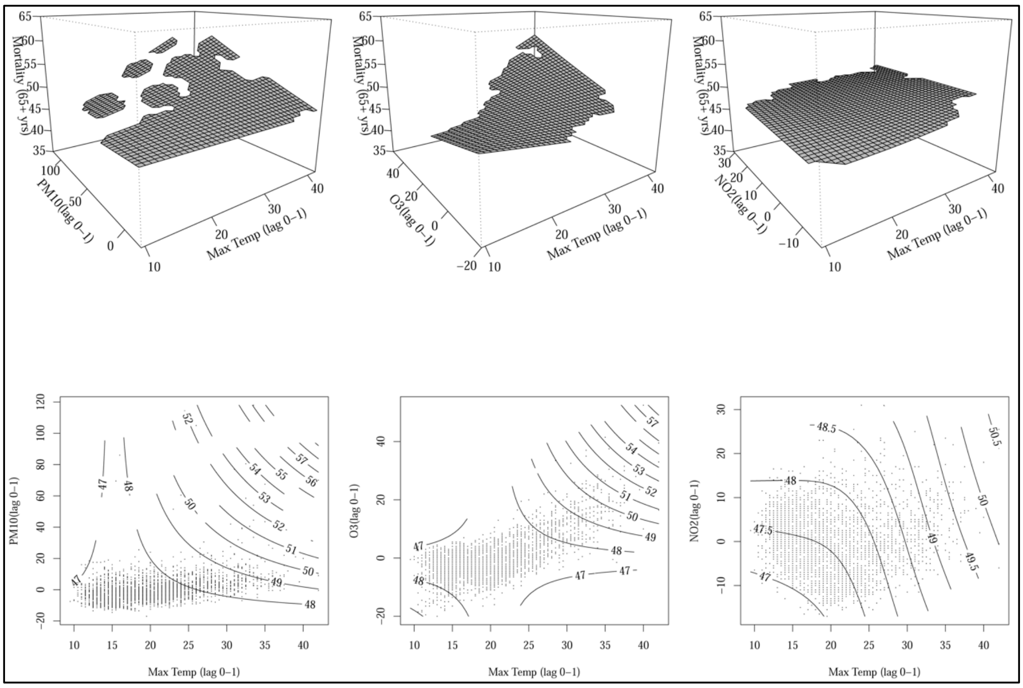

Figure 5.4: Poisson regression results using a bivariate tensor product smooth in the framework of a generalized additive model to estimate joint effects of temperature and air pollution on mortality. The upper row presents effect estimates using 3D perspective plots. The bottom row presents effect estimates as contours over the data distribution.

In order to test whether the bivariate air pollutant-temperature response surfaces are an improvement over the common additive approach (Peng, Dominici et al. 2006) - in our case replacing $te(AT_{t-i}, AP_{t-i})$ with $s_1(AT_{t-i}) + s_2(AP_{t-i})$, analysis of deviance was performed (Table 5.3).

The results showed that for O_3 a significant improvement in model fit occurred when a bivariate term was used. However, this improvement in model fit could be to the curvature of the pollutant effect, not effect modification. This was not the case for PM_{10} and NO_2. Even so, slightly lower deviance and degrees of freedom were reported by all bivariate models. Overall, bivariate surfaces suggest that as air pollution concentrations – particularly O_3 and PM_{10}, and temperatures increase, mortality increases.

Table 5.3: Analysis of deviance table comparing joint pollutant-temperature response surfaces to the traditional additive approach.

Pollutant	Model	Residual DF	Residual Deviance	Δ DF	Δ Deviance	X^2 p-value
PM_{10}	Additive	2862.4	3170.9			
	Bivariate	2861.5	3169.5	0.83	1.45	0.205
O_3	Additive	2866.1	3174.1			
	Bivariate	2866.1	3169.9	0.01	4.23	0.000
NO_2	Additive	2864	3173.30			
	Bivariate	2862.5	3171.9	1.54	1.42	0.409

Strata effects

Cut point analysis found that stratifying the effects of air pollution by various ranges of temperature resulted in changes in model fit (Figure 5.5). In short, this means that the ability of our model to explain mortality varies by the range of temperature used to estimate separate air pollutant effects. For PM_{10} and O_3, we found a threshold point of 28 °C maximum temperature at which our models resulted in the lowest AIC and for NO_2 a threshold of 30 °C was found (Figure 5.5).

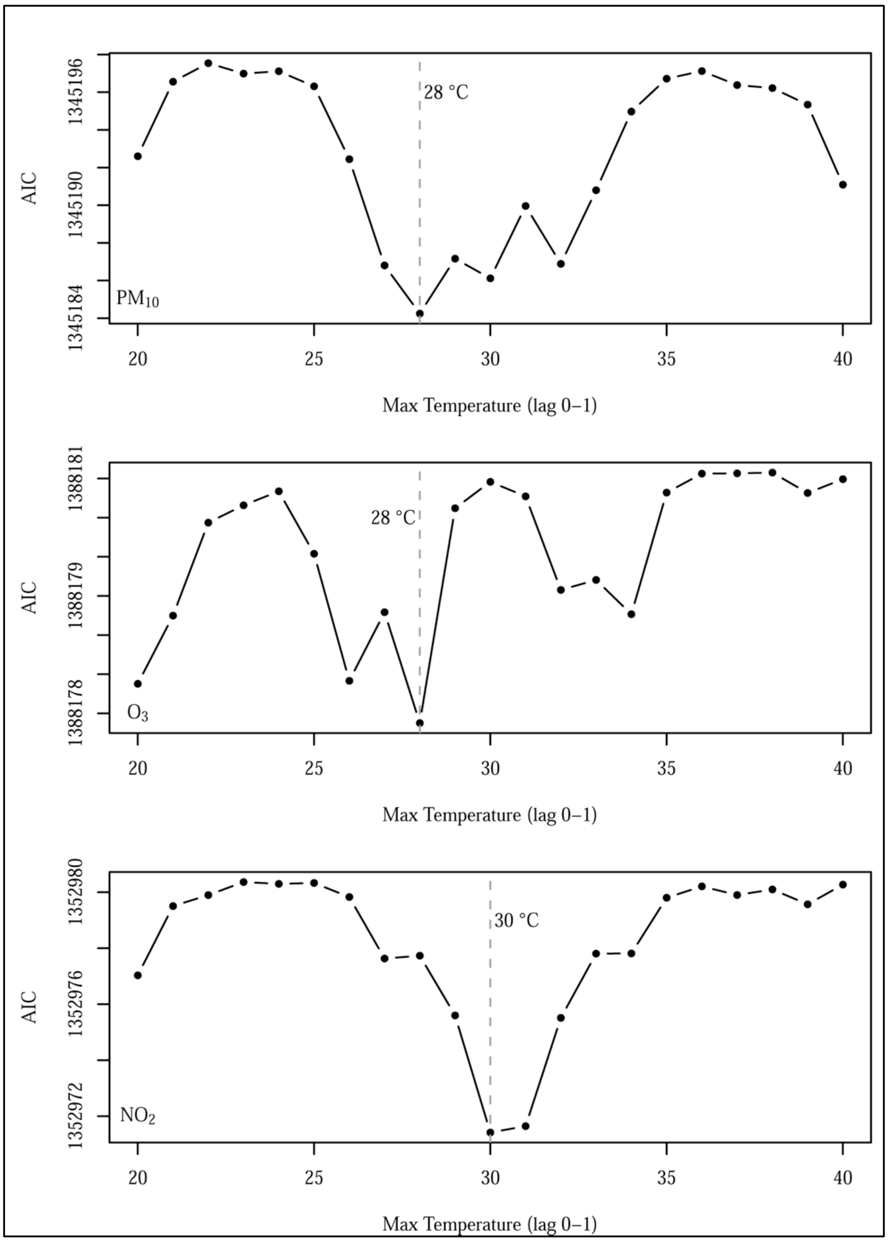

Figure 5.5: Results of cut point analysis using case-crossover analysis showing model performance based on the selected threshold for the upper temperature stratum. A lower AIC corresponds with a better model fit.

Therefore, a stratification criterion of LOWER = 9 to 16 °C, MID = 16 to 27 °C, and UPPER = 28 to 42 °C was implemented for PM_{10} and O_3; and a criterion of LOWER = 9 to 16 °C, MID = 16 to 29 °C, and UPPER = 30 to 42 °C for NO_2. The risk estimates (along with their 95% confidence intervals) from the temperature stratified CCO demonstrated that the air pollution effects were heterogeneous across specific temperature ranges, with the largest effect under the UPPER range of temperatures and the smallest effect under the LOWER range of temperatures (Figure 5.6). In the UPPER stratum, mortality increased 2.82% [0.84%, 4.85%] per 10 ppb increase in O_3, 3.14% [1.57%, 4.75%] per 10 µg increase in PM_{10}, and 5.05% [1.16%, 9.10%] per 10 ppb increase in NO_2. No significant effects were found in the LOWER and MID stratums. Like the previous CCO models, the covariates for DPT, FLU and MSLP were found to be insignificant.

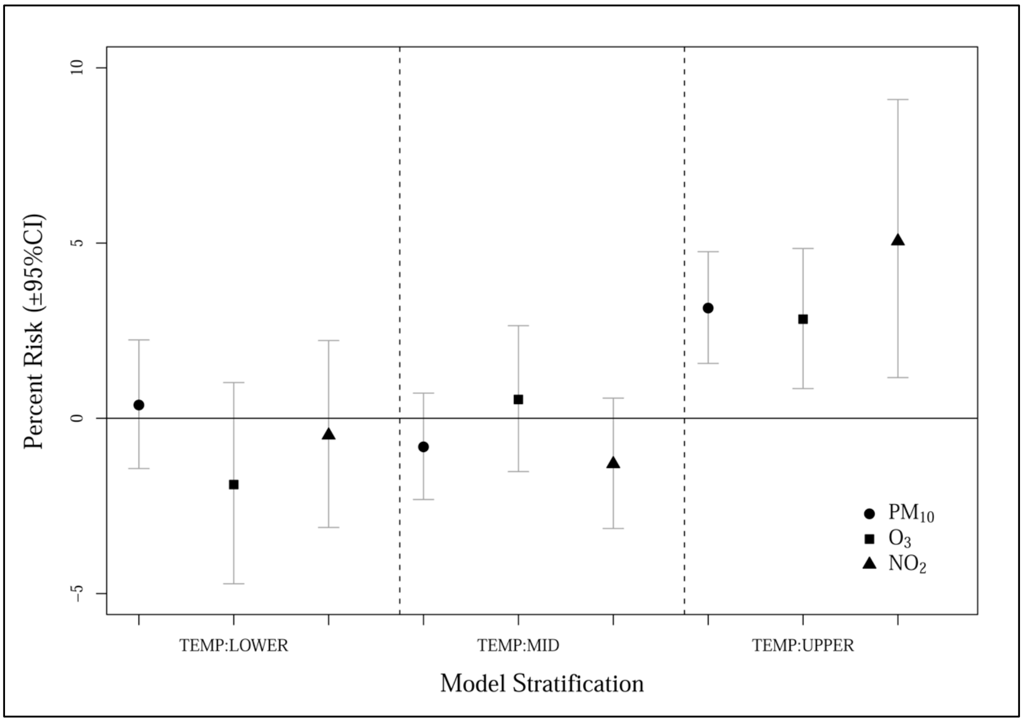

Figure 5.6: Case-crossover estimated increases (and 95% confidence intervals) in daily mortality (65+ years) corresponding to a 10-unit increase in pollutant concentration across three temperature stratums.

5.5 DISCUSSION

In this time series analysis, we used CCO and generalized additive Poisson regression to examine if temperature behaved as an effect modifier on the air pollution-mortality relationship in Melbourne, Australia. Our motivation was to explore the implications of increasing temperature on air pollutant health effects. To do this, we estimated the main, warm/cool season, joint, and temperature stratified effects of each air pollutant on mortality. Results suggest that increases in maximum daily temperature leads to an increased effect of air pollution on mortality (Figures 5.4 and 5.6). For example, main effects estimates for all pollutants were lower than effect estimates for the warm season (Figure 5.3). Moreover, effect estimates for the upper temperature stratum were higher than estimates for the warm season as a whole (Figure 5.6). This implies that a synergy exists between temperature and air pollution exposures and that the traditional design (i.e. assuming the effects of temperature and air pollutants are additive) may actually underestimate air pollutant health effects. This finding agrees with similar studies that examined the role of temperature as an effect modifier in the air pollution-mortality relationship (Roberts 2004; Ren and Tong 2006; Stafoggia, Schwartz et al. 2008). Therefore, it appears that when daily maximum temperatures in Melbourne are expected to be at or above 28 °C the risk for air pollution related mortality in the elderly is amplified. This finding implies that increases in temperature, be it due to a changing climate, the urban heat island effect, or natural climate variability, will have direct repercussions on air pollution related mortality.

It is clear that increases in mortality occur when temperatures are high and when air pollutant levels are high. However, is this effect the exclusive result of high maximum temperatures or increased air pollution concentrations or is it due to a combination of the two? Answering this question is not as simple as it may seem. Other researchers have addressed this question in other cities and their proposed explanations are (a) that synergism between the two exposures is biologically plausible because the response of the thermoregulatory system to heat stress can have direct or indirect effects on the entry of toxicants into the body (Gordon 2003), (b) that higher effects are due to increased exposure during warm periods (household windows kept open, more time outdoors) (Stafoggia, Schwartz et al. 2008) and (c) that the composition of the air environment at high temperatures is more toxic (Peng, Dominici et al. 2005).

In Melbourne, it is most likely that a combination of the proposed explanations is responsible for the increased effect of air pollution at high temperatures. We know that air pollutant concentrations in Melbourne respond positively to increasing temperature making the air environment more toxic during warmer periods (Pearce, Beringer et al. 2011). Additionally, it is very likely that personal exposure increases in the warmer months due to increased ventilation (many homes in Melbourne still lack air conditioning) and changes in outdoor activity patterns. What we do not know is if the biological toxicity of air pollution is enhanced as temperatures increase. This knowledge gap is of critical importance in protecting populations from air pollution – particularly during the warmer months. In summary, it is difficult to say that just because air pollution effects are higher during some seasons, and thereby some temperatures, than others that the cause is temperature.

There are several reasons why effect estimates could differ by season, and there are many factors that have seasonality besides weather. For example, there could be non-linearity in the exposure response relationship as the levels of pollutants can differ by season. More include: confounding by a co-pollutant with seasonal patterns, changes in indoor/outdoor activity patterns that affect exposure and thereby effect estimates. Additionally, the chemical composition of particles is known to differ by season, and there does exist evidence that seasonal variation in particles' effects relate to chemical composition, not necessarily due to weather.

One issue of concern when estimating the health effects of air pollutants is high correlation between air pollution and temperature (Table 5.2, Camalier, Cox et al.

2007; Carslaw, Beevers et al. 2007; Pearce, Beringer et al. 2011). We suggest that this can be particularly problematic when using traditional approaches that treat temperature as a confounding factor. This is because models that include highly collinear pairs of independent variables as predictors may sometimes have the effect of removing any significance for one independent variable even though that variable is an important driver. This occurs when variable A (e.g., air pollution) is so closely tied to variable B (e.g., temperature) that removing one essentially takes out all effect of the other. A typical example is ozone, as temperature is a well known driver in photo oxidant generation and concentrations are always higher in the warmer months. This may be partially responsible for the difficulty in identifying significant ozone effects at some study locations. That being said, in some locations A and B may be related but not highly collinear allowing the identification of the additional effect of A after controlling for the effect of B using the traditional approach. Therefore, we believe that interaction between temperature and air pollution should always be carefully considered when estimating environmental health effects.

Unfortunately, this study does have limitations. First, non-accidental-all-cause mortality was the health outcome used in all of our models. Because of the broad classification it is difficult to determine if environmental exposure is indeed the causative factor for deaths on any given day. Therefore, using daily counts of cardiovascular and respiratory related mortality would likely strengthen conclusions. Secondly, a distributed lag was not extensively assessed in this analysis. This may have resulted in the underestimation of effects as studies have shown that effects can be seen beyond the 48-hr window used in this study (Schwartz 2000). However, we do not believe that a substantial difference would be seen as other studies at our location have used the same lag period (Dennekamp, Akram et al. 2010; EPHC 2010).

Moreover, this analysis does suffer from limitations of sample size and lack of high pollution levels in the study area. Finally, in the development of our models we retained variables for dewpoint temperature, influenza, and mean sea-level pressure despite their level of significance. While this does conflict with the good practice of parsimonious modeling, risk estimates should not be affected.

In conclusion, we found that the effects of air pollution on daily mortality in the elderly differ by temperature in Melbourne. This was evident across multiple model approaches that found the highest expected daily deaths occurred under high temperatures and high air pollutant levels. In combination, these results suggest that increasing temperature may increase the sensitivity of the population to air pollution exposures. However, we cannot conclude that temperature is the cause behind the seemingly increased affect of air pollution at increased temperatures. These findings have direct implications for air quality management in the future.

5.6 ACKNOWLEDGEMENTS

The authors are grateful to Sean Walsh and Margaret Loughnan for their important contributions to the data used in this study. This study was supported through research funds provided by the Environmental Protection Authority Victoria and Monash University. Neville Nicholls involvement was supported by the Australian Research Council through Discovery Project DP0877417.

CHAPTER 6

CONCLUSION

6.1 INTRODUCTION

Air pollution and air pollution-related health effects are influenced by weather and are consequently expected to be sensitive to climate change. However, the degree and magnitude in which regional weather systems are affected by climate change is expected to vary geographically. To further complicate matters, the importance of meteorology as a driver of regional air pollution also varies geographically. Therefore, to begin to understand the issue for any given geographic region, site-specific studies are needed. Thus, the aim of this thesis was to do just that for Melbourne, Australia, by examining the nature and influence of weather on local air pollution and air pollution-related health effects in order increase understanding of the potential impacts of a changing climate.

Locally, the findings from this research are aimed at contributing to future planning directions and air quality management decisions for the assurance of a healthy living environment for Melbourne. Globally, these findings not only contribute to the understanding of climate change impacts on air quality but they also highlight the importance of meteorology as a driver of air quality in a relatively low pollution environment. Furthermore, the methodology presented here can also be replicated in other cities in order to provide information that could serve as a platform for discussing the need of conducting a more costly model simulation study.

6.2 SUMMARY OF RESEARCH

The objective of this thesis was achieved by performing three separate yet interrelated studies which focused on aspects of how weather influenced air pollution and how weather influenced the effect of air pollution on exposed populations. First, the effects of synoptic-scale circulations on three separate air pollutants – O_3, PM_{10}, and NO_2 were generated; showing how circulations across the entire range of expected synoptic conditions affected pollutant concentrations. Moreover, the explanatory power of synoptic conditions as a whole was presented. Then, an assessment of local-scale meteorology was conducted providing meteorological-pollutant curves for the typical range of variables used to describe local weather. These curves showed individual meteorological-pollutant relationships across the entire range of each element assessed without any linear assumptions and without the confounders of space-time, persistence, and other meteorological elements. Finally, the effects of air pollution on mortality as modified by temperature were presented. By doing this further evidence of how temperature is an important driver in all aspects for air quality was provided.

Taken as a whole, the research presented in this thesis identifies the important affect of meteorological changes on concentrations of O_3, PM_{10}, and NO_2 and their subsequent health effects. Several meteorological elements affected pollutant concentrations; however, changes in temperature were shown to cause the most substantial differences in concentrations. Moreover, temperature was also shown to influence air pollution related-health outcomes in a manner similar to its effect on concentrations. Thus, when temperatures increased over Melbourne, air pollutant concentrations increased as did the sensitivity of the local population to air pollution exposure. This critical finding provides evidence emphasizing the importance of temperature for air quality and potentially amplifying health effects.

6.3 IMPLICATIONS

The findings presented in this thesis, in combination with projections for future climate, suggest that it is reasonable to assume that climate change will exacerbate air quality problems in the future for Melbourne. For example, a projected increase in the number of days above 35 °C on the magnitude of 25% (~3 days) by 2030 and 50-100% (~7-14 days) by 2070 is expected for south-eastern Australia (DPCD, 2008). If this holds true, the relationships identified in Chapter 4 suggest that pollutant concentrations for O_3, PM_{10}, and NO_2 on these days have the potential to be at least 100% higher than the present day average. Additionally, the site-specific threshold identified in Chapter 5 clearly indicates that risk of mortality from air pollution exposure increases significantly when temperatures surpass 28 °C. Thus, when setting emission control standards for the future, it is advisable for air quality management to include climate knowledge into the decision making process. More specifically, this means that standards in the future will likely need to be even stronger to maintain air quality objectives because of the affect of increased temperatures on air pollution. This adverse affect from climate will work to counteract reductions in emissions – if they occur.

For example, the investigation of the response of air pollution to synoptic-scale circulation features demonstrated that air quality degrades in Melbourne under particular states of anticyclonic conditions and improves under particular states of cyclonic conditions. Consequently, this means that changes in cyclone frequency could have either negative or positive effects on air quality. For Melbourne, the projected trends in cyclone frequency anticipate an increase in anticyclones. This suggests that air quality in the future could degrade due to changes in large-scale events. The primary implication of the synoptic-scale study is how important climate driven changes at the synoptic-scale could be for air quality – particularly increases in anticyclone frequency. Thus, air quality management needs to consider shifts in synoptic-scale events in future planning.

In our second study, we generated meteorological-pollutant response curves that identified the nature of local relationships and measured the magnitude of pollutant response to changes in individual weather elements. These element-specific results can inform air quality management on the sensitivity of air pollutants (PM_{10}, O_3, and NO_2) to observed changes in each weather element. This is a powerful tool in understanding the potential effect of climate change as this analysis presents a statistically derived observational window. For instance, increasing temperature was found to have a strong positive effect on all pollutants considered. Therefore, it is probable that if climate change results in increases in temperature then increases in air pollutant concentrations will also occur. Of course, this is assuming all else remains equal. If a business-as-usual climate scenario with increasing emission trends occurs, then air pollution levels will likely increase even more. However, if emissions reductions occur, climate change will likely offset these benefits to some degree making even stronger emission controls necessary for attainment. This is an important implication for future air quality.

In our final study, we shifted our focus away from meteorological-pollutant relationships onto the paramount issue of air quality management – human health. At present, the population of Melbourne is exposed to relatively low to moderate air pollution levels; however, numerous studies – including ours - have associated increases in pollution with adverse health outcomes. As noted above, climate change has the potential to increase exposure levels of air pollution by increasing the frequency of atmospheric conditions conducive to poorer air quality. Importantly, our research indicates that climate change may also influence the exposure-response relationship as the magnitude of air pollutant health effects were found to vary with temperature. This implies that not only will air quality management need to account for increases in pollution due to climate change, but they will also need to contend with increasing population sensitivity to air pollution exposure under increasing temperatures.

At present, dealing with extreme heat events, especially for the most vulnerable groups, is a priority across the field of Public Health in Australia. However, awareness to the dangers of future air quality is growing and the development of relevant information and increased awareness for air quality managers of the importance of planning with climate in mind is becoming increasingly recognized. It is clear that proper understanding of climatological impacts on air quality is necessary if negative consequences are to be avoided or mitigated through risk assessment and standard setting. Finally, despite the potential shortfalls and difficulties in implementing climate change related policy, the key point of our findings is that air quality is sensitive to temperature and thus policy should be considered accordingly.

6.4 INNOVATION

The body of work presented in this thesis is innovative in several respects regarding its approach to addressing the issue of climate for air quality. To begin, this is the first comprehensive assessment of meteorological influences on air pollutants in Australia. This is important because of the unique environment presented in Australia – extreme weather conditions and relatively low air pollution levels. The subtleties of the relationships identified here can act as a guide for what the future air environment may hold for many parts of the world. Secondly, these relationships were assessed using sophisticated statistical analyses on observed data. This is important because results are derived from direct patterns in the data – not deterministic model outputs. We know that deterministic modeling of air pollution is not perfect; thus the use of statistical analysis provides a complementary alternative that can provide meaningful results as well as providing validation for deterministic studies. Of course, this does not mean that deterministic modeling studies are of lesser value but statistical approaches do allow us to produce results that may not have been expected. Moreover, the tiered approach to conducting the research presents a novel means to comprehensively address the issue of weather in air quality. Simply put, this thesis is unique in that it provides a breadth of pertinent information for air quality management as the information herein covers the response of air pollution to atmospheric processes across 1000s of kilometers as well local elements and the subtle influence of weather on the exposure-response relationship of the local population.

6.5 OVERARCHING LIMITATIONS AND GAPS

The limitations of this thesis as a whole must be acknowledged. First, it is important to note that specific limitations for each research chapter (Chapters 3 to 5) are presented therein and thus will not be repeated here. Moving on, this research suffers because it only focuses on the relationships between meteorology, air pollution, and air pollution-related health and does not assess the importance of pollutant emissions in these relationships. Of course, the main goal of this research was to make inferences regarding the importance of meteorology for air quality; however, in reality pollutant emissions play a fundamental role in determining air pollutant concentrations and thus should be considered in any projections of future air quality. That being said, the approach used in this research is a plausible way to highlight the affect that climate driven changes in meteorology could have on air pollution and air pollution-related health effects. Another overarching limitation of this research is that it only examines the relationships of interest over the short-term. Unfortunately, this only covers one part of the story, as the implications of long-term changes are likely to be just as important to air quality. Furthermore, this work is also limited because it is a single-city analysis and not a meta-analysis (i.e., multi-city). Clearly, a multi-city analysis would improve the robustness of findings.

From an atmospheric science point of view, the first two components of this research are limited because they use statistical models to describe a dynamic, interactive atmospheric processes. While statistical models used in this research are highly flexible and reveal patterns that are directly observed in the data they make the assumption that the process of interest – air pollution – can be explained additively. This is not always the case for a complex process such as air pollution and is rationale for the use of more complex numerical models. Consequently, if the combination of two variables has a synergistic effect on air pollution then our models would likely have underestimated the overall influence of our covariates. Nevertheless, individual pollutant-covariate relationships are representative of the data highlighting that statistical models are an excellent way to elucidate relationships that might not be expected.

From an epidemiological standpoint, this final component of this research is constrained by the limitations of using a population-level study design (i.e., ecological study) and thus cannot be used to model at the individual level. Moreover, by using city-wide measures for exposure the assumption is made that ambient pollutant concentrations represent an individual's actual exposure to pollutants. This clearly does not account for exposures that may occur indoors at home or work. Additionally, spatial variation in ambient pollutant concentrations across Melbourne is removed by using a spatially aggregated measure making the misclassification of exposure even more likely. Despite these limitations, associations detected at the population-level are a strong indication that a risk exists and that more research is needed.

6.6 CALL FOR MORE RESEARCH

The importance of daily weather on air pollution processes and related health effects has been clearly defined for Melbourne, Australia. We now know that meteorology is of the upmost importance in these processes – particularly in relation to temperature and over the short-term. What we do not understand is the role of emissions in these processes or the effect of a long-term shift in weather elements. Thus, we need additional studies that examine the changes in emissions and examinations of how long-term shifts in meteorological elements may influence pollution. This could be achieved using a state-of-the-art approach where a regional air quality model (AQM) is driven by output from global climate models (GCMs)-regional climate models (RCMs) simulations. Ideally, multiple simulations would be performed using a range of greenhouse gas emissions scenarios with an ensemble of GCMs. Then the RCM could be used to provide finer scale projections that would be appropriate for the AQM simulations for the Melbourne region. Projections for ten-year periods surrounding 2030, 2050, and 2100 would likely be beneficial in terms of planning for the region. Of course, simulations of the present day would also be important in order to provide a benchmark for comparison. Results from this work would give a strong indication of how future air quality will likely emerge. Unfortunately, this approach is very costly and time consuming. Even so, our findings indicate that it is warranted.

To provide results that are more robust to a global audience we suggest meta-analyses that expand upon the work presented in Chapters 3 and 4 that would include all major cities in Australia and New Zealand. This would allow us to identify the range of influence that weather exhibits on air sheds that are quite varied topographically and climatologically. Furthermore, identification of the most sensitive regions may be of interest on a national scale. Finally -- in order to better understand the contemporary health outcomes of weather induced changes in air pollution -- morbidity outcomes, respiratory mortality, and cardiovascular mortality all need to be investigated in an expansion of Chapter 5, again using a meta-analysis across all cities in Australia and New Zealand.

6.7 FINAL THOUGHTS

Through this research, contribution has been made to what is known about weather-air pollution relationships in Melbourne across two spatial scales important to climate change – synoptic and local. Findings demonstrated that the levels of air pollution in the region are intrinsically linked to weather – particularly temperature – and thus sensitive to climate change. To further complicate matters, the final stage of research identified that near surface air temperature influenced the magnitude of risk associated with air pollution exposure on mortality. Consequently, this means that climate change may present a greater challenge for air quality management than previously anticipated. Thus, if the occurrence of meteorological conditions (e.g., anticyclones, high temperatures, low wind speeds, and low humidity) that promote poor air quality increase in the future and emissions do not decrease, then human health will most likely suffer.

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APPENDIX

#############CHAPTERS 3 & 4 R CODE#######################

```
#Relationship Between Ozone, Synoptic Meteorology, and Local Meteorology in Melbourne
 for the period of 1999 to 2006
#John Pearce, School of Geography & Environmental Science, Monash University


#Ozone Analysis
#Analysis of 1999-2006
library(mgcv) #Load GAM package
library(nlme) #Load mixed model package

#Load Data for analysis
load("Local GAMs Correct Winds.RData
load("Synoptic GAMs.RData")
ls()#View objects in R environment


#Base Model
ozone.gam.base<-gam(log(O3)~s(DateNum, k=32) + s(DOW, k=4) + s(LAT,LONG, bs="tp",
 k=11) + s(lag1, k=4) , data=subset(ozone.grp, O3 > 4),  method="REML", gamma=1.4)
O3.base<-(cor(ozone.gam.base$fitted.values,ozone.gam.base$y))^2
O3.base


#Full Model
ozone.gam2<-gam(log(O3)~s(DateNum, k=32) + s(DOW, k=4) + s(LAT,LONG, bs="tp",
 k=10) + s(lag1, k=4)  + s(MAXTEMP) + s(VP) + s(U.Comp2) + s(V.Comp2) +
 s(SAT.GLOB.RAD) + s(MSP) + s(BLH_4pm) + s(PRECIP), data=subset(ozone.grp, O3 >
 4),  method="REML", gamma=1.4)
O3.full2<-(cor(ozone.gam2$fitted.values,ozone.gam2$y))^2
O3.full2
O3.full-O3.base

#Surface Parameter Evaluation
#Evaluation of Temperature
ozone.gam.base.temp<-gam(log(O3)~s(DateNum, k=32) + s(DOW, k=4) + s(LAT,LONG,
 bs="tp", k=10) + s(lag1, k=4) + s(MAXTEMP), data=subset(ozone.grp, O3 > 4),
 method="REML", gamma=1.4)
O3.base.temp<-(cor(ozone.gam.base.temp$fitted.values,ozone.gam.base.temp$y))^2
O3.base.temp
O3.base.temp-O3.base #Temperature + Base R2

ozone.gam.temp<-gam(log(O3)~s(DateNum, k=32) + s(DOW, k=4) + s(LAT,LONG, bs="tp",
 k=10) + s(lag1, k=4) + s(VP) + s(V.Comp,U.Comp) + s(SAT.GLOB.RAD) + s(MSP) +
 s(BLH_4pm) + s(PRECIP) , data=subset(ozone.grp, O3 > 4),  method="REML",
 gamma=1.4)
O3.full.temp<-(cor(ozone.gam.temp$fitted.values,ozone.gam.temp$y))^2
O3.full.temp
O3.full-O3.full.temp  #Full model minus Temp R2
rm(ozone.gam.temp)

#Evaluation of Vapour Pressure
ozone.gam.base.vp<-gam(log(O3)~s(DateNum, k=32) + s(DOW, k=4) + s(LAT,LONG,
 bs="tp", k=10) + s(lag1, k=4) + s(VP), data=subset(ozone.grp, O3 > 4),  method="REML",
```

```
 gamma=1.4)
O3.base.vp<-(cor(ozone.gam.base.vp$fitted.values,ozone.gam.base.vp$y))^2
O3.base.vp
O3.base.vp-O3.base #VP plus base R2
rm(ozone.gam.base.vp)

ozone.gam.vp<-gam(log(O3)~s(DateNum, k=32) + s(DOW, k=4) + s(LAT,LONG, bs="tp",
 k=10) + s(lag1, k=4) + s(MAXTEMP) + s(V.Comp,U.Comp) + s(SAT.GLOB.RAD) +
 s(MSP) + s(BLH_4pm) + s(PRECIP) , data=subset(ozone.grp, O3 > 4),  method="REML",
 gamma=1.4)
O3.full.vp<-(cor(ozone.gam.vp$fitted.values,ozone.gam.vp$y))^2
O3.full.vp
O3.full-O3.full.vp #Full model minus VP
rm(ozone.gam.vp)

#Evaluation of UV Wind Components
ozone.gam.base.uv<-gam(log(O3)~s(DateNum, k=32) + s(DOW, k=4) + s(LAT,LONG,
 bs="tp", k=10) + s(lag1, k=4) + s(U.Comp,V.Comp), data=subset(ozone.grp, O3 > 4),
 method="REML", gamma=1.4)
O3.base.uv<-(cor(ozone.gam.base.uv$fitted.values,ozone.gam.base.uv$y))^2
O3.base.uv
O3.base.uv-O3.base
rm(ozone.gam.base.uv)

ozone.gam.uv<-gam(log(O3)~s(DateNum, k=32) + s(DOW, k=4) + s(LAT,LONG, bs="tp",
 k=10) + s(lag1, k=4) + s(MAXTEMP) + s(VP) + s(SAT.GLOB.RAD) + s(MSP) +
 s(BLH_4pm) + s(PRECIP) , data=subset(ozone.grp, O3 > 4),  method="REML",
 gamma=1.4)
O3.full.uv<-(cor(ozone.gam.uv$fitted.values,ozone.gam.uv$y))^2
O3.full.uv
O3.full-O3.full.uv
rm(ozone.gam.uv)

#Evaluation of Radiation
ozone.gam.base.rad<-gam(log(O3)~s(DateNum, k=32) + s(DOW, k=4) + s(LAT,LONG,
 bs="tp", k=10) + s(lag1, k=4) + s(SAT.GLOB.RAD), data=subset(ozone.grp, O3 > 4),
 method="REML", gamma=1.4)
O3.base.rad<-(cor(ozone.gam.base.rad$fitted.values,ozone.gam.base.rad$y))^2
O3.base.rad
O3.base.rad-O3.base
rm(ozone.gam.base.rad)

ozone.gam.rad<-gam(log(O3)~s(DateNum, k=32) + s(DOW, k=4) + s(LAT,LONG, bs="tp",
 k=10) + s(lag1, k=4) + s(MAXTEMP) + s(VP) + s(V.Comp,U.Comp) + s(MSP) +
 s(BLH_4pm) + s(PRECIP) , data=subset(ozone.grp, O3 > 4),  method="REML",
 gamma=1.4)
O3.full.rad<-(cor(ozone.gam.rad$fitted.values,ozone.gam.rad$y))^2
O3.full.rad
O3.full-O3.full.rad
rm(ozone.gam.rad)

#Evaluation of MSP
ozone.gam.base.msp<-gam(log(O3)~s(DateNum, k=32) + s(DOW, k=4) + s(LAT,LONG,
 bs="tp", k=10) + s(lag1, k=4) + s(MSP), data=subset(ozone.grp, O3 > 4),  method="REML",
```

```
 gamma=1.4)
O3.base.msp<-(cor(ozone.gam.base.msp$fitted.values,ozone.gam.base.msp$y))^2
O3.base.msp
O3.base.msp-O3.base
rm(ozone.gam.base.msp)

ozone.gam.msp<-gam(log(O3)~s(DateNum, k=32) + s(DOW, k=4) + s(LAT,LONG, bs="tp",
 k=10) + s(lag1, k=4) + s(MAXTEMP) + s(VP) + s(V.Comp,U.Comp) + s(SAT.GLOB.RAD)
 + s(BLH_4pm) + s(PRECIP) , data=subset(ozone.grp, O3 > 4),  method="REML",
 gamma=1.4)
O3.full.msp<-(cor(ozone.gam.msp$fitted.values,ozone.gam.msp$y))^2
O3.full.msp
O3.full-O3.full.msp
rm(ozone.gam.msp)

#Upper Air Parameter Evaluation
#Evaluation of precip
ozone.gam.base.precip<-gam(log(O3)~s(DateNum, k=32) + s(DOW, k=4) + s(LAT,LONG,
 bs="tp", k=10) + s(lag1, k=4) + s(PRECIP), data=subset(ozone.grp, O3 > 4),
 method="REML", gamma=1.4)
O3.base.precip<-(cor(ozone.gam.base.precip$fitted.values,ozone.gam.base.precip$y))^2
O3.base.precip
O3.base.precip-O3.base
rm(ozone.gam.base.precip)

ozone.gam.precip<-gam(log(O3)~s(DateNum, k=32) + s(DOW, k=4) + s(LAT,LONG,
 bs="tp", k=10) + s(lag1, k=4) + s(MAXTEMP) + s(VP) + s(V.Comp,U.Comp) +
 s(SAT.GLOB.RAD) + s(MSP) + s(BLH_4pm) , data=subset(ozone.grp, O3 > 4),
 method="REML", gamma=1.4)
O3.full.precip<-(cor(ozone.gam.precip$fitted.values,ozone.gam.precip$y))^2
O3.full.precip
O3.full-O3.full.precip
rm(ozone.gam.precip)

#Evaluation of blh
ozone.gam.base.blh<-gam(log(O3)~s(DateNum, k=32) + s(DOW, k=4) + s(LAT,LONG,
 bs="tp", k=10) + s(lag1, k=4) + s(BLH_4pm), data=subset(ozone.grp, O3 > 4),
 method="REML", gamma=1.4)
O3.base.blh<-(cor(ozone.gam.base.blh$fitted.values,ozone.gam.base.blh$y))^2
O3.base.blh
O3.base.blh-O3.base
rm(ozone.gam.base.blh)

ozone.gam.blh<-gam(log(O3)~s(DateNum, k=32) + s(DOW, k=4) + s(LAT,LONG, bs="tp",
 k=10) + s(lag1, k=4) + s(MAXTEMP) + s(VP) + s(V.Comp,U.Comp) + s(SAT.GLOB.RAD)
 + s(MSP) + s(PRECIP), data=subset(ozone.grp, O3 > 4),  method="REML", gamma=1.4)
O3.full.blh<-(cor(ozone.gam.blh$fitted.values,ozone.gam.blh$y))^2
O3.full.blh
O3.full-O3.full.blh
rm(ozone.gam.blh)

#Evaluation of Synoptic Circulations
ozone.gam.base.syn<-gam(log(O3)~s(DateNum, k=32) + s(DOW, k=4) + s(LAT,LONG,
 bs="tp", k=10) + s(lag1, k=4)  + ERAI_20_10am, data=subset(ozone.grp, O3 > 4),
```

```
 method="REML", gamma=1.4)
O3.base.syn<-(cor(ozone.gam.base.syn$fitted.values,ozone.gam.base.syn$y))^2
O3.base.syn
rm(ozone.gam.base.syn)

ozone.gam.syn<-gam(log(O3)~s(DateNum, k=32) + s(DOW, k=4) + s(LAT,LONG, bs="tp",
 k=10) + s(lag1, k=4)  + s(MAXTEMP) + s(VP) + s(V.Comp,U.Comp) + s(SAT.GLOB.RAD)
 + s(MSP) + s(BLH_4pm) + s(PRECIP) + ERAI_20_10am, data=subset(ozone.grp, O3 > 4),
 method="REML", gamma=1.4)
O3.full.syn<-(cor(ozone.gam.syn$fitted.values,ozone.gam.syn$y))^2
O3.full.syn
O3.full.syn-O3.full
rm(ozone.gam.syn)


#Generate Table
variable.O3<-c("Base Model", "Full Model", "Temp", "VP",
          "UV", "Rad", "MSP", "precip", "BLH", "SYN")
var.base.O3<-c(O3.base, 0,O3.base.temp-O3.base, O3.base.vp-O3.base,
          O3.base.uv-O3.base, O3.base.rad-O3.base,O3.base.msp-O3.base,
          O3.base.precip-O3.base,O3.base.blh-O3.base,
          O3.base.syn-O3.base)

var.full.O3<-c(0, O3.full,O3.full-O3.full.temp,O3.full-O3.full.vp,
          O3.full-O3.full.uv,O3.full-O3.full.rad,O3.full-O3.full.msp,
          O3.full-O3.full.precip,O3.full-O3.full.blh,
          O3.full.syn-O3.full)

ozone.tab<-data.frame(variable.O3,var.base.O3,var.full.O3)
print(ozone.tab)



##########################################################################
 ####
#Devlop Marginal Effects Plots
pdf("Figure2.pdf", family="serif", pointsize=6, title="Partial Response of O3", width=5,
 height=5)
par(mfrow=c(4,2), mai=c(.35,.35,.05,.05), cex.axis=1.5, cex.lab=1.5)
pcplot(ozone.gam2, select=5,  xlab="Daily Maximum Temperature (C)", ylab="Marginal
 Effect (%)", scale=0)
pcplot(ozone.gam2, select=6,  xlab="Water Vapour Pressure (hPa)", ylab="Marginal Effect
 (%)",scale=0)
pcplot(ozone.gam2, select=7,  xlab="U (km/hr, E+)", ylab="Marginal Effect (%)",scale=0)
pcplot(ozone.gam2, select=8,  xlab="V (km/hr, N+)", ylab="Marginal Effect (%)",scale=0)
pcplot(ozone.gam2, select=9,  xlab="Radiation (MJ/m^2)", ylab="Marginal Effect
 (%)",scale=0)
pcplot(ozone.gam2, select=10,  xlab="Mean Sea-Level Pressure (hPa)", ylab="Marginal
 Effect (%)",scale=0)
pcplot(ozone.gam2, select=11,  xlab="4 p.m. Boundary Layer Height (m)", ylab="Marginal
 Effect (%)",scale=0)
pcplot(ozone.gam2, select=12,  xlab="Precipitation(mm)", ylab="Marginal Effect
 (%)",scale=0)
dev.off()
```

```
#Winds Plot
expvis.gam(ozone.gam, view=c("U.Comp2", "V.Comp2"),theta=35, phi=15,
 ticktype="detailed", color="bw",main = NULL,cex.axis=0.75, family="serif")

#Develop Synoptic Plots
pctermplot(ozone.gam.base.syn, terms=c("ERAI_10am"), se=T,
col.se="black", col.term="black", lwd.term=2, rug=F,
  xlab="Synoptic Circulation Pattern", cex.axis=1.25, cex.lab=1.5,ylab="Marginal Effect (%)
 on O3", family="serif")

  summary(ozone.gam.base.syn)

#Analyze Residuals
par(family="serif")
boxplot(residuals(ozone.gam) ~ ozone.grp2$ID,
ylab = "Raw Residuals", xlab = "EPA Monitor Location",
notch=T, varwidth = T, family="serif", cex.axis=0.75, pch=18, col="lightgrey")
abline(h=0, col="darkgrey")

coplot(log(ozone.gam$residuals)~ozone.grp2$DateNum | ozone.grp2$ID, pch=".",
 panel=panel.smooth, family="serif")

axis(3, labels=format(ref.pollutant, dig=2), cex.axis=0.8,
at=rank(ref.pollutant))
abline(h=0, col="darkgreen")
```

###############CHAPTER 5 R CODE########################

```
#Relationship Between Air Pollution, Temperature, and Mortality in Melbourne for the period
 of 1999 to 2006
#John Pearce, School of Geography & Environmental Science, Monash University

dir()
setwd("E:/Health Study/R Code TempAP")#Only set if directory does not show
 'melbourne.data.27SEP2010.csv'
library(mgcv) #Load mgcv package for generalized additive models
library(season) #Load season package for case-crossover analysis



#Import data set for Melbourne
melbourne<-read.csv("melbourne.data.5JAN2011.csv", header=T, sep=",")
melbourne$DOW<-as.factor(melbourne$DOW)
melbourne$date<-as.Date(melbourne$DATE, "%d/%m/%Y")
melbourne$month<-format.Date(melbourne$date, "%m")
head(melbourne)
tail(melbourne)
melbourne<-melbourne[1:2922,]
str(melbourne)

#Generate Time Series Plot
pdf("timeseries.plots.pdf", family="serif")
par(mfrow=c(5,1), mar=c(2,4,.5,1))
```

```
plot(melbourne$date, melbourne$DEATHS, type='l', col='darkgrey', xlab="Date",
 ylab="Mortality > 64 yrs")
plot(melbourne$date, melbourne$MXT, type='l', col='darkgrey', xlab="Date", ylab="Max
 Temperature °C")
plot(melbourne$date, melbourne$PM10, type='l', col='darkgrey', xlab="Date",
 ylab=expression(PM[10]))
plot(melbourne$date, melbourne$O3, type='l', col='darkgrey', xlab="Date",
 ylab=expression(O[3]))
plot(melbourne$date, melbourne$NO2, type='l', col='darkgrey', xlab="Date",
 ylab=expression(NO[2]))
dev.off()


#Classify Season
melbourne$SEASON[as.numeric(melbourne$month) %in% c(1:3) ]= 'WARM'
melbourne$SEASON[as.numeric(melbourne$month) %in% c(10:12) ]= 'WARM'
melbourne$SEASON[as.numeric(melbourne$month) %in% c(4:9) ]= 'COOL'
melbourne$SEASON<-as.factor(melbourne$SEASON)
str(melbourne)


#################################################################################
 ###############################################################################
 ###
#First, examine main effects
#Run simple additive model for each pollutant
  for (i in (16:18)) {
      fit=gam(DEATHS~DOW + FLU + s(DATENUM, k=8*7) + s(dpt01, k=3) + s(mslp01) +
 s(mxt01, k=6) + melbourne[,i], family=quasipoisson(), data=melbourne, na.rm=T)
          summ<-summary(fit)
      summ.beta=summ$p.coeff[9]
      summ.se=summ$se[9]
          summ.pvalue=summ$p.pv[9]
          per.risk<-100*(exp(summ.beta*10)-1)  #convert relative risk to percent risk
      per.ll<- 100*(exp(10*(summ.beta - 1.96*summ.se))-1)
      per.ul<- 100*(exp(10*(summ.beta + 1.96*summ.se))-1)
     tab<-as.data.frame(c(per.risk,per.ll,per.ul, summ.pvalue))
     rownames(tab)<-c("RISK","LL.95","UL.95","p-value")
     colnames(tab)<-i
     tab<-t(tab)
     nam <- paste("add.tab",names(melbourne)[i], sep=".")
      assign(nam, tab)
      nam2 <- paste("add.fit",names(melbourne)[i], sep=".")
      assign(nam2,fit)
        }

add.results<-rbind(add.tab.pm01, add.tab.o301, add.tab.no201)
rownames(add.results)=c("PM10","O3", "NO2")
print(add.results)

par(mfrow=c(1,3), family='serif')
plot(add.fit.pm01, select=4, xlab="Max Temp (lag 0-1)", ylab="Relative Risk (%)",
 rug=TRUE,
trans=function(x) exp(x))
```

```
abline(h=1, col='grey')

#Now use case-crossover analysis

   fit1<-casecross(DEATHS~FLU + mxt01 + dpt01 + mslp01 + pm01, matchdow=TRUE,
 data=melbourne)
   summ1<-summary(fit1, digits=4)
   tab1<-as.data.frame(summ1[5,])

   fit2<-casecross(DEATHS~FLU + mxt01 + dpt01 + mslp01 + o301, matchdow=TRUE,
 data=melbourne)
   summ2<-summary(fit2, digits=4)
   tab2<-as.data.frame(summ2[5,])

   fit3<-casecross(DEATHS~FLU + mxt01 + dpt01 + mslp01 + no201, matchdow=TRUE,
 data=melbourne)
   summ3<-summary(fit3, digits=4)
   tab3<-as.data.frame(summ3[5,])

   cc.tab<-cbind(tab1,tab2,tab3, deparse.level=0)
   cc.beta<-cc.tab[1,]
   cc.se<-cc.tab[3,]
   cc.p<-cc.tab[5,]
   rownames(cc.p)=c('coef')


   cc.risk<-100*(exp(10*cc.beta)-1)
   cc.ll<-100*(exp(10*(cc.beta-1.96*cc.se))-1)
   cc.ul<-100*(exp(10*(cc.beta+1.96*cc.se))-1)
   cc.tab<-as.data.frame(rbind(cc.risk,cc.ll,cc.ul,cc.p))
   rownames(cc.tab)=c('risk','low','high','p-value')
   colnames(cc.tab)=c('PM10',"O3","NO2")
   print(t(cc.tab))

################################################################################
 ##############################################################################
################################################################################
 ###############################################################################
 ###
#Second, examine seasonal effects
#Run simple stratified additive model for each pollutant
 for (i in (16:18)) {
     fit=gam(DEATHS~DOW + FLU + s(DATENUM, k=8*7) + s(dpt01, k=3) + s(mslp01) +
 s(mxt01, k=6) + SEASON:melbourne[,i], family=quasipoisson(), data=melbourne, na.rm=T)
          summ<-summary(fit)
      summ.beta=summ$p.coeff[9:10]
      summ.se=summ$se[9:10]
          summ.pvalue=summ$p.pv[9:10]
          per.risk<-100*(exp(summ.beta*10)-1)  #convert relative risk to percent risk
      per.ll<- 100*(exp(10*(summ.beta - 1.96*summ.se))-1)
      per.ul<- 100*(exp(10*(summ.beta + 1.96*summ.se))-1)
     tab<-as.data.frame(c(per.risk,per.ll,per.ul, summ.pvalue))
     rownames(tab)<-c("C.RISK","W.RISK","C.LL","W.LL","C.UL","W.UL",
 "C.pval","W.pval")
```

```
      colnames(tab)<-i
      tab<-t(tab)
      nam <- paste("seas.tab",names(melbourne)[i], sep=".")
       assign(nam, tab)
       nam2 <- paste("seas.fit",names(melbourne)[i], sep=".")
       assign(nam2,fit)
         }

season.results<-rbind(seas.tab.pm01, seas.tab.o301, seas.tab.no201)
rownames(season.results)=c("PM10","O3", "NO2")
print(t(season.results))

#Now use case-crossover analysis

   sfit1<-casecross(DEATHS~FLU + mxt01 + dpt01 + mslp01 + SEASON:pm01,
 matchdow=TRUE, data=melbourne)
   summ1<-summary(sfit1, digits=4)
   tab1<-as.data.frame(summ1[5:6,])

   sfit2<-casecross(DEATHS~FLU + mxt01 + dpt01 + mslp01 + SEASON:o301,
 matchdow=TRUE, data=melbourne)
   summ2<-summary(sfit2, digits=4)
   tab2<-as.data.frame(summ2[5:6,])

   sfit3<-casecross(DEATHS~FLU + mxt01 + dpt01 + mslp01 + SEASON:no201,
 matchdow=TRUE, data=melbourne)
   summ3<-summary(sfit3, digits=4)
   tab3<-as.data.frame(summ3[5:6,])

   cc.tab<-rbind(tab1,tab2,tab3, deparse.level=0)
   cc.beta<-cc.tab[,1]
   cc.se<-cc.tab[,3]
   cc.p<-cc.tab[,5]
   rownames(cc.p)=c('coef')


   cc.risk<-100*(exp(10*cc.beta)-1)
   cc.ll<-100*(exp(10*(cc.beta-1.96*cc.se))-1)
   cc.ul<-100*(exp(10*(cc.beta+1.96*cc.se))-1)
   cc.tab<-as.data.frame(cbind(cc.risk,cc.ll,cc.ul,cc.p))
   rownames(cc.tab)=c('PM10.C','PM10W',"O3.C","O3.W","NO2.C","NO2.W")
   colnames(cc.tab)=c("RISK", "LOW", "HIGH", 'p-value')
   print(cc.tab)


##############################################################################
 ##############################################################################
 #
#Second, examine joint effects using a tensor product smooth
     ylabs<-c("PM10(lag 0-1)", "O3(lag 0-1)", "NO2(lag 0-1)") #set up y-axis labels
     xlabs<-c("Max Temp (lag 0-1)","Max Temp (lag 0-1)","Max Temp (lag 0-1)")


   pdf("jointeffectsplot6.pdf",family='serif', pointsize=16) #Set up plotting window
```

```
   for (i in (16:18)) {
     fit=gam(DEATHS~DOW + FLU + s(DATENUM, k=8*7) + s(dpt01, k=3) + s(mslp01) +
 te(mxt01, melbourne[,i], k=c(6,3)), family=quasipoisson(), data=melbourne, na.rm=T)
      nam2 <- paste("ten.fit",names(melbourne)[i], sep=".")
      assign(nam2,fit)
      #3D Plot
      plot(fit, select=4, rug=T, ticktype="detailed", pers=TRUE, theta=-35, phi=15,
 expand=0.75, col="grey",main="Deaths > 64 yrs", ylab=ylabs[i-15],
                  xlab=xlabs[i-15],trans=function(x)exp(x), shift=mean(predict(fit)),
 zlim=c(35,65), cex.lab=1.0)
      #Contour Plot
      plot(fit, select=4, ticktype="detailed", pers=FALSE, ylab=ylabs[i-15], se=FALSE,
 main="",
                  xlab=xlabs[i-15],trans=function(x)exp(x), shift=mean(predict(fit)),
 cex.lab=1.0,col="black", labcex=1)
      }
      dev.off()

#Compare additive model versus interaction model
anova(add.fit.pm01,ten.fit.pm01, test="Chisq")
anova(add.fit.o301,ten.fit.o301, test="Chisq") #No p-value provided?
anova(add.fit.no201,ten.fit.no201, test="Chisq")



###############################################################################
 ########################################################################
#Threshold Analysis using case-crossover analysis
summary(melbourne$mxt01)
melbourne$date<-as.Date(melbourne$DATE, "%d/%m/%Y")
threshold=round(seq(from=20, to=40, by=1)) #set up temperature range to examine

pm.case.thresh=sapply(threshold,function(temp){
   melbourne$THRESH<-cut(melbourne$mxt01, breaks=c(9,16,temp,42))
   melbourne$THRESH<-as.factor(melbourne$THRESH)
   size<-as.numeric(table(melbourne$THRESH)[3])
   mxt<-as.numeric(temp)
   fit1<-casecross(DEATHS~ dpt01 + mslp01 + THRESH:pm01, matchdow=T,
 data=melbourne)
   summ1<-summary(fit1, digits=4)
   summ.beta.pm<-as.numeric(summ1[4,1])
   summ.se.pm<-as.numeric(summ1[4,3])
   cc.risk<-100*(exp(summ.beta.pm*10)-1)  #convert odds ratio to percent risk
   cc.ll<- 100*(exp(10*(summ.beta.pm - 1.96*summ.se.pm))-1)
   cc.ul<- 100*(exp(10*(summ.beta.pm + 1.96*summ.se.pm))-1)
   cc.p<-as.numeric(summ1[4,5])
   mod.out<-fit1$c.model
   aic.mod<- -2*mod.out$loglik[2] + 2*length(mod.out[1])
   cc.tab<-as.data.frame(cbind(cc.risk,cc.ll, cc.ul, cc.p, aic.mod,size, mxt))
   })
pm.case.thresh

plot(threshold,as.numeric(pm.case.thresh[5,]), type='b', ylab="AIC", xlab="Max Temperature
 (°C)", pch=20)
```

```
##################################################
#Ozone

o3.case.thresh=sapply(threshold,function(temp){
   melbourne$THRESH<-cut(melbourne$mxt01, breaks=c(9,16,temp,42))
   melbourne$THRESH<-as.factor(melbourne$THRESH)
   size<-as.numeric(table(melbourne$THRESH)[3])
   mxt<-as.numeric(temp)
   fit1<-casecross(DEATHS~ dpt01 + mslp01 + THRESH:o301, matchdow=T,
 data=melbourne)
   summ1<-summary(fit1, digits=4)
   summ.beta.o3<-as.numeric(summ1[4,1])
   summ.se.o3<-as.numeric(summ1[4,3])
   cc.risk.o3<-100*(exp(summ.beta.o3*10)-1)  #convert odds ratio to percent risk
   cc.ll.o3<- 100*(exp(10*(summ.beta.o3 - 1.96*summ.se.o3))-1)
   cc.ul.o3<- 100*(exp(10*(summ.beta.o3 + 1.96*summ.se.o3))-1)
   cc.p<-as.numeric(summ1[4,5])
   mod.out<-fit1$c.model
   aic.mod<- -2*mod.out$loglik[2] + 2*length(mod.out[1])
   cc.tab<-as.data.frame(cbind(cc.risk.o3,cc.ll.o3, cc.ul.o3, cc.p, aic.mod, size, mxt))
   })
o3.case.thresh
plot(threshold,as.numeric(o3.case.thresh[5,]), type='b', ylab="AIC", xlab="Max Temperature
 (°C)", pch=20)


##################################################
#NO2
no2.case.thresh=sapply(threshold,function(temp){
   melbourne$THRESH<-cut(melbourne$mxt01, breaks=c(9,16,temp,42))
   melbourne$THRESH<-as.factor(melbourne$THRESH)
   size<-as.numeric(table(melbourne$THRESH)[3])
   mxt<-as.numeric(temp)
   fit1<-casecross(DEATHS~ dpt01 + mslp01 + THRESH:no201, matchdow=T,
 data=melbourne)
   summ1<-summary(fit1, digits=4)
   summ.beta.no2<-as.numeric(summ1[4,1])
   summ.se.no2<-as.numeric(summ1[4,3])
   cc.risk.no2<-100*(exp(summ.beta.no2*10)-1)  #convert odds ratio to percent risk
   cc.ll.no2<- 100*(exp(10*(summ.beta.no2 - 1.96*summ.se.no2))-1)
   cc.ul.no2<- 100*(exp(10*(summ.beta.no2 + 1.96*summ.se.no2))-1)
   cc.p<-as.numeric(summ1[4,5])
   mod.out<-fit1$c.model
   aic.mod<- -2*mod.out$loglik[2] + 2*length(mod.out[1])
   cc.tab<-as.data.frame(cbind(cc.risk.no2,cc.ll.no2, cc.ul.no2, cc.p, aic.mod, size, mxt))
   })
no2.case.thresh
plot(threshold,as.numeric(no2.case.thresh[5,]), type='b', ylab="AIC", xlab="Average
 Temperature (°C)", pch=20)

#Evaluate Threshold by plotting change in risk as temperature cut point changes

 pdf("cc.threshold.plots.mxt01.pdf", width=5, height=7)
```

```
par(mfrow=c(3,1), family='serif', mar=c(4,4.5,.5,.5)) #Set up plotting window

plot(threshold,as.numeric(pm.case.thresh[1,]), type='b', ylab="Risk per 10 µg (±95%CI)",
xlab="Max Temperature (lag 0-1)", pch=20, ylim=c(-2,5)) #examine threshold based on or
lines(threshold,as.numeric(pm.case.thresh[2,]),lty=2)
lines(threshold,as.numeric(pm.case.thresh[3,]),lty=2)
abline(h=0,lty=1, col='lightgrey')
 text(20,-1.5,expression(PM[10]))

 plot(threshold,as.numeric(o3.case.thresh[1,]), type='b', ylab="Risk per 10 ppb (±95%CI)",
xlab="Max Temperature (lag 0-1)", pch=20, ylim=c(-2,5)) #examine threshold based on or
lines(threshold,as.numeric(o3.case.thresh[2,]),lty=2)
lines(threshold,as.numeric(o3.case.thresh[3,]),lty=2)
abline(h=0,lty=1, col='lightgrey')
 text(20,-1.5,expression(O[3]))

 plot(threshold,as.numeric(no2.case.thresh[1,]), type='b', ylab="Risk per 10 ppb (±95%CI)",
xlab="Max Temperature (lag 0-1)", pch=20, ylim=c(-2,5)) #examine threshold based on or
lines(threshold,as.numeric(no2.case.thresh[2,]),lty=2)
lines(threshold,as.numeric(no2.case.thresh[3,]),lty=2)
abline(h=0,lty=1, col='lightgrey')
 text(20,-1.5,expression(NO[2]))


 dev.off()

pdf("cc.AIC.plots.mxt01.pdf", width=5, height=7)
par(mfrow=c(3,1), family='serif', mar=c(4,4.5,2,.5)) #Set up plotting window
plot(threshold,as.numeric(pm.case.thresh[5,]), type='b', ylab="AIC", xlab="Max Temperature
 (lag 0-1)", pch=20, ylim=c(1449560,1449590) )
abline(v=28, col='darkgrey', lty=2)
text(28.75,1449585,"28 °C")
text(20,1449562,expression(PM[10]))
plot(threshold,as.numeric(o3.case.thresh[5,]), type='b', ylab="AIC", xlab="Max Temperature
 (lag 0-1)", pch=20, ylim=c(1449560,1449590))
abline(v=22, col='darkgrey', lty=2)
text(22.75,1449585,"22 °C")
text(20,1449562,expression(O[3]))
plot(threshold,as.numeric(no2.case.thresh[5,]), type='b', ylab="AIC", xlab="Max Temperature
 (lag 0-1)", pch=20, ylim=c(1449560,1449590))
abline(v=28, col='darkgrey', lty=2)
text(28.75,1449585,"28 °C")
text(20,1449562,expression(NO[2]))
dev.off()

#Break Average temps into a factor based on threshold analysis
quantile(round(melbourne$at01), probs = c(1,50,99)/100, na.rm=T)
melbourne$TCAT01[round(melbourne$at01) <=7]= '1'
melbourne$TCAT01[round(melbourne$at01) %in% c(10:25)]= '2'
melbourne$TCAT01[round(melbourne$at01) >=26]= '3'
melbourne$TCAT01<-as.factor(melbourne$TCAT01)
table(melbourne$TCAT01)

##############################################################################
```

```
############################################################################
#Investigate the stratified effects of each air pollutant using case-crossover analysis


summary(melbourne)

   fit1<-casecross(DEATHS~ dpt01 + mslp01 + TCAT01:pm01, matchdow=T,
 data=melbourne)
   summ1<-summary(fit1, digits=4)
   tab1<-as.data.frame(summ1[3:5,])


   fit2<-casecross(DEATHS~ dpt01 + mslp01 + TCAT01:o301, matchdow=T,
 data=melbourne)
   summ2<-summary(fit2, digits=4)
   tab2<-as.data.frame(summ2[3:5,])

   fit3<-casecross(DEATHS~ dpt01 + mslp01 + TCAT01:no201, matchdow=T,
 data=melbourne)
   summ3<-summary(fit3, digits=4)
   tab3<-as.data.frame(summ3[3:5,])

   cc.tab.strat<-rbind(tab1,tab2,tab3, deparse.level=0)
   cc.beta.strat<-cc.tab.strat[,c(1)]
   cc.se.strat<-cc.tab.strat[,c(3)]

   cc.or.strat<-100*(exp(10*cc.beta.strat)-1)   #convert odds ratio to percent risk
   cc.ll.strat<-100*(exp(10*(cc.beta.strat-1.96*cc.se.strat))-1)
   cc.ul.strat<-100*(exp(10*(cc.beta.strat+1.96*cc.se.strat))-1)
   cc.tab.strat<-as.data.frame(cbind(cc.or.strat,cc.ll.strat,cc.ul.strat))

 rownames(cc.tab.strat)=c("COLD.pm","MILD.pm","HOT.pm","COLD.o3","MILD.o3","HO
 T.o3", "COLD.no2","MILD.no2","HOT.no2")
   print(cc.tab.strat)

#Generate Plot

 #Rearrange data for plotting
 cc.or.plot<-cc.or.strat[c(1,4,7,2,5,8,3,6,9)]
 cc.ll.plot<-cc.ll.strat[c(1,4,7,2,5,8,3,6,9)]
 cc.ul.plot<-cc.ul.strat[c(1,4,7,2,5,8,3,6,9)]

 pdf("E:/Health Study/R Code TempAP/cc.strat.plot4.pdf", width=10, height=7)
 par(las=1, mar=c(5,5,2,2), family='serif')
 xaxis=c("","COLD (5:7 °C)","","","MILD (8:25 °C)","","","HOT (26:31 °C)","")
 plot(1:9, cc.or.plot,ylab="Percent Risk (±95%CI)", axes=FALSE, xlab="Temperature
 Stratum (lag 0-1)", ylim=c(-2,6), xlim=c(1,9.5), pch=c(21,22,24), col="black",bg="black",
 cex=1.25, cex.lab=1.5)
 axis(1, at=1:9, labels=xaxis)
 axis(2)
  arrows(1:9, cc.ll.plot, 1:9, cc.ul.plot, lwd=c(1,1,1,1,2,2,2,2,2),

 col=c("darkgrey","darkgrey","darkgrey","darkgrey","black","black","black","black","black")
 , angle=90, code=3, length=0.1)
```

```
points(1:9, cc.or.plot, pch=c(21,22,24), col="black",bg="black", cex=1.25)
abline(h=0, col="black")
abline(v=c(3.75,6.75),lty=2)
legend(8,-1, c(expression(PM[10]),expression(O[3]),expression(NO[2])),
col="black",pt.bg="black", cex=1.25, text.col='black', pch=c(21,22,24), bty='n')
box()

dev.off()
```