CHAPTER 5

Air Quality

HEALTH OF CANADIANS IN A CHANGING CLIMATE: ADVANCING OUR KNOWLEDGE FOR ACTION

Health Canada Santé Canada

Canada

Lead Authors

Marika Egyed, Health Canada

Phil Blagden, Health Canada

Contributing Authors

David Plummer, Environment and Climate Change Canada

Paul Makar, Environment and Climate Change Canada

Carlyn Matz, Health Canada

Mike Flannigan, University of Alberta

Morgan MacNeill, Health Canada

Eric Lavigne, Health Canada

Benny Ling, Health Canada

Diana Valencia Lopez, Health Canada

Betty Edwards, Health Canada

Radenko Pavlovic, Environment and Climate Change Canada

Jacinthe Racine, Environment and Climate Change Canada

Pierre Raymond, Health Canada

Robyn Rittmaster, Health Canada

Aaron Wilson, Health Canada

Guoliang Xi, Health Canada

Suggested Citation

Egyed, M., Blagden, P., Plummer, D., Makar, P., Matz, C., Flannigan, M., MacNeill, M., Lavigne, E., Ling, B., Lopez, D. V., Edwards, B., Pavlovic, R., Racine, J., Raymond, P., Rittmaster, R., Wilson, A., & Xi, G. (2022). Air Quality. In P. Berry & R. Schnitter (Eds.), *Health of Canadians in a Changing Climate: Advancing our Knowledge for Action*. Ottawa, ON: Government of Canada.

Table of Contents

Summary

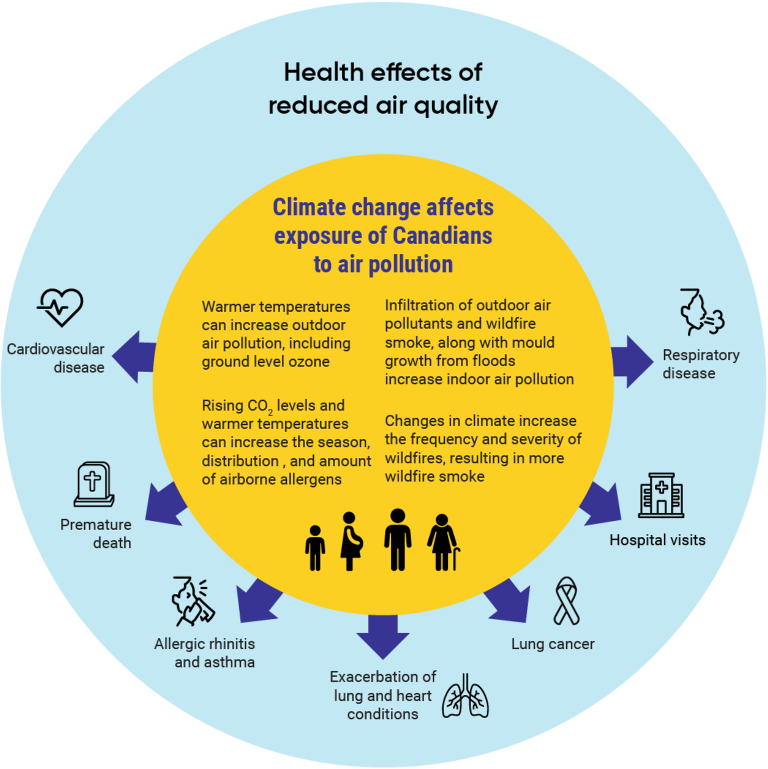
Climate change and air quality are intimately linked: changes in climate are affecting air quality in Canada, and several air pollutants contribute to climate change. Exposure to key air pollutants, including fine particulate matter and ozone, increases the risk of adverse health outcomes, ranging from respiratory symptoms to development of disease and premature death. A warming climate is expected to worsen air pollution levels in Canada. As the frequency and severity of wildfires are expected to increase due to climate change, emissions from wildfires represent one of the most significant climate-related risks to air quality in Canada. Climate change can also affect indoor air quality when elevated levels of outdoor air pollutants infiltrate buildings or when mould accumulates following extreme weather events, such as floods. Changes in the climate are affecting airborne allergens such as pollen by expanding the geographic distribution of plant species, extending pollen seasons, and increasing pollen counts.

Some groups are at increased risk of health impacts related to air pollution, including children, seniors, Indigenous Peoples, those with pre-existing conditions such as asthma or cardiac disease, and populations living in high air pollution areas. Mitigation efforts to reduce emissions of greenhouse gases can have substantial population health co-benefits related to improved air quality. These co-benefits can help offset the costs of climate mitigation, providing support for accelerated implementation of mitigation policies. Adaptation efforts that would prevent or alleviate the climate-related health impacts of air pollution include limiting exposure to air pollutants, including through the use of emergency shelters during wildfires; providing daily forecasts of air quality, wildfire smoke, and aeroallergens, such as the Air Quality Health Index; implementing flood prevention and ensuring that buildings have adequate ventilation and air filtration.

Key Messages

- Climate change and air quality are linked: overall, a warmer climate is expected to worsen air pollution in Canada, and some air pollutants contribute to climate change. If air pollution emissions remain unchanged, a warming climate will likely increase ozone levels in heavily populated and industrialized areas, including Southern Ontario and Southwestern Quebec. Projected effects on fine particulate matter are more modest and uncertain.
- The health impacts of air pollution in Canada, including premature death and disease, are expected to worsen in the future due to the influence of climate change. Unless these impacts are offset by reducing air pollution, hundreds of deaths annually are expected to result by mid-century. Today, air pollution is a leading environmental cause of death and illness in Canada, resulting in an estimated 15,300 deaths a year, with an economic value of $114 billion annually.

- Canada needs to prepare for a future with more wildfires. Increasing wildfire emissions are one of the most significant climate-related risks to air quality in Canada. Wildfire smoke, which can spread over vast areas of the country, contributed to an estimated 620 to 2700 deaths annually in Canada from 2013 to 2018. The public health burden of wildfire smoke is expected to increase in the future due to climate change.
- Climate change will increase the length of airborne allergen seasons, pollen counts, and the geographical distribution of allergens. Respiratory allergies and asthma are expected to affect more people more often in the future, increasing costs to the health system.
- Climate change can affect indoor air quality through increased infiltration of outdoor pollutants and allergens, and as a result of weather events such as floods that cause mould growth in buildings. At the same time, energy retrofits of buildings without adequate ventilation can reduce indoor air quality. Key adaptation strategies for indoor air include ventilation, filtration, and controlling pollutant sources.
- Daily forecasts of air quality, wildfire smoke, and airborne allergens presented in accessible formats, such as the Air Quality Health Index, are important tools to protect community health and a key adaptation strategy to inform populations at higher risk of health impacts resulting from a changing climate.
- Mitigation measures targeting climate pollutants, including methane and black carbon (soot), can have important immediate and long-term co-benefits for the health of local populations by reducing air pollution. The air quality benefits of these mitigation measures can help to offset the costs of climate action. Climate mitigation measures that improve air quality will also help avoid thousands of deaths annually in Canada by the middle of the century.



Health risks of climate change impacts on air pollution.

Overview of Climate-Related Health Impacts Associated with Air Quality

HEALTH IMPACT OR HAZARD CATEGORY	CLIMATE-RELATED CAUSES	POSSIBLE HEALTH EFFECTS
Air quality	• Higher levels of airborne particulate matter related to smoke from wildfires • Increased ground-level ozone (and potentially particulate matter) due to warming • Higher levels of airborne particulate matter related to droughts • Increased mould and chemical contaminants in indoor environments due to flooding, effects on ambient air quality and increased releases from indoor sources of air pollution • Extended season and geographical distribution of pollen, and increased production of pollen by plants and trees • Warming and changes to precipitation affecting growth and ranges of air/aerosol or droplet-borne pathogens	• Air quality–related respiratory and cardiovascular disease and premature deaths • Exacerbation of chronic respiratory diseases, such as asthma and chronic obstructive pulmonary disease • Lung cancer • Development and exacerbation of allergies • Eye, nose, and throat irritation, and shortness of breath • Exacerbation of mental health impacts • Impacts on health infrastructures and services • Impacts on health and social services • Risks from infectious diseases acquired by inhalation from environmental sources (e.g., cryptococcosis)

5.1 Introduction

Climate change and air quality are intimately linked in multiple ways that can have repercussions for human health. Changes in climate, such as increasing temperatures, can influence the levels of air pollutants that affect Canadians. Conversely, some air pollutants contribute to climate change and warming of the atmosphere. In addition, a changing climate is expected to create conditions that affect the frequency and severity of wildfires, an important source of air pollution, and the levels of aeroallergens such as pollen, which affect allergic diseases. A changing climate can also affect indoor air quality: elevated levels of outdoor air pollution (e.g., smog, wildfire smoke, and aeroallergens) can infiltrate into buildings; actions that increase energy efficiency of buildings without adequate ventilation can result in accumulation of indoor-generated air pollution; and extreme weather events such as flooding can lead to the presence of mould indoors.

Climate pollutants and air pollutants are often emitted from the same sources, such as those relating to the combustion of fossil fuels. As a result, greenhouse gas (GHG) mitigation strategies targeting reductions in climate pollutants can have important local health co-benefits due to accompanying reductions in air pollutant emissions.

This chapter first summarizes the effects of air pollution on health and describes how climate change and air quality are interrelated. It then describes the expected effects of a warming climate on air pollution health impacts in Canada, as outdoor air pollutant concentrations, aeroallergens, and indoor air quality are influenced by the changing climate. Consideration is given to populations at greater risk from air pollution, such as children, seniors, those with underlying health conditions, and Indigenous Peoples. Future wildfire activity under climate change, as well as the potential health co-benefits associated with mitigating GHG emissions, are highlighted. To help decision makers plan for climate change impacts on the health of Canadians, potential adaptation strategies are discussed.

5.2 Methods and Approach

The information presented in this chapter focuses on impacts in Canada, with provincial or regional information provided, where available. Results from studies done in the United States are also reviewed, as often only limited information specific to Canada is available, and the two countries are generally similar in multiple ways (e.g., economic development, population demographics, and air pollution emission controls). Overall, an emphasis has been placed on studies published from 2005 to 2019. Most of the literature reviewed was drawn from peer-reviewed English language journals and identified through searches of citation databases (i.e., PubMed, Medline, Scopus, and Google Scholar) and supplemented with manual scanning of reference lists in key papers. Both primary studies and review articles were consulted. In addition, relevant documents retrieved from the grey literature, including reports and web content from governments, agencies, and organizations, were reviewed.

This chapter also includes two new analyses conducted for this report. The first is an investigation of Canadian air pollution health impacts associated with climate change and the potential air pollution health co-benefits of GHG mitigation. The second focuses on the health impacts from wildfire smoke during recent years to better understand the implications of a changing climate on this air pollution source.

5.3 Health Effects of Outdoor Air Pollution

Air pollution is a leading environmental cause of death, both globally and in Canada (IHME, 2019), and the 11th largest cause of death overall in Canada (Alam et al., 2019). It is estimated that current levels of three major air pollutants — fine particulate matter ($PM_{2.5}$),[1] ozone, and nitrogen dioxide (NO_2) — together cause about 15,300 premature deaths in Canada annually, with an economic value of $114 billion (Health Canada, 2021). In addition, these air pollutants are estimated to result in many non-fatal health outcomes, including thousands of hospital visits and millions of days with asthma symptoms annually in Canada; thus, they represent an important population health issue. The scientific evidence indicates that there is no exposure threshold below which there is no risk for many of these health effects. Any incremental increase in air pollutant concentration, including a small increase, is associated with an increased risk of adverse health outcomes in the population. Hence, even in Canada, where air pollution levels are relatively low compared to many other countries, air pollution results in a considerable burden of disease.

The health effects of air pollution have been extensively studied and are well documented in the peer-reviewed scientific literature, which has been reviewed in depth by Health Canada (Health Canada, 2013; Health Canada, 2016), the US Environmental Protection Agency (US EPA) (US EPA, 2019; US EPA, 2020b), and by international organizations such as the World Health Organization (WHO, 2020). It is recognized that exposure to key air pollutants, including $PM_{2.5}$ and ozone, results in increased risk of a wide variety of adverse health outcomes in the population, ranging from respiratory symptoms to development of disease and premature death. This section provides an overview of the health effects of ozone and $PM_{2.5}$ to help understand current and possible future risks to the health of Canadians from climate change.

5.3.1 Particulate Matter

Particulate matter (PM) is a complex mixture of very small solid particles and liquid droplets made up of many different chemicals, including elemental carbon, organic compounds, metals, sulphates, and nitrates. These particles are small enough to remain suspended in the atmosphere for an appreciable period of time. PM is emitted directly from sources (primary PM, e.g., black carbon or soot) or formed in the atmosphere from precursor compounds that undergo chemical reactions (secondary PM). Both primary PM and

---

[1] $PM_{2.5}$ is used to denote particles with a mass median diameter of 2.5 microns or less. $PM_{2.5}$ can penetrate deep in the human lung (Health Canada, 2013). Particulate matter is also frequently referred to as “aerosols” or “aerosol particles,” particularly when discussing climate effects.

secondary PM precursors have a large variety of natural and anthropogenic sources, such as wildfires, motor vehicle exhaust, and coal combustion. The health effects of PM are directly associated with the size of the particles, as smaller particles penetrate deeper into the lungs. $PM_{2.5}$ is of significant concern for human health, has been extensively researched and reviewed, and is causally associated with the greatest range of health effects of any of the major (or "criteria") air pollutants.

Risk assessments published by Health Canada (2013) and the US EPA (2019) have evaluated the weight of evidence of health effects from short-term and long-term exposure to $PM_{2.5}$, and their conclusions are summarized here. Evidence indicates that short-term exposure (hours to days) to ambient $PM_{2.5}$ increases the risk of premature death, including all-cause, cardiovascular, and respiratory mortality. In addition, short-term exposure increases the risk of adverse cardiovascular and respiratory effects, including aggravation of pre-existing cardiovascular and respiratory diseases (e.g., asthma or chronic obstructive pulmonary disease [COPD]), changes in cardiac and lung function, increased respiratory symptoms, and deterioration in indicators of cardiovascular health. Short-term exposure also results in increased medical interventions. For example, population-based studies show that increases in short-term $PM_{2.5}$ exposure are associated with increases in emergency room visits and hospital admissions relating to respiratory and cardiovascular conditions.

Overall, the evidence indicates that long-term exposure (months to years) to ambient $PM_{2.5}$ increases the risk of non-accidental and cardiovascular mortality, and it may also be associated with respiratory mortality. As well, long-term $PM_{2.5}$ exposure increases the risk of adverse respiratory effects, including increased respiratory symptoms and effects on lung development in children, as well as cardiovascular effects, such as health parameters relating to atherosclerosis progression. Chronic exposure to $PM_{2.5}$ increases the risk of lung cancer, and the International Agency for Research on Cancer has classified outdoor PM as carcinogenic in humans (IARC, 2016).

In addition to the extensive evidence examining respiratory and cardiovascular outcomes of $PM_{2.5}$, there is emerging evidence that long-term exposure to $PM_{2.5}$ may be associated with adverse effects on the nervous system, including dementia (Fu et al., 2019). Exposure to $PM_{2.5}$ has also been increasingly associated with other health effects, including metabolic diseases such as diabetes (Chen et al., 2013a) and pregnancy outcomes such as low birth weight (Stieb et al., 2016; Lavigne et al., 2019).

5.3.2 Ozone

Ozone is a gas that is not directly emitted by air pollution sources. Rather, it forms in the atmosphere from reactions between precursor compounds and sunlight. The weight of evidence of health effects of short-term and long-term exposure to ozone have been evaluated by Health Canada (2013) and the US EPA (2020a), the conclusions of which are summarized here. Short-term exposure to ozone increases the risk of non-accidental and cardiopulmonary mortality. The evidence from short-term ozone-exposure studies in humans also demonstrates reduced lung function, increased respiratory symptoms (e.g., cough, shortness of breath), and airway hyper-responsiveness. Epidemiological studies provide evidence of exacerbation of asthma and COPD, respiratory infections, and increases in hospitalizations and emergency room visits for respiratory conditions such as asthma. There is also evidence that short-term ozone exposure is associated with metabolic effects and may also be associated with adverse effects on the cardiovascular system. Long-term exposure to ozone has been associated with reduced lung function growth in children (Gauderman et al., 2000), onset of asthma in subgroups that spend more time outdoors (e.g., children, outdoor workers), and increased respiratory symptoms in children with asthma. There is some evidence that long-term ozone exposure may contribute to premature mortality, in particular from respiratory causes (Turner et al., 2016).

5.3.3 Populations at Higher Risk

Evidence indicates that various subpopulations are at increased risk of health effects from poor air quality (Health Canada, 2013; US EPA, 2019; US EPA, 2020a). Increased risk of PM-related health effects has been identified for older adults and young children, as well as for individuals with pre-existing cardiovascular or pulmonary diseases, or genetic factors that make them more sensitive to the effects of PM. Similarly, some individuals have increased sensitivity to ozone exposure due to underlying health conditions, age, how they spend their time, or genetics. Specifically, ozone exposure increases health risks among children, older adults, subgroups that spend increased time outdoors, individuals with asthma or COPD, and individuals with polymorphisms in genes associated with oxidative stress responses and inflammation. Overall, it has been estimated that about one-third of the Canadian population has at least one risk factor that increases their susceptibility to air pollution (Stieb et al., 2019).

Populations living in areas with higher levels of air pollution are also at increased risk for adverse health effects. Studies of populations in large urban centres in Canada suggest that material and social deprivation are associated with increased exposure to air pollution (Pinault et al., 2016a; Pinault et al., 2017). In addition, children living in low-income areas, racialized populations, and immigrants are exposed to higher levels of air pollution than children in high-income areas, White populations, and non-immigrants, respectively (Pinault et al., 2016b; Pinault et al., 2017).

Indigenous Peoples[2] in Canada may also be more sensitive to adverse health effects from poor outdoor air quality. Overall, the burden of chronic respiratory disease, including asthma and COPD, has been reported to be disproportionally higher in First Nations people (Carriere et al., 2017) and in Métis people (Gershon et al., 2014). In addition, higher rates of lower respiratory tract infections have been reported in First Nations and Inuit children (Kovesi, 2012; McCuskee et al., 2014). This can increase health risks from air pollution. In addition, the risk of exposure to poor air quality may be elevated for Indigenous Peoples due to multiple factors, including overcrowded housing, inadequate ventilation, exposure to smoke from wood heating, and the geographic proximity of many Indigenous communities to forests and, therefore, increased risks from wildfire smoke (Reading & Halseth, 2013; NCCAH, 2017). The existing health and social inequities experienced by Indigenous populations can increase health risks related to poor outdoor and indoor air quality. Many of these inequities are underpinned by systemic racism and colonization (see Chapter 2: Climate Change and Indigenous Peoples' Health in Canada and Chapter 9: Climate Change and Health Equity).

---

[2] The term Indigenous is used in this chapter to collectively refer to the original inhabitants of Canada and their descendants, including First Nations, Inuit, and Métis peoples as defined under Section 35 of the *Constitution Act, 1982*. Wherever possible, clear distinctions are made between these three distinct, constitutionally recognized groups.

5.4 Interactions Between Climate Change and Outdoor Air Pollution

Many of the human activities that emit carbon dioxide (CO_2), the main driver of climate change, also emit compounds that contribute to outdoor air pollution. For example, in addition to climate pollutants such as CO_2 and methane, fossil fuel combustion also emits air pollutants, including $PM_{2.5}$, nitrogen oxides (NO_x, including NO_2), sulphur dioxide (SO_2), carbon monoxide (CO), and volatile organic compounds (VOCs). Reactions among these pollutants may form additional $PM_{2.5}$ and ground-level ozone. Some of these air pollutants can modify the radiative balance of the Earth, affecting climate, and are commonly referred to as short-lived climate forcers (SLCFs), as these compounds have a much shorter residence time in the atmosphere than CO_2. This section reviews the linkages among different SLCFs, climate change, and air quality, with a focus on PM, methane, and ozone. An overview of these linkages is presented in Figure 5.1.

The impacts of an SLCF on climate are quantified using estimates of their "radiative forcing," defined as "the net change in the energy balance of the Earth system due to some imposed perturbation" (Myhre et al., 2013, p. 664). An SLCF that produces a positive radiative forcing will result in some increase in the near-surface temperature, while ones producing a negative radiative forcing will have a net cooling effect. However, the efficacy of SLCFs in producing near-surface temperature changes for a given radiative forcing varies. Many forcing agents produce rapid adjustments in various components of the climate system — the effects of aerosols on clouds, for example — and the estimates of radiative forcing discussed here typically include these rapid adjustments, except where stated.

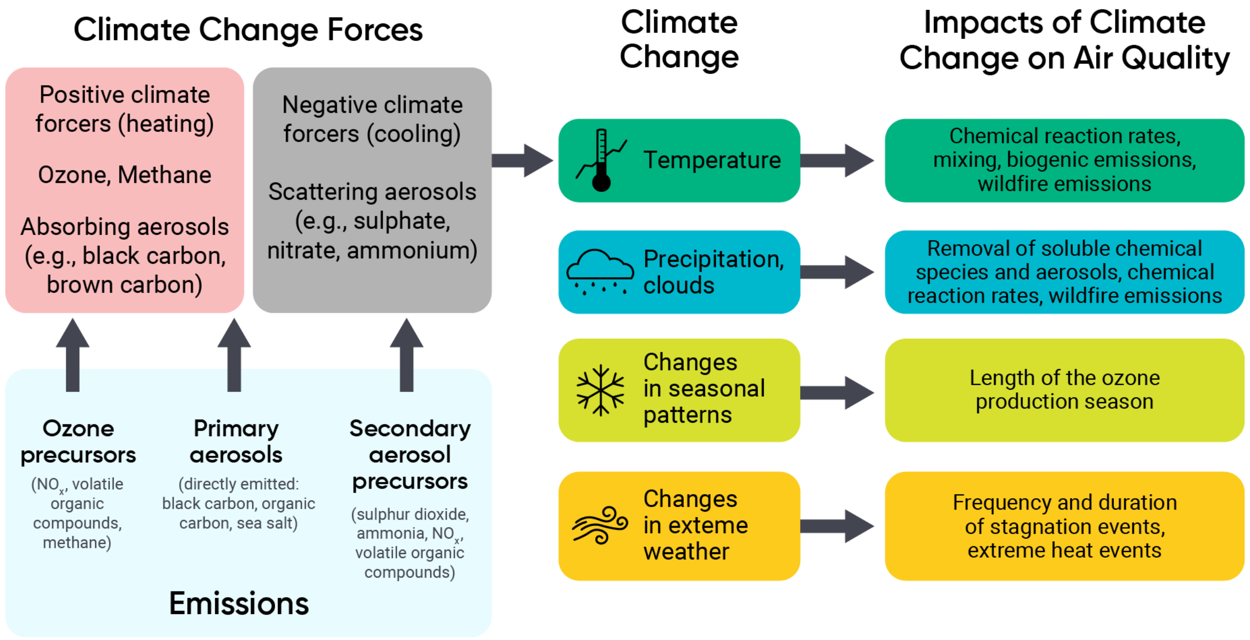


Figure 5.1 Linkages between air quality and climate change. The box colour for the SLCFs indicates either a new positive radiative forcing (red; warming) or a net negative radiative forcing (grey; cooling). Source: Adapted from Ravishankara et al., 2012.

5.4.1 Particulate Matter Effects on Climate

PM can affect climate through direct interactions with radiation, predominantly through scattering or absorption of incoming solar radiation (sunlight). Sulphate, nitrate, ammonium, and secondary organic aerosol particulates predominantly scatter incoming solar radiation, some of which is reflected back into space; therefore, these PM species have a net cooling effect on climate (Boucher et al., 2013). Conversely, PM species that absorb incoming solar radiation, such as black carbon and certain types of organic particulates referred to as brown carbon, have a net warming effect (Bond et al., 2013). The presence of clouds influences these effects (Chand et al., 2009), and, under clear skies, so does the nature of the Earth's underlying surface. PM over highly reflective surfaces such as snow has a stronger net warming effect than over less reflective surfaces because of the relatively larger fraction of incoming solar radiation that would have been reflected back to space without the presence of PM (Haywood & Shine, 1995). In addition, the mix of compounds within individual aerosol particles can also affect the direct radiative effects of PM. For example, absorption of solar radiation by black carbon is enhanced when PM is coated by more weakly absorbing compounds, such as sulphates (Jacobson, 2001; Liu et al., 2017).

PM can also affect climate through its effects on clouds, due to its role as the "seeds" upon which water may condense in either liquid or solid form — acting as cloud condensation nuclei and ice nuclei, respectively. Aside from the mass of particles in the atmosphere, there is also strong evidence that the number of particles influences cloud formation, with increased atmospheric particle number concentrations (total number of particles per unit volume of air) resulting in liquid water clouds with a larger number concentration of smaller cloud droplets, making the clouds more reflective and producing a cooling effect on climate (Boucher et al., 2013). However, PM–cloud interactions are subject to a large number of feedback processes that can locally modify the magnitude and direction of the radiative effects of these interactions, notably through changing precipitation production, which may alter the lifetime or spatial extent of the cloud, and changes involving ice particles in clouds (Fan et al., 2016). For these reasons, the effects of PM on clouds and climate are still uncertain (Seinfeld et al., 2016).

The Fifth Assessment Report[3] of the Intergovernmental Panel on Climate Change (IPCC) assessed the global net radiative forcing from anthropogenic PM, including through both direct radiative effects and aerosol–cloud interactions, at −0.90 (5% to 95% confidence interval [CI] −1.9 to −0.1) W/m^2 (i.e., a net cooling effect). This likely offsets a significant fraction of the estimated +2.83 (5% to 95% CI +2.54 to +3.12) W/m^2 radiative forcing due to the change in concentration of the well-mixed GHGs[4] (CO_2, methane, nitrous oxide, and halocarbons) between 1750 and 2011. However, the net forcing from PM combines significant positive forcing (warming) from black carbon emitted from fossil fuel and biofuel use, and negative forcing (cooling) from sulphates, nitrates, and primary and secondary organic aerosols. The net forcing from PM mentioned above does not include additional positive forcing from black carbon deposited on snow, estimated at +0.04 W/m^2, which has a two to four times larger impact on surface temperatures than CO_2 per unit radiative forcing (Myhre et al., 2013; Skiles et al., 2018).

---

[3] The uncertainties in the estimates of radiative forcing quoted here are the 5% to 95% confidence interval as given by the IPCC Fifth Assessment Report.

[4] Well-mixed GHGs are those with long residency time in the atmosphere, which results in their having a relatively homogeneous concentration throughout the troposphere.

Due to the short atmospheric lifetime (approximately 1 week) and highly variable spatial distribution of PM, impacts on radiation can be much larger on regional scales than the global average estimates given above. The chemical composition of PM influences the pathway and fate of incoming solar radiation as it travels through the atmosphere. Both PM that strongly scatters and PM that strongly absorbs incoming radiation reduce the amount of solar radiation reaching the Earth's surface. However, only the latter will warm the atmospheric layer where it occurs. These effects could have significant impacts on surface temperatures and rainfall by reducing the amount of energy absorbed at the surface and, in the case of absorbing PM, heating the atmosphere aloft (Ramanathan et al., 2005; Liu et al., 2012). Modelling studies have shown that, due to the response of the global wind and weather patterns, the effects of SLCFs on climate are not limited to the regions where the emissions occur and may be even larger in regions well removed from the emission sources (Shindell et al., 2015; Kasoar et al., 2016).

5.4.2 Methane and Ozone Effects on Climate

Methane, with an atmospheric lifetime of approximately 9.1 years (Prather et al., 2012), is relatively well mixed throughout the atmosphere and is emitted from a large variety of sources (e.g., natural wetlands, geological seeps, rice cultivation, ruminants, and fossil fuel extraction and use) with an approximately equal contribution from natural and anthropogenic sources at the global scale (Saunois et al., 2016). Methane acts directly as a GHG, with an estimated radiative forcing of +0.48 (5% to 95% CI +0.43 to +0.53) W/m^2 due to the approximately 150% increase in concentrations over the period 1750 to 2011 (Myhre et al., 2013). Methane is also chemically reactive in the atmosphere, participating in the photochemical production of ozone. Ozone, in addition to being an air pollutant, acts as a GHG (Forster & Shine, 1997), and methane, therefore, makes additional contributions to climate forcing through changes in ozone.

Ozone is formed photochemically in the lower atmosphere from precursor compounds, including methane, other VOCs, and NO_x. Ozone is a highly reactive gas, and few reliable measurements are available from before the 1950s (Cooper et al., 2014). As a result, estimates of the change in ground-level ozone and the effects of tropospheric ozone on climate are based on numerical models and are more uncertain than for the long-lived GHG gases (e.g., CO_2). These estimates show that the increase in ozone from pre-industrial levels due to precursor emissions (methane, CO, non-methane VOCs, and NO_x) has resulted in an estimated radiative forcing of +0.50 (5% to 95% CI +0.30 to 0.70) W/m^2, of which +0.24 (5% to 95% CI +0.11 to +0.37) W/m^2 is attributed to ozone from methane (Myhre et al., 2013).

5.4.3 Effects of Climate Change on Air Quality

Changes in climate can have significant effects on air quality. Studies show a strong correlation between temperature and ground-level ozone (Camalier et al., 2007). This relationship results from various factors: direct temperature effects on chemical reactions that produce ozone; increases in biogenic emissions of ozone precursors (e.g., VOC emissions from vegetation) with temperature; and the correlation of higher temperatures with meteorological conditions that are conducive to the production of ozone (Steiner et al., 2006; Schnell & Prather, 2017). While tropospheric ozone in industrial regions in some seasons is expected to increase with increasing temperatures, climate change alone is expected to result in a decrease in the global background concentration[5] of ozone, as increases in water vapour reduce levels of ozone through increased photochemical destruction in remote locations (Wu et al., 2008a; Stevenson et al., 2013).

The direct impacts of increased temperature on PM vary by season. Warmer temperatures in winter will result in a larger fraction of certain PM components remaining in the gas phase, resulting in lower PM concentrations. However, warmer temperatures in summer will favour the increased production of sulphates and higher PM concentrations (Dawson et al., 2007). Observations across the United States suggest additional contributions to PM concentrations result from other temperature-related factors (Tai et al., 2010). In particular, increased emissions of VOCs from biogenic sources, such as trees, with increasing temperature have been shown to have a strong impact on the formation of organic components of PM (Day & Pandis, 2011).

Climate change includes many complex changes beyond increasing temperature. By more rapidly warming the Arctic, climate change may lead to a greater persistence of hot and dry conditions associated with slow-moving high-pressure systems (Coumou et al., 2018). Summertime episodes of poor air quality over Southeastern Canada and the Northeastern United States are frequently associated with these large, slow-moving high-pressure systems and are brought to an end by the passage of a mid-latitude cyclone and associated cold front. Less frequent summertime mid-latitude cyclones have been observed in the recent past, negatively affecting air quality (Leibensperger et al., 2008), and less frequent mid-latitude cyclone passage has been projected under future climate scenarios in several studies (Mickley et al., 2004; Wu et al., 2008b). However, the magnitude of these specific effects on future changes in air pollution is uncertain. Tai et al. (2012) found that the changes in the frequency of mid-latitude cyclone passage had only a small impact on PM, while Horton et al. (2014), from an analysis of 15 different climate models, found relatively small increases in stagnation events, which may lead to higher pollutant concentrations for 2080 to 2099 along the west coast of North America, Northeastern United States, and Southeastern Canada.

5.4.4 Air Quality Projections Under a Changing Climate

A large number of studies have used regional-scale air quality models, combined with future climate projections, to estimate future changes in tropospheric ozone and PM across North America. It should be noted that projections of future air quality depend on model projections of changes in the physical climate, and these changes are more uncertain at regional (sub-continental and smaller) scales for a variety of reasons (Deser et al., 2012; Mearns et al., 2013; Giorgi & Gutowski, 2015). Future air quality will also be affected by changes in air pollutant and precursor emissions. Many studies estimate the effects of climate change alone by making future projections using constant, present-day emissions of air pollutants. The impact of climate change alone on air quality, discussed in this section, is frequently referred to as “the climate penalty” (Wu et al., 2008b).

While differences in modelling assumptions and scenarios make detailed comparisons difficult, some general conclusions can be drawn. Available studies generally find that, relative to current conditions,

---

[5] The “background concentration” of ozone refers to the concentration of ozone found in remote regions, far from significant anthropogenic emission sources. The background concentration varies with latitude, altitude, and season and is, nonetheless, significantly influenced by anthropogenic sources.

climate change will cause small increases across large parts of North America of 2 to 3 parts per billion by volume in the daytime average ozone concentration during the summer (June to August) by the middle of the century (Hogrefe et al., 2004; Nolte et al., 2008; Lam et al., 2011; Kelly et al., 2012). Isolated increases of about 5 parts per billion by volume were also found across different modelling studies. The largest increases were projected over industrialized regions, such as the Northeastern United States and adjacent regions of Southeastern Canada. These studies have also found some evidence for a larger impact of climate change on days with the highest ozone concentrations (Hogrefe et al., 2004; Nolte et al., 2008) and indications that climate change may lengthen the ozone season (Nolte et al., 2008; Trail et al., 2014).

Estimates of the impact of climate change alone on PM are more uncertain, as there are fewer studies, and the physical and chemical processes affecting PM are more complex and more sensitive to changes in physical climate variables that climate models represent poorly, such as the frequency of precipitation. Tagaris et al. (2007) found decreases of 10% to 20% (1 to 2 μg/m^3) in $PM_{2.5}$ over central and southeastern parts of the United States by about 2050, due to higher precipitation, although only three summers were simulated. Conversely, Kelly et al. (2012) simulated 10 summers leading to 2050 and found the effects of climate change alone resulted in increases in summer $PM_{2.5}$ of 0.5 to 1.0 μg/m^3 over much of the Eastern United States, extending into Southwestern Ontario. Increases of 0.3 to 0.5 μg/m^3 were projected over the broader region of Southern Ontario and Southern Quebec. The increases in $PM_{2.5}$ resulted from the net effect of increases in sulphates and secondary organic aerosols and decreases in nitrates. These changes can be contrasted with those found by Trail et al. (2014), who projected decreases in summer $PM_{2.5}$ of 1 to 2 μg/m^3 for periods around 2050 across most of the Central United States to the Great Lakes, due to increased dispersion of pollutants in the atmosphere because of projected changes in wind speed and precipitation. These results demonstrate the level of uncertainty in our current understanding of certain aspects of changes in physical climate and associated air quality projections, as represented by existing models. For changes in annual average $PM_{2.5}$, the available studies suggest very small changes in regional average values (Tagaris et al., 2007; Lam et al., 2011; Trail et al., 2014).

Two important caveats to the preceding discussion are worth noting. First, ozone and PM precursor emissions have been reduced significantly over the recent past in the United States and Canada because of emissions controls put in place to reduce the environmental and human health impacts of air pollution (Amann et al., 2013; Stieb et al., 2015; Fann et al., 2017; Jiang et al., 2018; Zhang et al., 2018). Air pollutant emissions are projected to decrease further over the coming decades as a result of air quality regulations, and available projections show significant net improvements in air quality can be realized in the future, accounting for the combined effects of anticipated reductions in air pollutant emissions and the negative impact of climate change (Tagaris et al., 2007; Nolte et al., 2008; Lam et al., 2011; Kelly et al., 2012; Pfister et al., 2014; Trail et al., 2014; Yahya et al., 2017). However, additional reductions in air pollution emissions would be required to counteract the effect of the climate penalty. Second, future ozone concentrations in North America will be affected by changes in the global concentration of methane (Yahya et al., 2017). Increases in the concentrations of methane result in a higher global background concentration of ozone (Stevenson et al., 2013), which affects ozone concentrations in North America, as regional-scale episodes of high ozone build on top of the global background.

Another impact of climate change on air quality is the expected increase in the length and severity of the wildfire season (Flannigan et al., 2013). Estimates of the effects of climate change on wildfire emissions of black and organic carbon aerosols suggest increases of 80% to 150% by 2050 (compared to 1997 to 2001) over the Western United States (Yue et al., 2013), with an increase in summer average $PM_{2.5}$ concentrations of 20% to 60% (Val Martin et al., 2015). These projections were based on increases in the average annual area burned each year of between 25% and nearly 170%, which are comparable to estimates of the increase in area burned by mid-century for Canada (Boulanger et al., 2014). Note that the Val Martin et al. (2015) study did not include any increase in wildfires in the Eastern United States because any climate change effect was found to be negligible, while projections of wildfires in Canada show increases from coast to coast (Boulanger et al., 2014; Wang et al., 2017). For more information on the impact of climate change on wildfires in Canada and the associated health impacts, see section 5.6 Climate Change and Air Pollution from Wildfires.

5.4.5 Air Quality Policies and Climate Change Mitigation

Because of the relatively short time that air pollutants remain in the atmosphere, reductions in their emissions quickly affect their atmospheric concentrations and the associated radiative forcing of these compounds. Actions to reduce certain air pollutants that are SLCFs, such as black carbon, are therefore recognized as a way to reduce climate warming over the near term (CCAC, 2020). This would allow more time for international CO_2 mitigation efforts to have an impact before critical temperature thresholds are crossed, while also simultaneously reducing the health and environmental burden of air pollution (IPCC, 2018). Shindell et al. (2012) projected a reduction in global mean warming of 0.5°C by 2050, relative to pre-industrial levels, resulting from a suite of controls on methane and black carbon emissions, and estimated additional, significant benefits for human health and agricultural yields. A smaller cooling benefit of 0.22°C by 2050 from a range of emission controls designed to maximize the climate benefit — largely targeting methane and black carbon — was found by Stohl et al. (2015). The authors also estimated that a warming of nearly 0.7°C would result from the complete removal of land-based anthropogenic SO_2 sources, illustrating the significant cooling effect of sulphate PM from SO_2 precursor emissions.

These results demonstrate the need to evaluate climate change and air quality policies in an integrated manner. Targeting SLCFs that have a net warming effect on climate, namely methane and black carbon, would have simultaneous benefits for climate and air quality. Conversely, certain actions designed to improve air quality may have unintended negative consequences for climate change (Stohl et al., 2015; Partanen et al., 2018). However, the strategic application of air pollution emission controls can have significant co-benefits for climate change, and well-designed GHG mitigation strategies, such as those targeting reductions in the use of fossil fuels, can have clear co-benefits for air quality and, hence, population health (Vandyck et al., 2018). Co-benefits for health associated with GHG reductions are reviewed in section 5.5 Air Pollution Health Impacts Due to Climate Change and the Health Co-Benefits of Greenhouse Gas Mitigation and also highlighted in Chapter 10: Adaptation and Health System Resilience of this report.

5.4.6 Key Uncertainties

One of the most significant risks to air quality throughout Canada associated with climate change is the projected increase in emissions from wildfires. An improved understanding of the effects of climate change on wildfires requires further development of the capacity to simulate interactions between climate and fire risk, taking into account anthropogenic influences such as fire management through suppression, for example, in terrestrial carbon cycle models. A second significant uncertainty in projections of climate change effects on air quality is the impact of future changes in biogenic emissions. As discussed above, increasing biogenic emissions due to warmer temperatures is an important mechanism by which climate change negatively affects air quality. However, the likely effects of higher CO_2 and water stress, as well as changes in the distribution of vegetation types on biogenic emissions, are not well understood (Fiore et al., 2015). Last, many of the studies estimating the impact of climate change on air quality have been based on only a few years of simulation because of the computational cost of the models. The short time periods make it difficult to remove the influence of random year-to-year variability from the effects of climate change (Barnes et al., 2016) or to robustly analyze climate change impacts on more extreme, but less frequent, climate events.

5.5 Air Pollution Health Impacts Due to Climate Change and the Health Co-Benefits of Greenhouse Gas Mitigation

Exposure to air pollutants can cause a range of adverse health effects, and even small increases in exposure are associated with an increase in risk (see section 5.3 Health Effects of Outdoor Air Pollution). It is anticipated that climate change will worsen air quality both globally and in Canada, due to the climate penalty (see section 5.4 Interactions Between Climate Change and Outdoor Air Pollution). Studies have tried to estimate the effect of the climate penalty on two key air pollutants that have major health impacts, ozone and $PM_{2.5}$, under possible future climates (Ebi & McGregor, 2008; Gao et al., 2013; Turnock et al., 2019). Conversely, many strategies that target long-lived GHG and SLCF emission reductions, such as reducing the use of fossil fuels, can be expected to reduce air pollutant emissions and improve air quality. The associated population health benefits are referred to as “air pollution health co-benefits” because they are not the primary motive behind development of those policies. Consequently, efforts undertaken to mitigate climate change in Canada may also have important air pollution health co-benefits. Many initiatives to reduce climate change are underway at global (e.g., the Paris Agreement [UNFCCC, 2015]), national (e.g., *A Healthy Environment and a Healthy Economy* [ECCC, 2020]), provincial (e.g., *Plan d’action 2013-2020 sur les changements climatiques* [Ministère de l'Environnement et *Lutte contre les changements climatiques, 2012*]), and municipal (e.g., *Climate 2050 Strategic Framework* [Metro Vancouver, 2018]) levels.

Scientists can quantify the number of population health impacts, such as premature deaths or asthma episodes, attributable to a given increase or decrease in air pollution. Economists estimate the social welfare value of these health impacts, including such things as medical costs, reduced workplace productivity, and the effects of increased mortality risk. These methods allow for estimation of the health impacts and associated monetary value of both the costs of air pollution increases under a warming climate and of the air pollution health co-benefits of mitigating climate pollutants. It is important to note that modelled air pollution health impacts or benefits of individual scenarios are likely underestimated, as not all adverse health outcomes that have been associated with air pollution exposure can be quantified and valued in this manner.

This section of the chapter provides a review of published studies that have assessed the air pollution health impacts of the climate penalty and the potential air pollution health co-benefits of climate mitigation, including the associated economic valuation of those outcomes. It also presents information on the health impacts of the climate penalty and potential health co-benefits of GHG mitigation in Canada, including those associated with eliminating coal-fired electricity generation in Canada.

5.5.1 Air Pollution Health Impacts Due to Climate Change

Studies have investigated how a changing climate could influence future population health impacts of air pollution. Researchers have assessed these health impacts both regionally and globally, under various climate projection scenarios, including the Representative Concentration Pathways (RCPs)[6] and the Special Report on Emissions Scenarios (SRES)[7] published by the IPCC, and for multiple future time periods. Most studies have examined the influence of climate change on ozone, and some have also addressed $PM_{2.5}$. Both premature mortality and morbidity outcomes have been considered. In the studies reviewed here, the authors estimated the incremental air pollution health impacts in the future due to climate change alone (i.e., the impact of a warming climate on air pollution concentrations), keeping other factors constant.

More recent studies have used the RCPs described in the Fifth Assessment Report of the IPCC (2014) and other publications (van Vuuren et al., 2011; van Vuuren & Carter, 2014) to model future climate change projections. The RCPs, identified as RCP2.6, RCP4.5, RCP6.0, and RCP8.5 (referring to different levels of radiative forcing in watts per square metre), represent global climate policy scenarios ranging from more stringent (RCP2.6) to less stringent (RCP8.5) in terms of GHG reductions and concentration trajectories. The RCPs also include projected policies to control air pollution. The more stringent RCP scenarios are also associated with lower air pollutant emissions globally and in Canada, given that many changes in sector activities to reduce GHG emissions (e.g., transition to cleaner energy sources) would also reduce air pollution emissions. The four RCP scenarios project average global temperature increases for 2081 to 2100 (relative to 1986 to 2005) of 1.0°C, 1.8°C, 2.2°C, and 3.7°C, respectively. The RCPs are more in-depth explorations of the impacts of potential mitigation strategies on GHG emissions and are an extension and further development of the GHG emissions-based SRES projections used in previous IPCC assessment reports (IPCC, 2014).

There are no recent published analyses from Canada using the RCP scenarios, but findings have been published for the United States and for global analyses. In a national analysis, Fann et al. (2015) estimated

---

[6] The Representative Concentration Pathways (RCPs) are four different plausible consequences of future climate policy scenarios (RCP2.6, RCP4.5, RCP6.0, RCP8.5) with emission and concentration trajectories of GHGs and other air pollutants and land-use pathways (IPCC, 2014).

[7] The SRESs describe emission projection scenarios organized into four storylines — A1 (subgroup A1B), A2, B1, and B2 — describing plausible futures according to assumptions for various driving forces, such as social and economic development, technology innovation, and environmental changes (Nakicenovic et.al, 2000).

that summer average eight-hour daily maximum ozone concentrations would increase by about 1 to 5 ppb for parts of the United States in the near future (2030) due to climate change, under RCP6.0 and RCP8.5 scenarios. This analysis compared air quality impacts under the 2030 climate, as projected by the RCPs, with air quality impacts under climatic conditions in 2000. The ozone-related health outcomes in 2030 are presented in Table 5.1. The ranges in the projections reflect estimates of the health effects of ozone from multiple epidemiological studies. Overall, health impacts and the associated societal costs under RCP8.5 are projected to be an order of magnitude higher than under RCP6.0.

Table 5.1 Projected increases in annual ozone-related health impacts (counts) in 2030 in the United States under two climate scenarios

CLIMATE PROJECTION	PREMATURE DEATHS	HOSPITAL ADMISSIONS	EMERGENCY ROOM VISITS	ACUTE RESPIRATORY SYMPTOMS	MISSED DAYS OF SCHOOL	TOTAL ECONOMIC VALUATION (IN 2010 USD)
RCP6.0	37–170	360	89	210,000	67,000	0.32–1.4 billion
RCP8.5	420–1900	3900	1200	1,900,000	650,000	3.6–15 billion

Source: Fann et al., 2015

Much lower US health impacts for the year 2050 (less than 100 ozone-related premature deaths) were reported in a separate analysis of the climate penalty under RCP4.5 and RCP8.5 (Stowell et al., 2017).

Climate change impacts on both ozone and $PM_{2.5}$ health effects under the RCP8.5 scenario have also been assessed globally, with estimates of 43,600 and 215,000 excess deaths annually by 2100 due to ozone and $PM_{2.5}$ increases, respectively, including almost 10,000 deaths in North America. Global impacts in the near term (2030) were estimated to be almost 60,000 excess deaths annually for the two pollutants combined (Silva et al., 2017). Multiple models show an increase in $PM_{2.5}$ health risks due to the climate penalty in nine of the 10 regions of the world, including North America (Park et al., 2020).

Several investigators have applied the older SRES A1B scenario, which is roughly comparable to RCP6.0 and estimates a global temperature increase of 1.59°C in 2050, to project the air pollution health impacts of climate change. These studies estimate hundreds of ozone-related and several thousand $PM_{2.5}$-related deaths annually in 2050 in the United States due to the warming climate, along with thousands of hospital visits, bronchitis incidents, and asthma attacks (Selin et al., 2009; Tagaris et al., 2009). Higher estimates of ozone and $PM_{2.5}$-related deaths attributable to climate change were reported for the year 2090, with 13,000 deaths in North America and more than 100,000 deaths globally (Fang et al., 2013). A study of 50 US cities under the SRES A2 scenario in the 2050s, which predicts a rise in average summer temperature for the Eastern United

States of 1.6°C to 3.2°C, projected an increase in mortality (0.11% to 0.27%) and hospital admissions for asthma (2.1%) and COPD (0.2% to 1.6%) due to ozone increases under the future climate (Bell et al., 2007).

A Canadian analysis estimated the air pollution health impacts for a hypothetical increase of 4°C in ambient temperatures for 2002, with anthropogenic air pollution emissions held at 2002 levels (Séguin, 2008). Nationally, increases in summer ozone concentrations due to the higher temperature were associated with 156 excess premature deaths and a total economic value of $750 million due to mortality and morbidity outcomes. This increased to 658 premature deaths ($3.2 billion) when increases in biogenic emissions, expected under a warmer climate, were also included. However, the same study projected, with considerable uncertainty, a net reduction in $PM_{2.5}$ concentrations under the warmer climate, so that the net effect of the two pollutants under a 4°C increase was estimated to be an excess of approximately 300 premature deaths and a social cost of $1.4 billion.

Overall, these studies show that climate change is expected to worsen air quality and cause considerable deaths and illnesses related to air pollution, with hundreds to thousands of excess annual deaths in North America over this century and 100,000 annual excess deaths estimated globally by 2100. Studies that examined both ozone and $PM_{2.5}$ generally estimated that effects of climate change on $PM_{2.5}$-related health outcomes are about an order of magnitude higher than those associated with ozone.

5.5.2 Air Pollution Health Co-Benefits of Mitigation of Greenhouse Gas Emissions

Mitigation measures to reduce GHGs and SLCFs help stabilize the climate by reducing pollutants that play a role in increasing global temperature and accelerating climate change. However, climate change mitigation strategies can also have a range of indirect health co-benefits (Markandya et al., 2018). Key among these are health co-benefits from reductions in air pollution because of the following: climate mitigation will limit the climate penalty; some climate pollutants (e.g., black carbon, ozone) contribute to air pollution either directly or as precursors; and changes in industrial and transportation activities to reduce long-lived GHG and/or SLCF emissions can also reduce co-emitted conventional air pollutants (Smith et al., 2014). For example, the widespread use of electric vehicles powered by low-carbon energy can be a strategy for GHG mitigation that could also result in improved air quality. Consequently, climate change mitigation measures represent an opportunity to integrate objectives to stabilize the climate with those to reduce air pollution health impacts, which may offset the costs of implementing the measures (Thompson et al., 2016).

Studies examining air pollution health co-benefits associated with GHG mitigation strategies typically compare a policy scenario that includes strong GHG mitigation and accompanying reductions in air pollutant emissions with a scenario with limited GHG mitigation and associated higher air pollutant emissions. Economic valuation of air pollution health co-benefits allows co-benefits to be incorporated into a broader climate change framework, and the cost-effectiveness and potential optimization of various GHG mitigation measures, which can impose an economic burden on society, can be examined. Studies report monetized health co-benefits as total dollar value or as the marginal benefit per tonne of CO_2 equivalent that is reduced. Both $PM_{2.5}$- and ozone-related health co-benefits have been evaluated in the studies reviewed below, unless otherwise stated.

In a global analysis of GHG mitigation measures associated with RCP4.5, it was estimated that Canadian air pollution health co-benefits would range from about 4500 to 6500 avoided premature deaths annually between 2030 and 2100, relative to the reference scenario associated with the RCP projections (West et al., 2013) (Table 5.2). US co-benefits were estimated to be about an order of magnitude higher (West et al., 2013; Zhang et al., 2017). Globally, changes in emissions were estimated to avoid 0.5 million, 1.3 million, and 2.2 million deaths from combined exposure to ozone and $PM_{2.5}$ in 2030, 2050, and 2100, respectively. The global average monetized marginal health co-benefits were estimated to be \$50 to \$380 per tonne of CO_2 (tCO_2) (in 2005 USD), which exceed the marginal GHG abatement costs in 2030 and 2050 in this analysis, and partially offset the costs in 2100 (West et al., 2013). For the United States, the marginal value of avoided ozone- and $PM_{2.5}$-related deaths was estimated to be \$45 to \$137/tCO_2 (in 2005 USD) (Zhang et al., 2017).

Table 5.2 Air pollution health co-benefits associated with the projected scenario of RCP4.5 (annual avoided premature deaths)

STUDY	2030		2050		2100	
	OZONE	$PM_{2.5}$	OZONE	$PM_{2.5}$	OZONE	$PM_{2.5}$
Canada (West et al., 2013)	368	4270	792	5750	1180	2880
United States (West et al., 2013)	2440	19,300	7550	29,500	24,800	35,400
United States (Zhang et al., 2017)			8000	16,000		
Global (West et al., 2013)	0.5 million		1.3 million		2.2 million	

Other studies have evaluated the avoided air pollution deaths associated with climate mitigation pathways based on specific targets for increases in global mean surface temperature in the future compared to a reference scenario. The United States air pollution heath co-benefits of reducing the climate penalty were estimated for two climate mitigation scenarios, reflecting a 2.5°C and 2.0°C increase in global mean surface temperature (Saari et al., 2019). The two mitigation scenarios reduced the combined ozone and $PM_{2.5}$ health impacts by thousands and tens of thousands of deaths in 2050 and 2100, respectively, with 40% greater $PM_{2.5}$ benefits by 2100, under the more stringent policy compared to the less stringent policy. In 2100, air pollution (ozone and $PM_{2.5}$) risks associated with climate change were attenuated by 70% to 88% in the policy scenarios. Similarly, United States health co-benefits of limiting global warming in 2100 to a 1.5°C rise was estimated to result in 11,000 and 52,000 avoided premature deaths across the United States in 2050 and 2100, respectively, compared to a reference scenario with a 6°C increase in global mean surface temperature. These health co-benefits were valued at 150 billion USD and 1.3 trillion USD (in 2005 USD), respectively, which were equivalent to a rate of $25 (95% CI $9–$42) per tonne of CO_2 equivalent (tCO_2e) and $122 (95% CI $45–$207)/tCO_2e, respectively (Garcia-Menendez et al., 2015).

In a global analysis, Vandyck et al. (2018) examined the air quality implications in 2100 of two GHG mitigation scenarios, the first based on GHG emission reductions implied by the Nationally Determined Contributions,[8] leading to a 2.5°C to 3.2°C rise in global mean temperature by 2100, and the second a 2°C rise by 2100. Globally, avoided deaths due to air pollution in 2050 were estimated at 0.3 to 0.5 million for the Nationally Determined Contributions scenario and 0.7 to 1.5 million for the 2°C scenario, including about 20,000 to 25,000 avoided deaths in the United States. The timing of taking action can affect the magnitude of the accrued health co-benefits. For example, it was estimated that about 150 million deaths due to air pollution globally could be avoided over this century if carbon emission reductions to limit warming to 2ºC are implemented sooner rather than later in the 21st century (Shindell et al., 2018).

Another global study specifically examined the potential air pollution health co-benefits of meeting the Paris Agreement temperature change targets of 1.5°C and 2°C, and compared them to the associated mitigation costs of the climate policy scenarios (Markandya et al., 2018). Globally, the economic value of the health co-benefits were estimated to outweigh the mitigation costs for all scenarios examined, with the ratio of health co-benefits to climate change mitigation costs ranging from 1.4 to 2.45. The greatest potential health co-benefits were observed for China and India, but significant co-benefits were also estimated for the European Union and the United States. Overall, climate change mitigation measures to meet the targets established in the Paris Agreement can be cost-effective under the modelled scenarios and, depending on geographical regions, when air quality associated health co-benefits are considered.

Other studies have estimated thousands of avoided deaths due to ozone for North America and hundreds of thousands globally under various climate change mitigation scenarios, including those targeting methane (West et al., 2007; West et al., 2012). The latter were valued at $13 to $17/$tCO_2e$ (2005 USD) in 2030, which outweighed the costs of measures to reduce methane emissions (West et al., 2012).

Some studies have examined the co-benefits of climate change mitigation strategies targeting specific SLCFs or emission sectors. The annual health co-benefits of full electrification of the vehicle fleet in the Greater Toronto and Hamilton Area (Canada) were estimated at 260 and 330 avoided deaths, based on electricity

---

[8] Nationally Determined Contributions are country-specific emission reduction pledges to meet the Paris Agreement under the United Nations Framework Convention on Climate Change (UNFCCC, 2015).

derived from natural gas or from renewables, respectively (Gai et al., 2020). The North American health co-benefits of ambitious clean energy and clean transportation scenarios in the United States were estimated at 175,000 and 120,000 avoided deaths due to air pollution, for the 2015 to 2030 period, respectively, with additional annual benefits thereafter (Shindell et al., 2016). The majority of benefits were from $PM_{2.5}$ reductions, with a smaller fraction from ozone reductions, and 3% to 4% of all benefits (thousands of avoided deaths for 2015 to 2030 period) were estimated for Canada. Near-term US benefits of the policies were valued at 250 billion USD (95% CI 140 billion to 1.05 trillion USD) annually.

A global analysis of strategies to reduce methane and black carbon reported that implementation of the measures would avoid 0.7 to 4.7 million deaths per year globally, with the largest impact attributable to black carbon reductions, and a marginal benefit of 700 to 5000 USD per tonne of methane reduction (Shindell et al., 2012). Carbon policies for the United States addressing clean electricity production, on-road transportation, and an economy-wide cap-and-trade system were estimated to have air pollution health co-benefits associated with $PM_{2.5}$ and ozone reductions that would offset approximately 26% to 1050% of GHG mitigation costs (Thompson et al., 2014). The marginal health co-benefits of $PM_{2.5}$ reductions under the cap-and-trade scenario were estimated at \$6/tCO_2, which exceeded the implementation costs, and at \$8/tCO_2 (in 2005 USD) under the clean energy scenario, which partially offset the implementation costs (Saari et al., 2015). Marginal health co-benefits of \$148/tCO_2 (in 2006 USD) compared to GHG mitigation costs of \$126/tCO_2 under a clean energy scenario, and marginal health co-benefits of \$80/tCO_2 compared to costs of \$15/tCO_2 under the cap-and-trade scenario were reported for a subnational carbon policy in the Northeastern United States. It was noted that costs varied widely due to regional differences in energy consumption (Thompson et al., 2016).

It is important to consider all aspects of emission changes when evaluating the health co-benefits of climate change mitigation strategies. The US air pollution health co-benefits associated with electric vehicle (EV) adoption were found to vary, depending on the source mix of electricity generation (Peters et al., 2020). It was estimated that 25% of EV use powered by the current US electricity grid mix would result in 437 (95% CI 295 to 578) avoided premature deaths due to $PM_{2.5}$ and 98 (95% CI 33 to 162) due to ozone reductions annually, with a total value of about 16.8 billion USD. The health co-benefits were estimated to be about twice as high for the same EV adoption rate if the fraction of emission-free energy sources in the grid mix was doubled, although the associated reduction in CO_2 emissions would only increase by about 10%.

Overall, studies of the potential air pollution health co-benefits associated with climate mitigation scenarios targeting maximum increases in global mean temperature of about 1.5°C to 3°C consistently indicate that substantial health benefits would be realized. Annually, the burden of disease would be reduced by up to tens of thousands of premature deaths related to $PM_{2.5}$ and ozone in the United States in 2050 and 2100, and by up to 1 million to 2 million premature deaths globally. When directly compared in a study, more stringent mitigation policies confer greater co-benefits, and there is evidence that larger co-benefits will be realized if policies are implemented sooner rather than later. Air pollution health co-benefits represent a significant public health gain for individual countries and globally, and the socio-economic value can contribute to or even offset the costs of mitigation. Moreover, the joint consideration of climate change and air pollution mitigation options provides a framework to optimize the overall public health benefits and to identify potential unintended consequences, such as inadvertent risks to health. This was made evident following the promotion of diesel-fuelled vehicles in some countries because of their lower GHG emissions, which resulted in elevated emissions of air pollutants (Cames & Helmers, 2013; Haines, 2017).

Box 5.1 The elimination of coal-fired power in Canada: A case study of air pollution health co-benefits

In 2012, the Government of Canada published the *Reduction of Carbon Dioxide Emissions from Coal-fired Generation of Electricity Regulations* (Government of Canada, 2012). These regulations set a CO_2 emission performance standard for new coal-fired electricity generation units and for those reaching the end of their useful lifespan of 50 years. The objective was to ensure a permanent transition to power generation options that emitted low amounts or no CO_2, such as renewable energy or high-efficiency natural gas, and to ensure that no new high-emitting units would be built in Canada. In 2010, Alberta, Ontario, Saskatchewan, Nova Scotia, New Brunswick, and Manitoba had coal-fired power generation capacity, in decreasing order of magnitude.

It was estimated that during the 2015 to 2035 period the Regulations would result in avoiding 6820 megawatts of coal-fired electricity production, with 74% in Alberta, 14% in Nova Scotia, and 11% in Saskatchewan. This would result in emissions reductions of 219 Mt CO_2 equivalent, which was valued at $5.6 billion (in 2010 CAD), based on the social cost of carbon[9] (Government of Canada, 2012).

In addition to emitting GHGs, coal-fired power generation is a source of air pollutant emissions. The air quality and health co-benefits of the reductions in air pollution emissions associated with the Regulations were estimated by the Government of Canada. The following cumulative reductions in air pollutant emissions were estimated for 2015 to 2035: 1156 kt SO_x, 546 kt NO_x, 9 kt $PM_{2.5}$, and 48 kt CO. Air quality modelling was undertaken to see how these changes in emissions would affect air pollutant concentrations.

For the 2015 to 2035 period, it was estimated that the incremental improvements in air quality associated with the Regulations would result in avoiding 900 premature deaths, 800 hospital visits, 120,000 asthma episodes, and 2.7 million episode-days when individuals experience breathing difficulty or must reduce activity. Of these cumulative impacts, 590, 140, 80, and 57 avoided deaths were estimated for Alberta, Saskatchewan, Manitoba, and Ontario, respectively. The national health co-benefits were estimated to have a value in 2015 of $4.2 billion (in 2010$), of which about 70% were attributable to reductions in ambient $PM_{2.5}$ and about 26% to reductions in ambient ozone. The air pollution health co-benefits of $4.2 billion contributed a significant portion of the estimated value of the overall benefits of the Regulations ($23.3 billion), which was found to outweigh the estimated associated costs of $16.1 billion (Government of Canada, 2012). The Regulations were later amended to accelerate the phase-out of conventional coal-fired power in Canada by 2030. It was estimated that the associated improvements in air quality from 2019 to 2055 would result in an additional 260 avoided premature deaths and 40,000 avoided asthma episodes, with a total socio-economic value of $1.2 billion (Government of Canada, 2018).

This case study illustrates that the air pollution health co-benefits of GHG mitigation strategies can represent a substantial value to Canadians, although reducing air pollution may not be the primary goal of the regulation or policy in question. In addition, while some of the climate change benefits of GHG reductions may be fully realized only over the medium to long term (Zickfeld & Herrington, 2015), air pollution health co-benefits start immediately following the reduction of emissions, accrue over time, and occur in the area where mitigation is implemented.

---

[9] The social cost of carbon is a monetary measure of the global damage expected from climate change from one additional tonne of CO_2 emissions in a given year.

5.5.3 Canadian Research Highlight: Quantifying Canadian Air Pollution Health Impacts of a Warming Climate and the Potential Health Co-Benefits of a Greenhouse Gas Mitigation Pathway

Health Canada undertook an analysis of the potential air pollution health impacts in Canada due to the climate penalty and the potential health co-benefits of a specific GHG mitigation pathway for the purposes of this assessment report. In a previous study, scientists at Environment and Climate Change Canada modelled mid-century air quality in North America to examine the influence of both a warming climate and GHG emission controls (Kelly et al., 2012). Air quality estimates were made using a unified regional air-quality modelling system (AURAMS), specifically, a comprehensive Canadian system. Specifically, air pollution concentrations were modelled for the June to August period under the current climate (based on meteorological data from 1997 to 2006) and under a future climate in 2041 to 2050 (based on meteorological conditions in SRES A2). Air pollutant emissions, which were derived from the 2002 Canadian emissions inventory, were held constant between the two scenarios in order to examine the impact of changing climatic conditions alone. Biogenic emissions and the effect of temperature on those emissions were included, but wildfire emission changes associated with climate change were not addressed. In addition, the study estimated air pollutant concentrations under the same 2041 to 2050 future climate, with a 2050 emissions inventory informed by RCP6.0 (a moderate GHG mitigation scenario), which was used to scale North American air pollutant emissions from 2002 to 2050 levels (Kelly et al., 2012). RCP6.0 was chosen in part due to the high level of detail available in its accompanying emissions activity changes, allowing a more precise calculation of the air pollutant reductions associated with the GHG-emitting activity reductions. This scaling reflects both air pollutant emission reductions associated with GHG mitigation as well as assumed air pollution controls. Air pollutant concentrations for each scenario were estimated on a national grid at 45 km resolution, and effects on both ozone and $PM_{2.5}$ were considered. As noted in section 5.4 Interactions Between Climate Change and Outdoor Air Pollution, effects of a changing climate on $PM_{2.5}$ are more complex and less well studied than effects on ozone. Therefore, projections of changes in $PM_{2.5}$ are more uncertain.

For the current analysis, the air quality output from the three scenarios modelled by Kelly et al. (2012) was used as input for Health Canada's Air Quality Benefits Assessment Tool (AQBAT 3.0). AQBAT is a national model used to estimate the human health (mortality and morbidity outcomes) and welfare benefits or damages associated with incremental changes in ambient concentrations of air pollutants, including $PM_{2.5}$ and ozone, in Canada (Health Canada, 2019). AQBAT includes concentration-response functions (CRFs), which characterize the increase in per capita risk of a given adverse health outcome per unit increase in ambient air pollutant concentration, based on published scientific studies. AQBAT estimates health impacts by combining the following parameters: the baseline risk of the adverse outcome in the population, population counts, CRFs, and the change in air pollutant concentrations between the scenarios. In addition, AQBAT provides economic valuation estimates for the health outcomes; these consider the potential social welfare consequences of the health outcomes, including medical costs, reduced workplace productivity, pain and suffering, and the effects of increased mortality risk.

The air pollutant concentration estimates from these three scenarios were used to answer the following questions.

What would the incremental air pollution health impacts in Canada be in 2050 due to:

- a changing climate, if air pollution precursor emissions remain unchanged (i.e., the climate penalty)?
- reductions in air pollution precursor emissions associated with the RCP6.0 projection, under future climate conditions?
- a changing climate and reductions in air pollution emissions associated with the RCP6.0 projection, compared to recent climatic conditions and emissions?

The Canadian population health impacts calculated for 2050 (June to August) and the associated economic valuation estimates are presented in Tables 5.3 and 5.4.

Table 5.3 National, provincial, and territorial ozone and $PM_{2.5}$ premature deaths and valuation for June to August[a]

REGION	CHANGE IN NUMBER OF PREMATURE DEATHS (ECONOMIC VALUATION)		
	IMPACT OF CHANGING CLIMATE IN 2050[b]	IMPACT OF REDUCED AIR POLLUTANT EMISSIONS AS PER RCP6.0 IN 2050[c]	IMPACT OF CHANGING CLIMATE AND REDUCED AIR POLLUTANT EMISSIONS AS PER RCP6.0 IN 2050[d]
Canada	850 ($2,700,000,000)	−5,200 (−$16,000,000,000)	−4,400 (−$14,000,000,000)
British Columbia	−56 (−$170,000,000)	−81 (−$250,000,000)	−140 (−$420,000,000)
Alberta	−9 (−$27,000,000)	−220 (−$680,000,000)	−230 (−$710,000,000)
Saskatchewan	9 ($29,000,000)	−58 (−$180,000,000)	−49 (−$150,000,000)
Manitoba	15 ($48,000,000)	−62 (−$190,000,000)	−47 (−$150,000,000)
Ontario	620 ($1,900,000,000)	−2,900 (−$9,100,000,000)	−2,300 (−$7,300,000,000)
Quebec	270 ($830,000,000)	−1,500 (−$4,700,000,000)	−1,200 (−$3,900,000,000)
New Brunswick	18 ($56,000,000)	−120 (−$380,000,000)	−110 (−$330,000,000)

REGION	CHANGE IN NUMBER OF PREMATURE DEATHS (ECONOMIC VALUATION)		
	IMPACT OF CHANGING CLIMATE IN 2050[b]	IMPACT OF REDUCED AIR POLLUTANT EMISSIONS AS PER RCP6.0 IN 2050[c]	IMPACT OF CHANGING CLIMATE AND REDUCED AIR POLLUTANT EMISSIONS AS PER RCP6.0 IN 2050[d]
Nova Scotia	−5 (−$16,000,000)	−160 (−$500,000,000)	−170 (−$520,000,000)
Prince Edward Island	4 ($12,000,000)	−30 (−$94,000,000)	−26 (−$82,000,000)
Newfoundland and Labrador	−9 (−$28,000,000)	−50 (−$150,000,000)	−59 (−$180,000,000)
Yukon	<1	<1	<1
Northwest Territories	<1	<1	<1
Nunavut	<1	<1	<1

a. Counts and valuations represent mean estimates of premature mortality. Counts and valuation estimates are rounded to the nearest integer and given to a maximum of two significant figures. Valuations are in 2018 Canadian dollars and discounted from 2050 to 2019 using a 3% discount rate. Totals may not add up due to rounding.

b. Impact of a changing climate: (2045 climate @ 2002 emissions) – (2002 climate @ 2002 emissions)

c. Impact of reducing air pollutant emissions as per RCP6.0: (2045 climate @ 2002 emissions) – (2045 climate @ 2045 RCP6.0 emissions)

d. Impact of changing climate and reducing air pollutant emissions as per RCP6.0: (2045 climate @ 2045 RCP6.0 emissions) – (2002 climate @ 2002 emissions)

This analysis estimates that the impact of a warming climate in 2050 would result in 850 excess deaths across the country in the summer months, with clear regional differences. Most of the deaths would occur in Ontario (620) and Quebec (270), and some regions would experience small reductions in air pollution and the associated health impacts (e.g., British Columbia). This is consistent with the air quality changes projected by Kelly et al. (2012), which estimate for North America that the largest increases in ozone and $PM_{2.5}$ would occur in industrialized regions such as the Northeastern United States and adjacent regions of Southeastern Canada. Reductions in air pollutant emissions associated with projections under the RCP6.0 scenario would have national population health benefits of about 5200 avoided premature deaths annually in 2050, with benefits in all provinces but particularly in Ontario (2900 avoided premature deaths) and Quebec (1500 avoided premature deaths). These reflect the air pollution health co-benefits of the GHG mitigation strategies in RCP6.0, along with the assumptions in RCP6.0 of more stringent air pollution controls globally over time. The final scenario examined considered the effect of both a warming climate and air pollutant emission reductions. The results show that the benefits of air pollution mitigation are attenuated by the negative effects of a warmer climate. As indicated above, modelled projections of changes in $PM_{2.5}$ concentrations are considered to be more uncertain than those for ozone, which is also true for the health impacts derived from those projections.

Table 5.4 Canadian morbidity and mortality impacts (counts) and economic valuation due to ozone and $PM_{2.5}$ for June to August[a]

HEALTH END POINT	POLLUTANT	IMPACT OF CHANGING CLIMATE IN 2050[b]	IMPACT OF REDUCED AIR POLLUTANT EMISSIONS TO RCP6.0 IN 2050[c]	IMPACT OF CHANGING CLIMATE AND REDUCED AIR POLLUTANT EMISSIONS TO RCP6.0 IN 2050[d]
Mortality				
Mortality	Ozone	610	−3500	−3000
Mortality	$PM_{2.5}$	250	−1700	−1400
Mortality	Ozone + $PM_{2.5}$	850	−5200	−4400
Morbidity				
Acute respiratory symptom days	Ozone, $PM_{2.5}$	1,900,000	−12,000,000	−9,600,000
Adult chronic bronchitis cases	$PM_{2.5}$	160	−1100	−900
Asthma symptom days	Ozone, $PM_{2.5}$	160,000	−920,000	−770,000

HEALTH END POINT	POLLUTANT	IMPACT OF CHANGING CLIMATE IN 2050[b]	IMPACT OF REDUCED AIR POLLUTANT EMISSIONS TO RCP6.0 IN 2050[c]	IMPACT OF CHANGING CLIMATE AND REDUCED AIR POLLUTANT EMISSIONS TO RCP6.0 IN 2050[d]
Child acute bronchitis episodes	$PM_{2.5}$	670	−4400	−3700
Respiratory and cardiac emergency room visits	Ozone, $PM_{2.5}$	970	−6000	−5000
Respiratory and cardiac hospital admissions	Ozone, $PM_{2.5}$	210	−1300	−1100
Restricted activity days	$PM_{2.5}$	210,000	−1,500,000	−1,200,000
Minor restricted activity days	Ozone	340,000	−2,000,000	−1,600,000
Valuation (mean [2.5th percentile, 97.5th percentile])				
All premature deaths	Ozone, $PM_{2.5}$	$2.7 billion [$0.92 billion, $5.1 billion]	−$16 billion [−$5.7 billion, −$31 billion]	−$14 billion [−$4.8 billion, −$26 billion]
All end points	Ozone, $PM_{2.5}$	$2.7 billion [$0.97 billion, $5.2 billion]	−$17 billion [−$6.0 billion, −$32 billion]	−$14 billion [−$5.0 billion, −$27 billion]

a. Counts represent mean estimates of health outcomes. Counts and valuation estimates are rounded to the nearest integer and given to a maximum of two significant figures. Valuations are in 2018 Canadian dollars and discounted from 2050 to 2019 using a 3% discount rate. Totals may not add up due to rounding.

b. Impact of a changing climate: (2045 climate @ 2002 emissions) – (2002 climate @ 2002 emissions)

c. Impact of reducing air pollutant emissions to RCP6.0: (2045 climate @ 2002 emissions) – (2045 climate @ 2045 RCP6.0 emissions)

d. Impact of changing climate and reducing air pollutant emissions to RCP6.0: (2045 climate @ 2045 RCP6.0 emissions) – (2002 climate @ 2002 emissions)

Exposure to ozone and $PM_{2.5}$ also increases the risk of multiple non-fatal outcomes, such as asthma symptoms and cardiorespiratory hospital admissions. These represent important aspects of population health. In this analysis, the effect of the climate penalty on worsening air quality is estimated to result in 1.9 million acute respiratory symptom days and tens of thousands of asthma symptom days and restricted activity days, contributing to a substantial public health burden. This burden is in contrast to the larger annual health co-benefits that would result from air pollution reductions associated with the RCP6.0 scenario, including thousands of avoided cardiorespiratory hospital visits, as well as respiratory illnesses. About 70% of the mortality impacts are attributable to estimated changes in ambient ozone concentrations, with the remainder attributable to $PM_{2.5}$.

The economic value of the damage to health due to air pollution in Canada associated with a warming climate, based on analysis of the summer period in 2050, is estimated to be $2.7 billion, with a range of $0.97 billion to $5.2 billion, representing the 2.5th to the 97.5th percentile. This primarily reflects the value associated with increased risk of premature death. Importantly, this analysis also shows the potentially large socio-economic co-benefits that could be realized from air quality improvements delineated in the RCP6.0 projection, which were estimated to be $17 billion, with a range of $6.0 billion to $32 billion, representing the 2.5th to 97.5th percentile, in 2050.

5.5.4 Conclusion

Overall, the scientific evidence shows that climate change will affect atmospheric processes and, consequently, air quality. Multiple studies estimate that hundreds to thousands of ozone-related deaths may result annually in the United States during this century as a result of a warming climate. Fewer studies have examined $PM_{2.5}$ health impacts, but those that have suggest that $PM_{2.5}$ health impacts will be about one order of magnitude larger than those from ozone (Tagaris et al., 2009; Fang et al., 2013; Silva et al., 2017). Only one study was identified that included an analysis of health impacts in Canada. This focused only on ozone health impacts and estimated that the climate penalty under SRES A1B would result in 45 excess annual deaths in Canada in 2050 (Selin et al., 2009). In a new analysis for Canada conducted for this national assessment report, the net summer season health impacts in 2050 due to the climate penalty are estimated to be 850 excess deaths due to air pollution (total value of impacts $2.7 billion), with approximately 70% of those due to ozone and the remainder to $PM_{2.5}$, and with the vast majority of health impacts occurring in Ontario and Quebec. Note that this analysis did not include the air quality and health impacts resulting from wildfire smoke, which is discussed later in this chapter.

Strategies to mitigate GHG emissions can result in significant air pollution health co-benefits, the societal value of which can, in turn, offset the costs of GHG mitigation measures. Again, studies in the scientific literature generally estimate the $PM_{2.5}$ co-benefits to be about an order of magnitude larger than those from ozone reductions. The new Canadian analysis, presented above, estimates net benefits of following the RCP6.0 pathway to be about 5200 avoided deaths for the summer season for a single year in 2050, mainly in Ontario and Quebec, with a total value of benefits of $17 billion. Air pollution co-benefits are anticipated for each year following GHG emission reductions, and the cumulative benefits over decades would be expected to be much larger.

Air pollution health co-benefits of climate change mitigation represent important potential near-term gains and are realized locally, where mitigation measures are implemented. The incorporation of health co-benefits into a climate change framework can provide justification for more stringent or accelerated climate change mitigation measures or incentives to implement actions on climate change (Shindell et al., 2018). Appropriate analyses could allow for the strategic selection of climate mitigation pathways in Canada that also optimally target reductions in population health impacts due to air pollution, given the health and economic burden these represent to society. It would also allow for identification of potential unintended adverse health consequences.

5.5.5 Key Uncertainties

The modelling of air quality in future scenarios under the effects of climate change represents a challenge, due to inherent uncertainties with respect to global atmospheric dynamics and chemistry, downscaling from global to regional scales, chemical–climate interactions, emissions projections, and meteorological conditions (Tagaris et al., 2009; Fang et al., 2013; Silva et al., 2016; Zhang et al., 2017). Most studies relevant to Canada have focused only on ozone, while fewer analyses have captured $PM_{2.5}$ impacts. Modelled population health impacts are likely underestimated, as not all health outcomes that have been associated with air pollution exposure (e.g., neurological outcomes) can currently be quantified. Variability in the CRFs and monetization of health outcomes used in different studies may influence results (West et al., 2013; Thompson et al., 2014; Silva et al., 2017). Furthermore, the studies undertaken to date have largely used different approaches, assumptions, and modelling methodologies to investigate different questions, limiting the ability to synthesize the information.

5.6 Climate Change and Air Pollution from Wildfires

5.6.1 Wildfires in Canada in a Changing Climate

On average, 7000 wildfires burn about 2.5 million hectares — about half the size of Nova Scotia — every year in Canada. The area burned by wildfire in Canada has doubled since the early 1970s, and this has been attributed to human-caused climate change (Gillett et al., 2004). For example, human influences on the climate were found to be an important contributor to the severity of the 2017 wildfire season in Canada (Kirchmeier-Young et al., 2019). The increase in area burned results in higher air pollution emissions from wildfires, as well as greater associated health risks to humans (Reisen et al., 2015; Matz et al., 2020). The four factors that influence wildfire emissions include area burned, fuel consumed, combustion completeness (efficiency), and emission factor, which is the amount of pollutant released, measured in grams per kilogram.

The Canadian climate is warming, and this is having profound and immediate impacts on fire activity in Canada. The reasons for increasing wildfire activity due to rising temperatures are threefold. First, warmer temperatures extend the fire season, and a longer fire season has already been observed in parts of Canada. For example, interior British Columbia, Alberta, and Northern Ontario have longer fire seasons today than in 1959 to 2000 (Albert-Green et al., 2013; Hanes et al., 2019). Second, warmer conditions result in increased lightning, and, all things being equal, more lightning will result in more fire (Romps et al., 2014). Third, warmer temperatures lead to drier fuels, unless there is a significant increase in precipitation (Flannigan et al., 2016). Almost all future climate change scenarios for Canada do not have sufficient increases in precipitation to compensate for the drying effect from warmer temperatures. Drier fuels make it is easier for fire to start, spread, and burn more intensely, which, in turn, makes the fires more difficult to control or to extinguish and results in greater air pollution emissions.

Wildfire activity varies temporally and spatially. For the period 1959 to 2015, the number of large fires and area burned by wildfire increased in Western Canada (Hanes et al., 2019). This trend is expected to continue, with the largest increase in wildfires and smoke projected for Western Canada through to 2050. Increasing fire activity is anticipated across all of Canada in the last half of this century (Flannigan et al., 2005; Flannigan et al., 2009).

Scientists are investigating whether recent increases in wildfires in Canada are a direct result of climate change. The observed increases in large wildfires are consistent with what is anticipated with climate change (Flannigan et al., 2009; Hanes et al., 2019). Recent research suggests that extreme fire risk in Western Canada during the last decade increased by 1.5 to 6 times due to human-caused climate change (Kirchmeier-Young et al., 2017). Tan et al. (2019) suggested that extreme spring fire weather in Western Canada in 2016 was very likely an outcome of human-caused climate warming, which increased the occurrence of a weather pattern associated with warmer and drier conditions. Kirchmeier-Young et al. (2019) suggested that human-caused climate change increases the area burned by a factor of seven to 11 times during extreme fire seasons, such as the 2017 fire season in British Columbia.

Amiro et al. (2009) suggested a doubling of wildfire emissions in Canada by the end of this century using the Canadian Global Circulation Model. The increases were largely due to increases in area burned rather than increases in the fuel burned per unit of area (known as depth of burn). Recent research, using three global circulation models (HadGEM2, CanESM2, and CSIRO-MK3.6.0) and three RCP scenarios (2.6, 4.5, and 8.5), however, suggested that the proportion of days in the fire seasons with the potential for significant fuel consumption (depth of burn) by wildfire will increase across Canada's forests, more than doubling for British Columbia and the rest of the boreal forest by 2100 (Wotton et al., 2017). The doubling of fuel consumption due to depth of burn only may occur as early as the 2030s in British Columbia. Wotton et al. (2017) suggested that the proportion of days with high-intensity wildfires that are difficult to impossible to extinguish will increase by two to three times for British Columbia and the boreal forest by 2100.

Smoke from wildfires can travel long distances and significantly affect communities that can be 1000 km or more from the fire. As a result, large population centres are not immune to the adverse health impacts from wildfire smoke, although they may be in an area with low wildfire risk (Matz et al., 2020). In addition, Indigenous communities, which may be in close proximity to forests, may be more affected by wildfires and the need for and effects of evacuation (see Chapter 2: Climate Change and Indigenous Peoples' Health in Canada).

Given projected increases in wildfire activity, the potential for interactions between society and fire in Canada (such as more evacuations of communities) will increase. Overall, Canadians need to be prepared for a future with more wildfires and increased levels of wildfire air pollution due to the changing climate.

5.6.2 Health Effects of Air Pollution from Wildfires

Wildfire emissions contain many different air pollutants, including PM, CO, NO_X, methane, polycyclic aromatic hydrocarbons, and VOCs, and contribute to the formation of ozone and secondary PM (Naeher et al., 2007). The health effects of some of these air pollutants are discussed in section 5.3 Health Effects of Outdoor Air Pollution. For each wildfire, the exact composition of the smoke is highly variable and determined by many factors, including type of vegetation burning (e.g., wet or green vegetation versus dead or dry vegetation), type of combustion (e.g., flaming versus smoldering conditions), and weather conditions (Adetona et al., 2016; Black et al., 2017).

The health effects of wildfire smoke are an active area of research and have been well reviewed, as they are an important global issue (Benmarhnia et al., 2013; Youssouf et al., 2014; Liu et al., 2015; Adetona et al., 2016; Reid et al., 2016; Black et al., 2017; Cascio, 2018). These reviews considered studies conducted around the world, including in North America, South America, Europe, Australia, and Asia. The methods used to assess exposure to wildfire smoke varied across the individual studies and included use of land-based PM monitors, satellite imagery, air quality modelling, comparison of fire versus non-fire periods, and self-reports. Based on these reviews, epidemiological studies have identified that exposure to wildfire smoke is associated with an increase in all-cause mortality; however, more studies are required to identify which specific causes of mortality are most affected. In addition, the literature indicates a strong association between exposure to wildfire smoke and respiratory morbidity, specifically exacerbations of asthma and COPD, and increased respiratory infections. Numerous studies have reported significant increases in health care utilization, including hospital admissions, emergency room visits, physician visits, and/or medication use, for these respiratory conditions associated with an increase in wildfire smoke. For cardiovascular morbidity, including health outcomes such as myocardial infarction, stroke, heart failure, and heart rhythm disturbances, the association with wildfire smoke remains inconclusive, due to a small number of studies indicating an effect and many studies reporting null findings. Many studies reported no association with cardiovascular diseases as a group, and the results are inconsistent among the studies evaluating specific outcomes. A small number of studies have evaluated other health effects, including birth outcomes, mental health, and diabetes, although further research is required to assess the impact of wildfire smoke on these and other health effects.

Studies conducted in Canada on the health effects of wildfire smoke exposure support the conclusions of the reviews discussed above. Specifically, these studies have identified increased asthma-related physician visits (Henderson et al., 2011; McLean et al., 2015; Dodd et al., 2018a), asthma medication dispensed (Henderson et al., 2011; Elliot et al., 2013; McLean et al., 2015), and respiratory-related physician visits and hospital admissions (Henderson et al., 2011; Dodd et al., 2018a) associated with exposure to wildfire smoke. However, no associations were observed for cardiovascular-related physician visits or hospital admissions (Henderson et al., 2011; Dodd et al., 2018a).

How wildfire smoke causes health effects is not fully understood, although the evidence suggests the mechanisms may be similar to those identified for ambient PM. Studies in humans and animals exposed to wildfire or wood smoke have suggested that the health effects result from increased oxidative stress and inflammatory responses, as well as possible interaction of particulates with the autonomic nervous system and a possible reduction in immune responses (Adetona et al., 2016; Reid et al., 2016; Black et al., 2017; Cascio, 2018).

5.6.3 The Health Burden of Wildfire Smoke in Recent Years

Health impact analyses have estimated the health burden of wildfires attributable to the increase in air pollutant concentrations from wildfire smoke. On a global scale for 1997 to 2006, the average annual mortality attributable to $PM_{2.5}$ from landscape fire smoke was estimated at 339,000 deaths (interquartile range 260,000 to 600,000) (Johnston et al., 2012). For this study, landscape fires included forest, grass, and peat fires, and were associated with an estimated annual wildfire-$PM_{2.5}$ exposure concentration of 0 to 45 µg/m^3 and a population-weighted average concentration of 2.1 µg/m^3. The greatest impacts were noted for sub-Saharan Africa (157,000 premature deaths) and Southeast Asia (110,000 premature deaths). The importance of climatic variability on wildfire activities was also evident from the sensitivity analysis of strong La Niña and El Niño years. During the El Niño period, associated with dry conditions and greater fire activity, the estimated annual global mortality attributable to $PM_{2.5}$ from landscape fire smoke was estimated at 532,000, compared to an estimate of 262,000 for the La Niña period.

In a national assessment for the continental United States, wildfire episodes in 2008 to 2012 were associated with an annual population-weighted mean wildfire-$PM_{2.5}$ exposure of 0.6 to 1.1 µg/m^3, depending on the wildfire activity of a given year (Fann et al., 2018). These annual wildfire-$PM_{2.5}$ exposures were associated with an estimated 1500 to 2500 premature deaths from short-term exposure; 8700 to 32,000 premature deaths from long-term exposure; 3900 to 8500 respiratory hospital admissions; and 1700 to 2800 cardiovascular hospital admissions. The economic valuation over the five-year period was 63 billion USD (in 2010 USD) for the short-term premature deaths and combined hospital admissions. The long-term premature deaths had a valuation of 450 billion USD (in 2010 USD) across the five-year period. There was considerable regional variation in wildfire smoke exposure and attributable health impacts, with greater wildfire activity occurring in the western and southeastern states.

There is only limited information quantifying the health and associated monetary impacts from wildfire-related air pollution in Canada in the published literature. A 2001 fire in Chisholm, Alberta, was used as a case study to estimate the acute health impacts associated with short-term increases in $PM_{2.5}$ (Rittmaster et al., 2006; Rittmaster et al., 2008). This seven-day fire burned approximately 116,000 ha and had significant impacts on the air quality of Edmonton (160 km south of Chisholm), Red Deer (125 km south of Edmonton), and the surrounding area. Health damage associated with $PM_{2.5}$ concentrations above the Canada Wide Standard of 30 µg/m^3, which was applicable at the time, were estimated to be $2 million to $3 million (in 1996 CAD).

5.6.4 Quantifying Recent Canadian Air Pollution Health Impacts from Wildfire Smoke

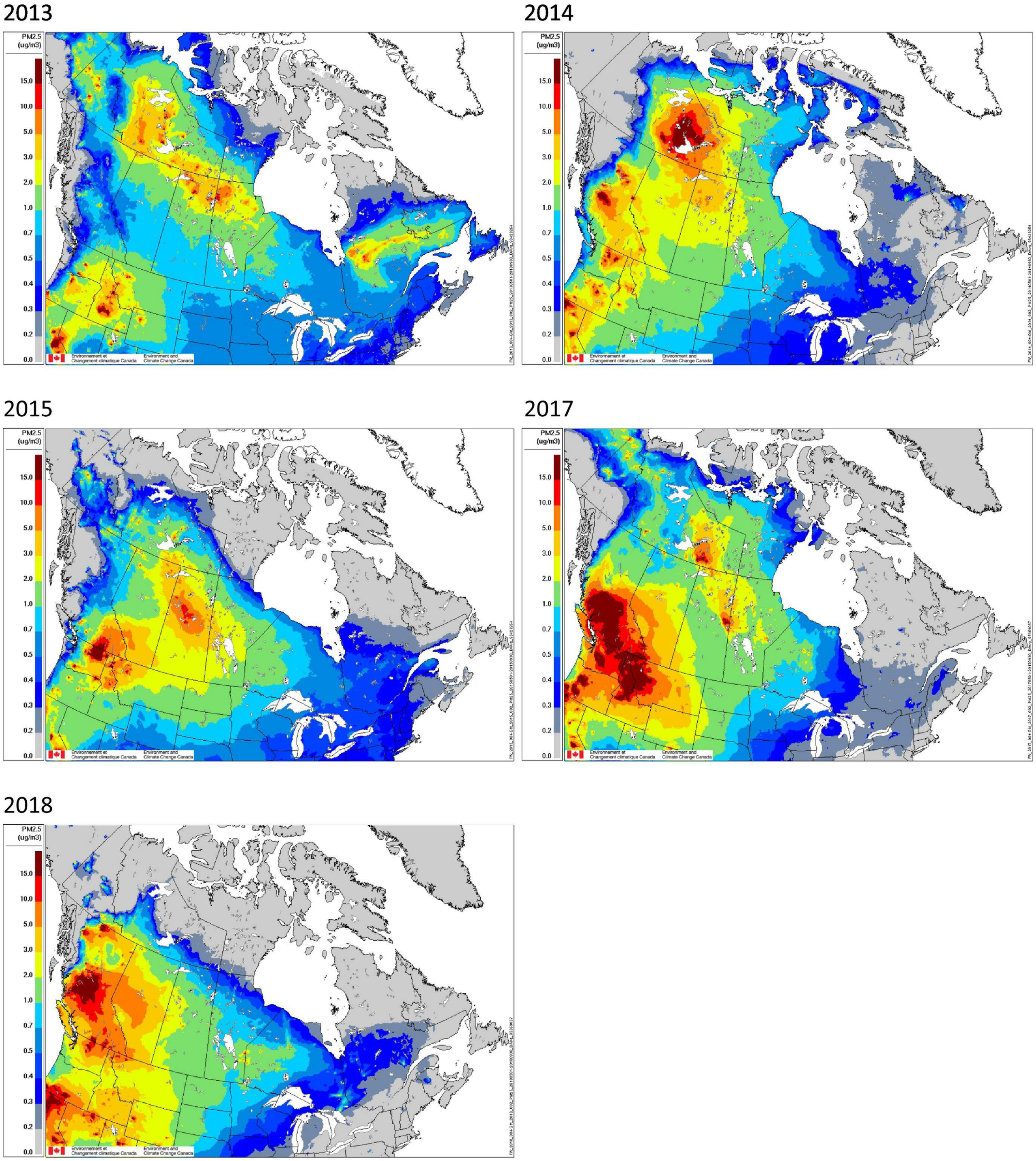
As discussed in section 5.6.1 Wildfires in Canada in a Changing Climate, wildfire activity in Canada is expected to increase under a warming climate. Currently, there is too much uncertainty to predict with sufficient accuracy where in Canada wildfires will occur by mid-century and hence to estimate the population health impacts of wildfires under potential future climates. However, it is informative to assess the current air pollution health impacts from wildfires to better understand the magnitude of this population health issue. Health Canada and Environment and Climate Change Canada undertook an analysis of the Canadian air quality and human health impacts of air pollution from wildfires in recent years, the results of which are presented in this section (Matz et al., 2020).

Although intermittent in nature, wildfire smoke is recognized as a major contributor to air quality issues in North America. To provide forecast guidance for air quality alerts that could reduce air pollution exposure and protect human health during a wildfire smoke event, Environment and Climate Change Canada developed FireWork, a comprehensive operational air quality forecast system based on near-real-time biomass-burning emissions data. This system delivers operational forecasts of modelled air pollutant concentrations from biomass burning, in particular $PM_{2.5}$, on a daily basis over North America (Pavlovic et al., 2016).

A multi-year retrospective analysis of FireWork wildfire-$PM_{2.5}$ forecasts was conducted to estimate Canadian population exposure to $PM_{2.5}$ from wildfires (Munoz-Alpizar et al., 2017). Emissions from wildfires across North America were included in the modelling, which was limited to a five-month period from May to September, for calendar years 2013 to 2018. Due to substantial changes to the modelling grid used for 2016, there is a high level of uncertainty in the model output. As such, the results for that year were excluded from further analysis.

Comparisons of average monthly forecasted surface $PM_{2.5}$ concentrations due to wildfires for the 2013 to 2018 period showed large year-to-year variations in both the timing and the spatial locations of impacts (see Figure 5.2). Additionally, wildfires can sometimes affect the same location many times during a single season. The frequent presence of wildfire-$PM_{2.5}$, especially in Western North America, has implications for regional achievement of $PM_{2.5}$ air quality standards. The percent of Canadian landmass with wildfire-$PM_{2.5}$ and the percent of the Canadian population affected by wildfire-$PM_{2.5}$ above specified concentrations are provided in Figure 5.3. For 2013 to 2017, more than 60% of the landmass of Canada had average (May to September) wildfire-$PM_{2.5}$ concentrations of 0.2 $\mu g/m^3$ or more, affecting more than 90% of the population and demonstrating the widespread nature of wildfire smoke. Additionally, for 2013 to 2018, approximately 25% to 40% of the land mass of Canada had average wildfire-$PM_{2.5}$ concentrations of 1 $\mu g/m^3$ or more, affecting approximately 20% to 30% of the population. At the higher-threshold concentration levels, the percentages of landmass and population affected were further reduced. Since the population of Canada is not evenly distributed over the large landmass, the proximity of fire activity to population centres is a key determinant of the population affected by higher wildfire-$PM_{2.5}$ levels. For example, in 2017, the wildfire activity and smoke plume affected large population centres in British Columbia, and the proportion of the population affected was greater than the proportion of the landmass. Data for the provinces and territories with wildfire-$PM_{2.5}$ concentrations of 1 $\mu g/m^3$ or more, affecting more than 5% of the landmass and more than 5% of the population, for 2013 to 2018, are presented in Figure 5.4. In multiple years between 2013 and 2018, about

60% to 100% of the landmass and over 80% of the population in the four Western provinces and the Northwest Territories experienced average (May to September) wildfire-$PM_{2.5}$ concentrations of at least 1 µg/m^3.



2013

2014

2015

2017

2018

Figure 5.2 Wildfire $PM_{2.5}$ concentrations (May to September) for 2013 to 2018 in Canada. The colours on the maps show a range from grey to deep red indicating a range of wildfire-$PM_{2.5}$ from 0.0 to 15.0 µg/m^3. Source: Matz et al., 2020.

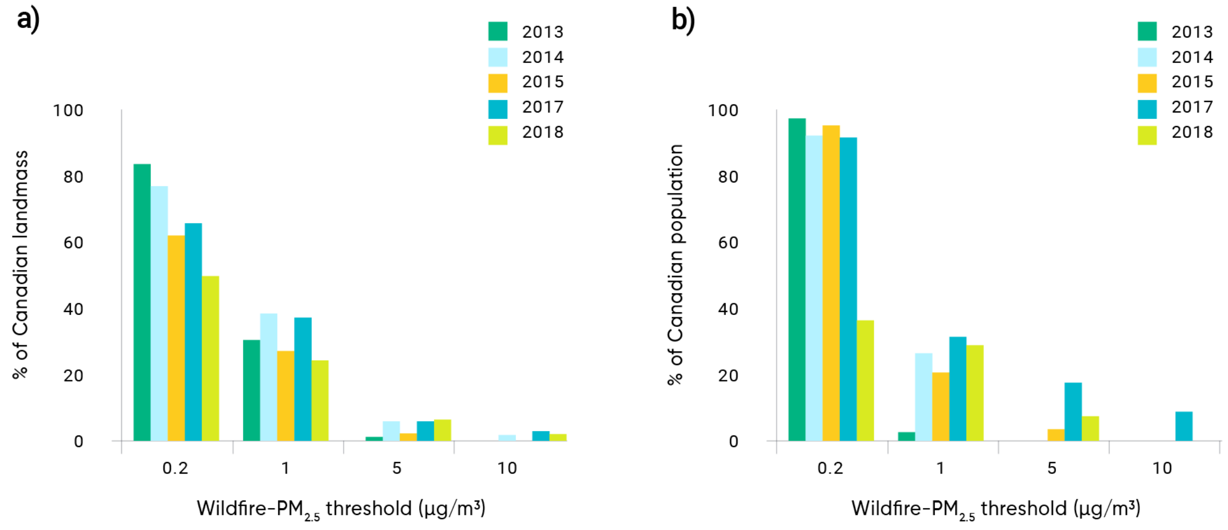


Figure 5.3 Percent of Canadian landmass and of Canadian population with average (May to September) wildfire-$PM_{2.5}$ concentrations above given thresholds. Panel A shows the percent of Canadian landmass with average May to September wildfire-$PM_{2.5}$ concentrations above given thresholds, and panel B shows the percent of population with May to September wildfire-$PM_{2.5}$ concentrations above given thresholds. Source: Matz et al., 2020.

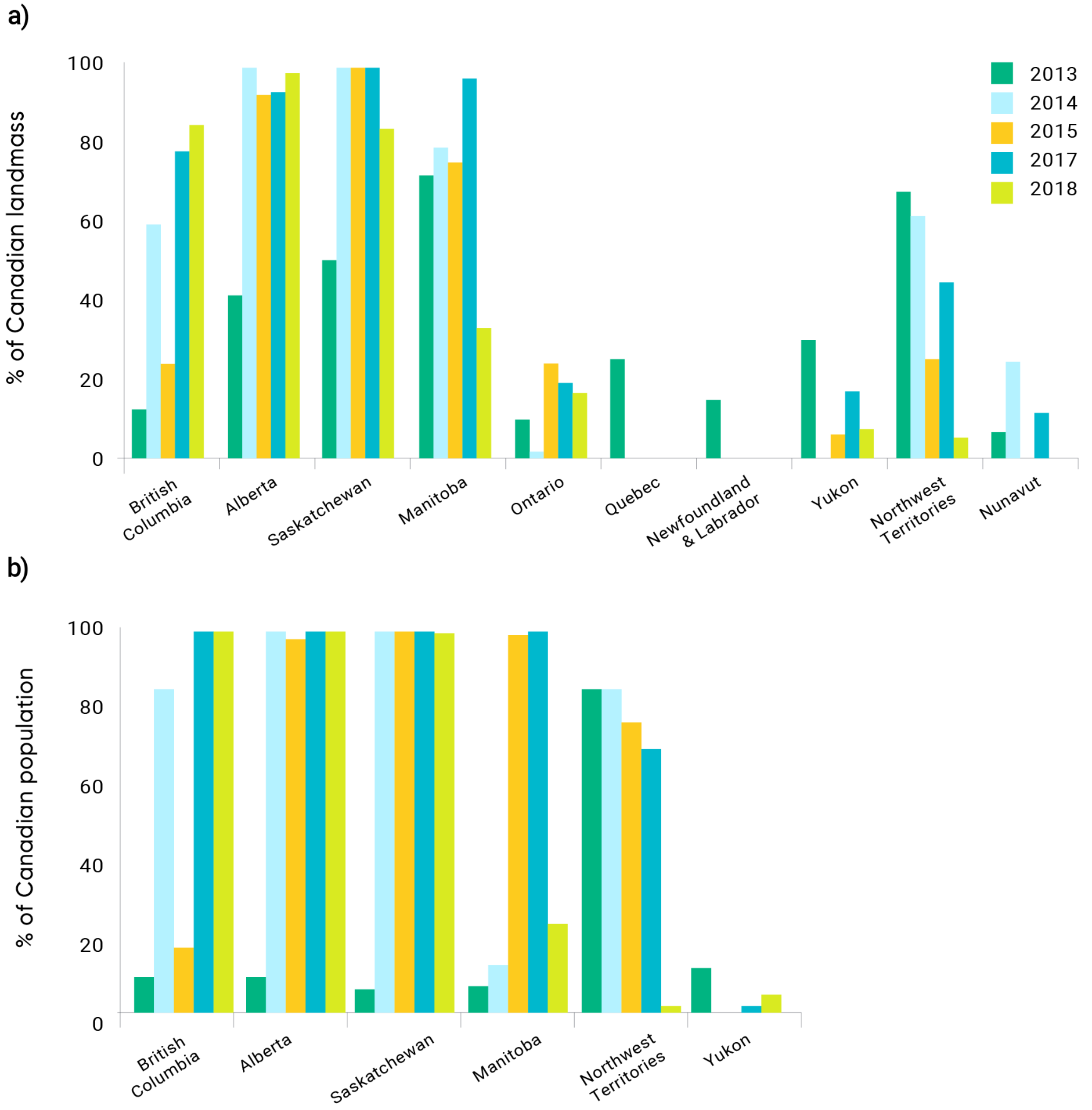


Figure 5.4 Percent of landmass and percent of population by province and territory with average (May to September) wildfire-$PM_{2.5}$ concentrations ≥1 μg/m^3 for 2013 to 2018. Panel A shows the percent of landmass with average wildfire-$PM_{2.5}$ concentrations of 1 μg/m^3 or more for 2013 to 2018 (May to September). Panel B shows the percent of the population exposed to average wildfire-$PM_{2.5}$ concentrations of 1 μg/m^3 or more for 2013 to 2018 (May to September). Only provinces and territories affected at more than 5% (of landmass or population) are included in the panels. Source: Matz et al., 2020.

The Canadian population health impacts attributable to wildfire $PM_{2.5}$ for 2013 to 2015 and 2017 to 2018 were estimated using Health Canada's AQBAT 3.0. The mortality and morbidity results, including the economic valuation, are presented in Table 5.5 and Table 5.6. Nationally, 54 to 240 premature deaths due to short-term exposure and 570 to 2500 premature deaths due to long-term exposure per year were attributable to wildfire-$PM_{2.5}$, as well as many non-fatal cardiorespiratory health outcomes. The most frequent morbidities were days with acute respiratory symptoms and days with restricted activity. The substantial year-to-year variation in wildfire activity is reflected in the health impact analysis, with the greatest impacts estimated for 2017 and much smaller estimates for 2013. This variability is also driven by whether the air pollution plume from the fires disperses over highly populated areas. Over the five calendar years assessed, the economic value of the population health impacts was estimated at $410 million to $1.8 billion per year for acute health impacts and $4.3 to $19 billion per year for chronic health impacts.

Table 5.5 Acute health impacts and economic valuation[a] from wildfire $PM_{2.5}$, for 2013 to 2015 and 2017 to 2018

	2013	2014	2015	2017	2018
Acute mortality	54	70	97	240	131
Acute mortality valuation[b]	**$410 million** [$120 million–$830 million]	**$520 million** [$160 million–$1.1 billion]	**$730 million** [$220 million–$1.5 billion]	**$1.8 billion** [$530 million–$3.7 billion]	**$980 million** [$280 million–$2.0 billion]
Acute respiratory symptom days	1,400,000	1,900,000	2,500,000	6,100,000	3,400,000
Asthma symptom days[c]	100,000	140,000	190,000	420,000	240,000
Child acute bronchitis episodes	2,600	3,400	4,600	10,000	6,000
Respiratory emergency room visits	170	230	310	710	420
Respiratory hospital admissions	34	45	61	140	83

	2013	2014	2015	2017	2018
Cardiac emergency room visits	60	75	110	250	140
Cardiac hospital admissions	46	57	80	190	110
Restricted activity days	750,000	1,000,000	1,400,000	3,200,000	1,800,000
Acute morbidity valuation[b]	**$73 million** [$13 million–$177 million]	**$97 million** [$17 million–$240 million]	**$131 million** [$24 million–$320 million]	**$310 million** [$58 million–$750 million]	**$170 million** [$33 million–$420 million]

a. The dollar values in Table 5.5 are socio-economic values associated with small changes in the risk of various health outcomes. AQBAT provides economic valuation estimates of those health impacts, considering the potential social, economic, and public welfare consequences of the health outcomes, including medical costs, reduced workplace productivity, pain and suffering, and the impacts of increased mortality risk.

b. Values represent mean valuation of multiple iterations; [2.5th–97.5th percentiles].

c. Asthma symptom days are only estimated for children (five to 19 years of age).

Table 5.6 Chronic health impacts and economic valuation[a] from wildfire $PM_{2.5}$, for 2013 to 2015 and 2017 to 2018

	2013	2014	2015	2017	2018
Chronic mortality	570	730	1000	2500	1400
Chronic mortality valuation[b]	**$4.3 billion** [$1.5 billion-$8.2 billion	**$5.5 billion** [$2.0 billion–$11 billion]	**$7.6 billion** [$2.7 billion–$15 billion]	**$19 billion** [$6.7 billion–$35 billion]	**$10 billion** [$3.8 billion–$20 billion]
Adult chronic bronchitis cases	530	710	960	2300	1300
Chronic morbidity valuation[b]	**$230 million** [$0–$620 million]	**$320 million** [$0–$830 million]	**$420 million** [$0–$1.1 billion]	**$1.0 billion** [$0–$2.6 billion]	**$560 million** [$0–$1.5 billion]

a. The dollar values in Table 5.6 are socio-economic values associated with small changes in the risk of various health outcomes. AQBAT provides economic valuation estimates of those health impacts, considering the potential social, economic and public welfare consequences of the health outcomes, including medical costs, reduced workplace productivity, pain and suffering, and the impacts of increased mortality risk.

b. Values represent mean valuation of multiple iterations; [2.5th – 97.5th percentiles].

A breakdown of estimated national premature deaths, by province and territory, is provided in Table 5.7. In 2013, the greatest impacts were estimated for Ontario and Quebec, reflecting the wildfire activity in Northwestern Quebec that year. For the other years (2014 to 2018), the greatest impacts were estimated in the provinces of British Columbia and Alberta, reflecting the substantial wildfire activity in Western Canada and in the United States during these years. During this period, health impacts were also noted for Saskatchewan, Manitoba, Ontario, and Quebec, indicating that the long-range transport of wildfire-$PM_{2.5}$ can affect population health at great distances from the wildfire locations.

Table 5.7 Estimated premature deaths from acute and chronic exposure to wildfire $PM_{2.5}$, by province and territory, for 2013 to 2015 and 2017 to 2018

	2013		2014		2015		2017		2018	
	ACUTE	CHRONIC	ACUTE	CHRONIC	ACUTE	CHRONIC	ACUTE	CHRONIC	ACUTE	CHRONIC
Canada	54	570	70	730	97	1000	240	2500	131	1400
British Columbia	6	59	23	240	25	260	170	1700	69	720
Alberta	7	71	19	200	28	290	42	440	42	430
Saskatchewan	3	30	6	60	9	96	6	62	5	53
Manitoba	3	35	4	37	7	74	4	45	4	40
Ontario	19	200	11	110	17	180	10	110	6	66
Quebec	15	150	7	69	10	100	7	71	5	49
New Brunswick	1	9	0	4	1	6	0	5	0	4
Prince Edward Island	0	1	0	0	0	1	0	1	0	1
Nova Scotia	1	8	0	3	1	5	0	5	0	3
Newfoundland and Labrador	1	6	0	1	0	2	0	1	0	1
Yukon	0	1	0	0	0	0	0	1	0	0

	2013		2014		2015		2017		2018	
	ACUTE	CHRONIC	ACUTE	CHRONIC	ACUTE	CHRONIC	ACUTE	CHRONIC	ACUTE	CHRONIC
Northwest Territories	0	2	1	5	0	1	0	1	0	0
Nunavut	0	0	0	0	0	0	0	0	0	0

5.6.5 Air Pollution Health Impacts of Wildfire Smoke Under Climate Change

In recent studies, projected wildfire activity under climate change scenarios has been used to estimate population exposures to wildfire smoke and health impacts in the future. Mills et al. (2018) projected wildfire smoke exposure across the continental United States under RCP4.5 and RCP8.5 for the years 2050 and 2090. In 2050, a projected 3 million more people would be exposed to wildfire smoke under RCP8.5 compared to RCP4.5 and the difference would increase to 10 million in 2090. Significant regional variation is noted across the country, with the greatest air quality impacts in the northeast and southwest. The study methods took a conservative approach and did not capture long-range transport of wildfire smoke. Ford et al. (2018) also considered the RCP4.5 and RCP8.5 scenarios to model wildfire and biomass burning, and $PM_{2.5}$ emissions for the continental United States. Annual average wildfire $PM_{2.5}$ emissions were estimated to increase due to projected increases in emissions during the peak fire season and lengthening of the fire season, for 2050 and 2100. The largest projected increases in emissions were noted for the Southeastern United States and along the Canadian border. Additionally, due to projected decreases in anthropogenic sources of $PM_{2.5}$ under RCP4.5 and RCP8.5, the relative contributions of fire-$PM_{2.5}$ were projected to increase from approximately 25% in 2000 to approximately 50% in 2050 and 2100. From a modelled baseline of 17,000 premature deaths attributable to fire-$PM_{2.5}$ in 2000, projections estimated increases in deaths to 42,000 (RCP4.5) or 32,000 (RCP8.5) by 2050, and 32,000 (RCP4.5) or 44,000 (RCP8.5) by 2100.

Liu et al. (2016) used projections of wildfire $PM_{2.5}$, based on the SRES A1B climate scenario to estimate respiratory hospital admissions for seniors across the Western United States. Short-term increases in $PM_{2.5}$ from wildfires were associated with an increase of 178 respiratory hospital admissions for people 65 years or older for 2046 to 2051 compared to 2004 to 2009. The estimates were greatest for the population centres in Southern and Central California, Western Washington, Central Colorado, and Central Utah.

5.6.6 Populations at Higher Risk

For some of the air pollutants associated with wildfire smoke, such as PM, numerous epidemiological studies have identified populations that may be at increased risk (see section 5.3 Health Effects of Outdoor Air Pollution). In comparison, fewer studies have investigated possible populations or conditions that may increase the risk of adverse health effects from exposure to wildfire smoke. Limited evidence suggests that young children, seniors, people with pre-existing conditions such as asthma or COPD, and people with lower socio-economic status may be at increased risk (Liu et al., 2015; Reid et al., 2016). A recent review of North American studies reported evidence of a greater effect of wildfire smoke on women compared to men for health care utilization due to respiratory effects in healthy adults and in those with COPD (Kondo et al., 2019). The review also reported a slightly lower relative risk for health care utilization for respiratory effects in youth compared to adults, but the data were insufficient to assess any effect modification due to income, education, access to care, or other personal characteristics. Further research is needed to better identify the subpopulations at greatest risk to the health effects of wildfire smoke.

Indigenous populations may be more sensitive to health effects from wildfire air pollution (see Chapter 2: Climate Change and Indigenous People's Health in Canada). In Canada, First Nations and Métis people have a higher burden of chronic respiratory diseases such as asthma and COPD (Gershon et al., 2014; Carrière et al., 2017), making them more susceptible to the adverse effects of air pollution overall. A Canadian study suggested that adverse respiratory outcomes, including emergency room visits and clinic visits for cough, asthma, and pneumonia, increased during the prolonged 2014 wildfire season in the Northwest Territories compared to the previous two years, although this study was not specific to Indigenous Peoples (Dodd et al., 2018a). In addition, Indigenous populations living in remote locations may be at increased risk of exposure to wildfire smoke due to proximity.

Given their occupation, wildland firefighters are exposed more frequently and to greater levels of wildfire smoke than the general public. Studies of firefighters have indicated acute health effects of wildfire smoke exposure, including reduced lung function, lung inflammation, pulmonary and systemic oxidative stress, and respiratory symptoms (Youssouf et al., 2014; Adetona et al., 2016; Black et al., 2017; Groot et al., 2019). However, the long-term effects of cumulative occupational exposure to wildfire smoke have not been identified.

5.6.7 Conclusion

Scientific studies have identified numerous adverse health effects associated with wildfire smoke, including premature mortality and respiratory health effects. In addition, new research is evaluating possible associations with cardiovascular health effects, as well as with birth outcomes, mental health outcomes, and diabetes. Furthermore, significant population health impacts attributable to wildfire $PM_{2.5}$ have been estimated for Canada, the United States, and globally. Specifically, for the Canadian population over the period from 2013 to 2018, 620 to 2700 deaths per year were attributed to $PM_{2.5}$ from wildfires, along with many non-fatal adverse health outcomes. Climate change is anticipated to increase the number and severity of wildfires in Canada and globally, due to greater fire activity and lengthening of the fire season. The increased emissions from wildfires will result in a greater public health burden from air pollution and will require expanded adaptation efforts by public health agencies and other government organizations.

5.6.8 Key Uncertainties

Although increasing fire activity is expected in Canada over this century, it is difficult to estimate population health impacts under future climate change scenarios, as population exposure will depend on the location and size of individual fires, as well as meteorological conditions, all of which are hard to predict with the required spatial resolution. Air quality modelling includes inherent uncertainties, given the complexity of atmospheric processes. In addition, modelled population health impacts are likely underestimated, as not all health outcomes that have been associated with air pollution exposure can be quantified and included in the analysis. In addition, epidemiological studies of the health effects of wildfire air pollution specifically are still limited, and effects on health may differ from those caused by exposure to ambient air pollution. There is global recognition of the increasing importance of wildfire smoke as a source of air pollution exposure, and investigation of the health effects attributable to wildfire smoke is an active area of research. This may result in the development of source-specific CRFs for use in health impact assessments.

5.7 Adaptation and Risk Mitigation for Health Effects of Outdoor Air Pollution

In addition to strategies targeting reductions in air pollution emissions, there are multiple initiatives to help reduce exposure to and, hence, health risks from outdoor air pollution in Canada. These address outdoor air pollution in general, and, more recently, specific actions regarding wildfire smoke. There have been reductions in the number of smog-based advisories in recent years in Canada and the United States as a result of reductions in air pollution emissions, but the frequency of wildfire smoke episodes has increased. Given that climate change can contribute to the deterioration of air quality and increased wildfires, these measures take on added importance in the protection of public health.

5.7.1 Outdoor Air Pollution

Canada developed the Air Quality Health Index (AQHI) to convey the health risks of air pollution to the public on a day-to-day basis and inform decisions to protect health. The AQHI forecast is designed to help Canadians know when to monitor their symptoms, limit their exposure to air pollution, and make other behavioural changes, such as adjusting their exercise activities. The index represents the combined impact of the air pollution mixture and conveys health risks in relative terms. Rather than anchoring the index to threshold values derived from air quality standards, the AQHI calculates risks from epidemiological analysis of the population health impacts associated with short-term (daily) exposure to air pollutants, with a value of 10 based upon the highest-risk day observed during the period 1998 to 2001. The formula incorporates concentrations of NO_2, ozone, and $PM_{2.5}$ according to their contribution to increasing the risk of mortality from non-accidental causes (Stieb et al., 2008). The AQHI is reported on a scale of 1 to 10+, and the higher the number, the higher the health risk (see Figure 5.5). The AQHI is therefore an initiative to reduce the risk of air pollution that promotes adaptation through behavioural change; it has garnered international attention for its efficacy and clarity (Chen et al., 2013b; Oakes et al., 2014; Du et al., 2020).

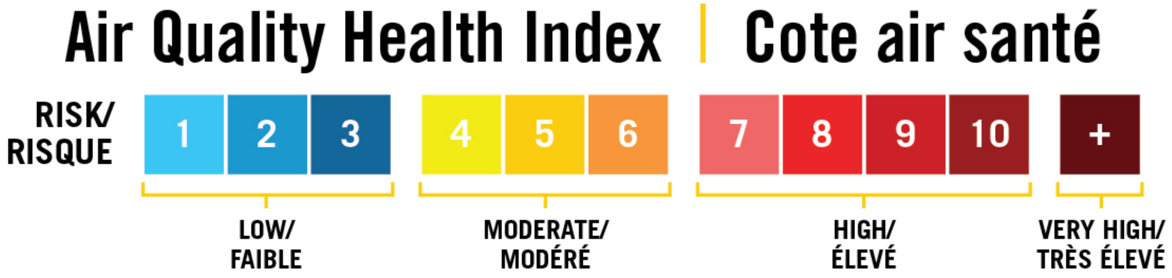


Figure 5.5 Air Quality Health Index scale. Source: Government of Canada, 2019b.

The usefulness of the AQHI as a tool for adaptation depends on the ability to anticipate air quality conditions. Environment and Climate Change Canada forecasts the AQHI across the country (Government of Canada, 2019a), accompanying weather forecasts in all regions except Quebec, which is the only province or territory that has not adopted the AQHI. As part of its Info-Smog program, Quebec has an air quality index that calculates a subindex for each pollutant relative to a provincial air quality standard (Government of Canada, 2019b).

Health messages and health protection advice communicated through the AQHI distinguish between the general population and the populations that may experience increased risk (Table 5.8). People with heart and lung conditions are described as most affected by air pollution, while other populations at higher risk include those with diabetes, young children, seniors, and people who are active outdoors. Advice to reduce exposure is primarily directed at avoiding elevated exposure to air pollution by changing schedules and relocating activities (e.g., indoors or outdoors and away from traffic), while, at the same time, encouraging appropriate exercise activities. Individuals are encouraged to check the AQHI forecast and become familiar with how their health may be affected at different AQHI values (e.g., recognizing when they experience symptoms). Those who may be at increased risk are encouraged to monitor their symptoms and limit outdoor activities at higher AQHI values. In addition to the information accompanying the AQHI forecasts, some provinces, municipalities, and media outlets post AQHI guides and information on webpages and provide adaptation advice to the public. The Air Health Check awareness campaign, which ran from 2015 to 2019, disseminated additional information on factors that increase vulnerability, symptoms, and protective actions (Scout Environmental, 2019).

Bilateral arrangements between federal and local authorities govern local air quality alerts, also called advisories, warnings, and special air quality statements. Air quality alerts may be based upon a forecast AQHI of seven or above (high risk), or when a specific pollutant exceeds a concentration chosen by the province. The health protection messages for advisories are generally consistent with the AQHI messaging for high or very high risk. However, they may be more detailed, including advice to avoid traffic, stay inside where there is central air conditioning, have an adequate supply of medication on hand, and reduce generation of air pollutants indoors, along with messages advising personal actions to reduce pollution, such as limiting vehicle use and outdoor combustion (OMECP, 2010). The Quebec Info-Smog program issues advisories when a pollutant reaches, or is forecast to reach, the "poor" category and makes use of similar risk reduction messages (Santé Montréal, 2017).

Table 5.8 AQHI health messages

HEALTH RISK	AIR QUALITY HEALTH INDEX	HEALTH MESSAGES	
		AT-RISK POPULATION*	GENERAL POPULATION
Low	1–3	Enjoy your usual outdoor activities.	Ideal air quality for outdoor activities.
Moderate	4–6	Consider reducing or rescheduling strenuous activities outdoors if you are experiencing symptoms.	No need to modify your usual outdoor activities unless you experience symptoms such as coughing and throat irritation.
High	7–10	Reduce or reschedule strenuous activities outdoors. Children and the elderly should also take it easy.	Consider reducing or rescheduling strenuous activities outdoors if you experience symptoms such as coughing and throat irritation.
Very high	Above 10	Avoid strenuous activities outdoors. Children and the elderly should also avoid outdoor physical exertion.	Reduce or reschedule strenuous activities outdoors, especially if you experience symptoms such as coughing and throat irritation.

* People with heart or breathing problems are at greater risk. Follow your doctor's usual advice about exercising and managing your condition.

Source: Government of Canada, 2015

The AQHI does not measure the effects of odour, pollen, dust, heat, or humidity on human health. Additive health effects have been observed between heat and air pollution, leading to increases in mortality and hospital admissions for cardiovascular and respiratory diseases (European Academies' Science Advisory Council, 2019). However, seasonal and regional variations in heat responses, as well as other factors, have thus far precluded consideration of a combined index for heat and air quality or the incorporation of a heat indicator into the AQHI. Only minimal messaging on the combined risk of air pollution and heat is included in AQHI factsheets. When air quality and heat warnings are issued simultaneously, messages acknowledge the combined risks of the two hazards; however, integrated adaptation messaging has not been established. Some local jurisdictions have been using ad hoc combinations of content from Health Canada's work on extreme heat and air pollution (Anderson, 2016), and there is a need to develop joint messaging.

There is no direct experimental evidence to demonstrate the effectiveness of air quality forecast programs such as the AQHI in reducing population health risks. However, many studies show that health risks increase with increasing exposure to air pollution and, therefore, actions that effectively reduce exposure will reduce risk (Abelsohn & Stieb, 2011). Several studies have investigated the effectiveness of air quality advisory programs in altering behaviour (Wen et al., 2009; Spurr et al., 2014; Radisic et al., 2016). They found that individuals who were more vulnerable to air pollution (e.g., asthma sufferers) and those with more knowledge of air pollution were more likely to respond to the information provided. However, few studies have been able to address the question of whether incidence of illness decreases. A study of the impact of phone messaging of alerts to sensitive patients, with follow-up interviews on actions and symptoms, did not detect an effect on health (Mehiriz & Gosselin, 2019). Another study investigated the impact of air quality advisories in Toronto and found that the announcement of alerts was associated with a reduction of emergency department visits for asthma by 25% but could not detect an effect for any other health outcome (Chen et al., 2018).

Enhancing urban green spaces can play a role as an adaptive measure to deal with climate change, while also providing potential health and social co-benefits. There has been considerable research indicating a positive association between exposure to green space and improved health outcomes, such as hypertension and cardiovascular end points (Twohig-Bennett & Jones, 2018). More recently, researchers are attempting to elucidate how green space and air pollution, two key elements of the urban environment, interact to affect health. There is emerging evidence that increased green space may attenuate the effect of air pollution (Crouse et al., 2019). For example, urban green space can provide an environment away from higher-pollution microenvironments (such as near roadways), where people can go, including to exercise. Green infrastructure, such as vegetation barriers along busy roadways, can reduce the transfer of air pollutants to nearby environments (Baldauf, 2016). Green space can also provide a cooling effect to help reduce the urban health island effect (see Chapter 3: Natural Hazards) and function as a carbon sink.

5.7.2 Wildfire Smoke

Wildfire smoke has become a frequent summertime feature of air quality conditions over large areas in Canada, particularly in Western Canada, with resulting attention to public health impacts and adaptation measures. Models suggest that wildfires will continue to increase in both severity and frequency due to climate change (see section 5.6 Climate Change and Air Pollution from Wildfires). Adaptation measures in response to wildfire smoke are primarily related to warning individuals and providing them with the means to reduce their exposure, particularly those who may be at increased risk of related health impacts. A qualitative study of the health impacts of the 2014 wildfires in the Northwest Territories observed a decrease in mental and emotional health among the majority of interviewees and recommended comprehensive planning and education to reduce risks in Indigenous and other communities (Dodd et al., 2018b).

Recent years have seen advances in air quality modelling to incorporate wildfire smoke into air quality forecasts. Environment and Climate Change Canada developed its FireWork model as part of its operational air quality forecast system and produces twice-daily forecasts of wildfire $PM_{2.5}$ for the next 48 hours. The model output is incorporated into air quality forecasts. Maps and animations of predicted smoke paths are produced, and air quality alerts are issued when necessary (Government of Canada, 2019c). The relative cohesion of a smoke plume over large distances means that wildfire episodes present particular challenges to forecasting and to the adoption of appropriate adaptive measures.

Currently, the AQHI formula is being evaluated for effectiveness in the context of wildfires. Concerns have arisen that, during smoke events, the AQHI readings do not correspond to the sensory experiences of the public in the area. An analysis was conducted with data from British Columbia, and an AQHI+ version of the index, using a formula based on one-hour $PM_{2.5}$ only, was determined to have a better fit to asthma outcomes and respiratory physician visits, although not to mortality and cardiovascular outcomes. The new version is now used in British Columbia throughout the year, in conjunction with the AQHI (Yao et al., 2019). The Northwest Territories has developed a self-assessment guide for wildfire smoke and health, based on visibility (NT HSS, 2016a).

Extreme ambient $PM_{2.5}$ concentrations can occur, and adaptive measures that go beyond the risk reduction advice associated with the AQHI may be required. The British Columbia Centre for Disease Control (BC CDC) carried out an analysis of public health responses to wildfire events and identified 13 priorities for action. While all are relevant to adaptation to smoke events, three are of particular interest: developing guidance for clean air shelters that can be used during smoke events; engaging public health practitioners in wildfire emergency response; and raising public awareness of protective actions (Maguet, 2018). Evidence reviews of air filtration in institutions were conducted (Keefe, 2014), the use of clean air shelters was explored (Barn, 2014), and advice on outdoor activities and the use of protective masks was developed (BC CDC, 2014). A series of public information factsheets were developed in 2019 (BC CDC, 2019). The Manitoba Office of Disaster Management has developed guidelines for protecting community health and well-being from wildfires (Manitoba Health, 2019), as have the Northwest Territories (NT HSS, 2016b). The federal government, as well as several provinces and territories, have reports, web pages, and factsheets that advise the public on matters relating to health protection and risk reduction (OHLTC, n.d.; Saskatchewan Environment Public Health and Safety, n.d.; Nova Scotia Department of Health and Wellness, 2018; NT HSS, 2018; Alberta Health Services, 2019; Government of Canada, 2019d; Ministère de l'Environnement et Lutte contre les changements climatique, 2019; Yukon Health and Social Services, 2019; Health Canada, 2020).

5.8 Impacts of Climate Change on Indoor Air Quality and Health

Canadians spend approximately 90% of their time indoors (Leech et al., 2002; Matz et al., 2014). Exposure to poor indoor air quality has been shown to cause or exacerbate a wide range of health effects, such as asthma, allergies, COPD, and other respiratory diseases, and certain commonly measured indoor air pollutants are recognized carcinogens (Zhang & Smith, 2003; Dales et al., 2008a; Hulin et al., 2012). Climate change, along with efforts to mitigate GHG emissions, can affect indoor air quality in a number of ways that have the potential to significantly affect human health in Canada.

5.8.1 Building Airtightness

Building airtightness (also called envelope airtightness), defined as the resistance to inward or outward air leakage through unintentional leakage points or areas in the building envelope, is an important consideration for indoor air quality. Nearly two-thirds of energy use in residential and commercial buildings in Canada is for heating and cooling (NRCan, 2018). Since the early 1980s, the energy efficiency of Canadian homes has dramatically improved as a result of updated building codes as well as energy efficiency certification programs, including R-2000 and Energy Star (Hamlin & Gusdorf, 1997; Parekh et al., 2007). However, an increase in airtightness often comes at the expense of natural ventilation. Furthermore, the increased use of air conditioning (NRCan, 2016), which may continue to grow due to warm temperatures associated with climate change, can result in reduced natural ventilation, which can increase the accumulation of indoor-generated air pollutants such as VOCs and $PM_{2.5}$. Increased airtightness can also trap moisture inside homes, resulting in mould and the proliferation of dust mites (Bone et al., 2010). Heat-recovery ventilators offer an energy efficient means of providing adequate ventilation in homes (CMHC, 1998; Health Canada, 2018), and their use has been associated with improved health, for example, reductions in respiratory disorder symptoms, among occupants, according to Canadian and international studies (Leech et al., 2004; Kovesi et al., 2009; Maidment et al., 2014). As efforts continue in Canada to increase residential energy efficiency in order to support GHG mitigation, it is important to ensure that homes remain sufficiently ventilated to avoid poor indoor air quality and prevent adverse health outcomes associated with inadequate ventilation (Hernberg et al., 2014; Sharpe et al., 2015). Inadequate ventilation could also have implications for the transmission of infectious respiratory diseases, such as tuberculosis, which is a particular concern for First Nations (on- and off-reserve) and Inuit communities (Beggs et al., 2003).

5.8.2 Impact of Changing Ambient Conditions on the Indoor Environment

Changes to the outdoor environment due to climate change can alter air quality in the indoor environment. For example, higher outdoor ozone concentrations can result in higher concentrations of indoor pollutants, such as formaldehyde, acrolein, other aldehydes, acids, and ultrafine particles, as outdoor ozone has been shown to react with other compounds as it moves indoors (Nazaroff & Weschler, 2004; Weschler, 2006). Furthermore, ambient conditions, such as temperature, relative humidity, and wind speed, can also influence indoor air quality. For example, higher wintertime outdoor temperatures contribute to lower ventilation rates by reducing infiltration that results from rising warm air within a building, and strong wind gusts increase ventilation rates by creating a pressure differential between the inside and the outside of a building (Health Canada, 2018). Increased indoor temperatures resulting from higher outdoor temperatures have also been associated with higher air pollutant emissions rates from building materials and higher indoor VOC concentrations in homes (Wallace et al., 1996; Heroux et al., 2010; Xiong et al., 2013). Conversely, adaptive measures, such as increased home airtightness and increased use of air conditioning, could reduce the impact of poor ambient conditions on indoor air quality, although retaining pollutants from indoor sources may be an issue if ventilation is inadequate. Overall, these interactions are complex, variable, and influenced by building parameters and local factors.

5.8.3 Extreme Weather Events and Wildfires

Climate change in Canada has increased the frequency of some extreme weather events (e.g., extreme heat events, heavy precipitation events) and is expected to continue to do so in the future (Bush & Lemmen, 2019) (see Chapter 3: Natural Hazards). These events can have a variety of indoor air quality impacts that adversely affect human health. Power outages associated with extreme weather events, such as floods or severe wind or ice storms, may result in individuals using portable gas-powered generators, oil and gas space heaters, fireplaces, and/or candles indoors (Warren & Lemmen, 2014). These devices can lead to high levels of indoor air pollutants, such as CO, $PM_{2.5}$, black carbon, ultrafine particles, NO_2, and polycyclic aromatic hydrocarbons, which have been associated with a range of adverse health effects, including increased risk of death. For example, in 1998, 28 deaths reported during an ice storm that resulted in large power outages across much of Eastern Canada were largely attributed to CO poisoning (Hartling et al., 1998; Berry et al., 2008). Power outages can also lead to the failure of mechanical ventilation systems, resulting in under-ventilated homes and buildings and, consequently, the build-up of air pollutants generated indoors (IOM, 2011). As the climate continues to change, extreme weather events are expected to increase the risk of impacts to energy infrastructure across Canada (CEA, 2018), indicating a need for adaptive measures to mitigate the adverse health effects of deteriorating indoor air quality as a result of power outages.

Climate change is also projected to result in a higher frequency of heavy precipitation events, as well as increased storm surges and coastal flooding in Canada (Bush & Lemmen, 2019) (see Chapter 3: Natural Hazards and Chapter 7: Water Quality, Quantity, and Security). Such events can result in flooding and water intruding into indoor space, creating conditions favourable to the growth of bacteria and fungi such as mould (Health Canada, 2007). For example, Hurricane Katrina in 2005 resulted in extensive proliferation of mould inside many homes in the affected regions of Louisiana (Solomon et al., 2005). Exposure to mould has been associated with eye, nose, and throat irritation; coughing and phlegm build-up; wheezing and shortness of breath; as well as increased prevalence of asthma symptoms (Health Canada, 2007). Furthermore, intrusion of water indoors can result in water damage to building materials and higher air pollutant emission rates from damp building materials (Korpi et al., 1998; Wolkoff, 1998; Huang et al., 2016). In 2013, a major flood in Alberta resulted in the evacuation of 100,000 residences and caused insured property damage that exceeded $1.74 billion

(Insurance Bureau of Canada, 2013). Finally, renovations or repairs that follow interior water damage can result in higher VOC exposures due to emissions from new building materials (Weschler, 2009; ECA, 2013).

Wildfire events may also increase in severity and frequency under a changing climate, which could worsen both indoor and outdoor air pollution in affected areas (see section 5.6 Climate Change and Air Pollution from Wildfires). The impact of wildfire smoke on the indoor environment depends on a variety of housing characteristics; therefore, certain residences may be more affected than others. As wildfires become more prevalent, adaptive housing measures, such as air conditioners and stand-alone air cleaners, should be employed to protect occupants against the adverse health effects of wildfire smoke. For example, the infiltration of PM has been shown to be higher in parts of Canada with moderate climates where air conditioning use is less prevalent and houses are often less airtight (Clark et al., 2010). Moreover, infiltration of PM is lower in newer homes and residences with air cleaners (Barn et al., 2008; Hystad et al., 2009; Clark et al., 2010; MacNeill et al., 2012; Kearney et al., 2014; MacNeill et al., 2014; Wheeler et al., 2014).

In an emergency situation, people may need to congregate in a cleaner-air shelter. However, such shelters can pose unique indoor air quality challenges related to elevated CO_2 levels, indoor temperatures, and relative humidity (Barn, 2014; Keefe, 2014; US EPA, 2016) due to high occupancy. Maintaining clean air in a shelter during periods of high ambient air pollution can require specific strategies such as filtration, air cleaning, and air conditioning (Health Canada, 2020). Public health risks from infectious diseases are also a concern when sheltering people during emergencies. Public health protection during the 2020 COVID-19 pandemic has required approaches to cleaner-air shelters to be adjusted, including guidance regarding screening of symptomatic individuals, implementation of physical distancing, and provision of prevention supplies (US CDC, 2020).

5.8.4 Populations at Higher Risk

People with pre-existing health conditions are especially susceptible to the health impacts of poor indoor air quality (Dales et al., 2008a; To et al., 2009; Potera, 2011; Fann et al., 2016). Children have also been shown to be more susceptible to environmental pollutants (Faustman et al., 2000), and aging can lead to the deterioration of immune defences and lung function, as well as a predisposition to respiratory infections (Viegi et al., 2009).

Other factors, such as an individual's ability to adapt to or mitigate the adverse effects of climate change on their indoor environment can also influence their vulnerability to related health impacts. People who live in multi-family dwellings and/or rent their home may not be able to control temperature or humidity levels, which can influence indoor air quality by increasing pollutant emissions from building materials or enhancing the growth of mould, respectively. Renters may not be able to make modifications to their home to protect against water intrusion and infiltration of wildfire smoke (IOM, 2011; Romero-Lankao et al., 2014). In addition, those who lack financial resources or knowledge may not be able to take the necessary protective actions when faced with changes to their indoor air quality as a result of climate change (IOM, 2011). Poorly designed and poorly maintained dwellings can cause increased exposure to chemicals, moulds, and pathogens; poorly vented combustion apparatus contributes to acute and chronic disease; and exposure to environmental tobacco smoke is a significant health risk to adults and children (Sequel et al., 2017). Geographic location can also increase an individual's vulnerability to health impacts, as certain areas will be more prone to extreme climate-related events such as flooding or wildfires (see Chapter 3: Natural Hazards).

First Nations, Inuit, and Métis peoples may experience disproportionate health impacts from poor indoor air quality, given the existing unequal burden of illness in some Indigenous communities. For example, First Nations and Inuit children have been shown to have increased rates of severe lower respiratory tract infections requiring hospitalization (Kovesi, 2012; McCuskee et al., 2014) and an increased prevalence of bronchiectasis was reported in Inuit children (Das & Kovesi, 2015). Smoking is more prevalent among Indigenous populations, with 27% of off-reserve First Nations, 26% of Métis, and 49% of Inuit people aged 12 and older smoking daily, compared to 15% of non-Indigenous people (Statistics Canada, 2015). Importantly, while high-risk behaviours, such as smoking, have adverse health impacts, their prevalence is symptomatic of "deeper social and economic issues, as well as the legacy of colonialism" (ITK, 2014).

In Canada, tuberculosis rates are four times higher among Métis people, 57 times higher among First Nations people living on reserve, 24 times higher among First Nations people living off reserve, and 284 times higher among Inuit compared to Canadian-born non-Indigenous people (PHAC, 2018; Vachon et al., 2018). In addition, heart disease has been found to be 1.5 times higher among First Nations adults living on reserve compared with the general Canadian population (Indigenous Services Canada, 2018). These existing health inequities can compound the health risks related to climate change impacts on indoor air quality.

First Nations, Inuit, and Métis peoples commonly experience higher rates of poverty, overcrowding in homes, and poor housing quality (Adelson, 2005; NCCAH, 2017; Statistics Canada, 2017), which can increase the risk of health impacts from poor indoor air quality. For example, 27% of First Nations people with registered or "Treaty Indian" status and 26% of Inuit lived in a dwelling in need of major repairs in 2016 (Statistics Canada, 2017), and more than half of adults in First Nations communities reported the presence of mould or mildew in their homes (Health Canada, 2014). Similarly, the 2011 census indicated that, among Inuit, one-third of all dwellings were in need of major repairs in comparison to 14% of Métis homes and 7% for the overall Canadian population (NCCAH, 2017). Climate change impacts on indoor air quality are expected to exacerbate health risks related to poor quality and overcrowded housing.

5.8.5 Adaptation

Adaptation strategies to address indoor air quality in Canada require a multi-faceted risk management approach that incorporates pollution source control, ventilation, and filtration (Poulin et al., 2016). Increasing the airtightness of building envelopes can contribute to a build-up of indoor air pollutants in the absence of adequate ventilation. This can be addressed through the installation and proper maintenance of mechanical ventilation systems (IOM, 2011; Poulin et al., 2016; Health Canada, 2018) as well as by reducing or eliminating indoor sources of air pollutants, for example, through the use of low-VOC products (Poulin et al., 2016). In addition, the influence of deteriorating ambient conditions can be reduced by ensuring a tightly sealed building envelope, using air conditioning and stand-alone air cleaning devices, as well as by installing high-efficiency filters on furnaces. Many of these measures may be inaccessible to some individuals and subpopulations across the country, including Indigenous communities, due to social inequities that result in inadequate housing, low socio-economic status, and insufficient resources to implement protective measures. Government actions to improve ambient air quality will also help to improve air quality in the indoor environment (Poulin et al., 2016).

Flood-prevention measures can also mitigate or reduce home damage from flood and other water infiltration events (Warren & Lemmen, 2014). Furthermore, installation of CO alarms in every residence can help prevent deaths from CO poisoning during power outages. Finally, targeted actions such as increased poison control and other medical services, as well as support for building and infrastructure improvements, provided to rural, geographically affected, or low-income communities can help protect those who may be at increased risk.

5.8.6 Key Uncertainties

There are several key uncertainties regarding the magnitude of climate change impacts on indoor air quality in Canada. For example, climate change may affect patterns of activity, resulting in people spending more time indoors in some conditions and more time outdoors in others. The Canadian housing stock may change as environmental sustainability and climate resilience are increasingly considered. This might include more construction that meets standards such as net zero (buildings that generate as much on-site renewable energy as they consume [Singh et al., 2019]) or passive homes (ultralow-energy building design [Wright & Klingenberg, 2015]), or result in the development of novel building materials engineered to withstand water damage or increase fire resistance. It also remains unclear whether the new adaptive strategies available to homeowners will be adopted and implemented, what forces will drive them (e.g., costs, insurance requirements, etc.), and whether they will be sufficient to mitigate the indoor air quality issues associated with climate change.

5.9 Impacts of Climate Change on Aeroallergens

5.9.1 Impact of Climate Change on Pollen Concentrations, Distribution, and Seasonal Length in Canada

The levels of aeroallergens, including tree pollen, grass pollen, ragweed pollen, and fungal spores are increasing in specific regions of the world and in Canada, and some of this increase has been linked to climate change (Ariano et al., 2010; Sierra-Heredia et al., 2018; Ziska et al., 2019). The timing and seasonal length of aeroallergens, as well as the production and allergenic content of pollen grains, will continue to be affected by climate change (see Figure 5.6) (Ariano et al., 2010; Ziska & Beggs, 2012; Bonofiglio et al., 2013).

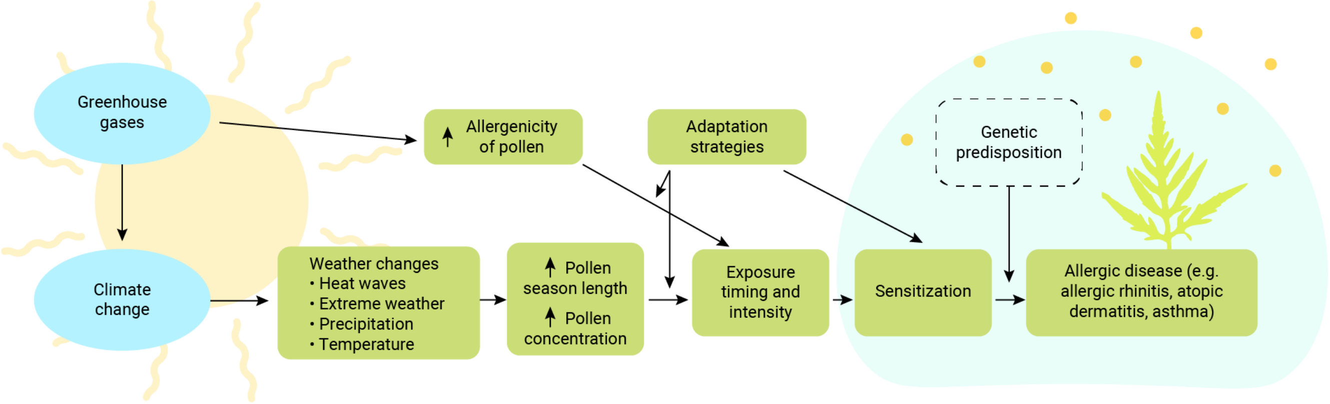


Figure 5.6 Effects of climate change on aeroallergens in Canada.

Due to climate change and related increases in CO_2 emissions, temperatures will increase and, as a result, aeroallergen seasons will start earlier and end later (Traidl-Hoffmann et al., 2003; D'Amato et al., 2014; Rice et al., 2014; D'Amato et al., 2016). In North America, the ragweed pollen season was found to have increased by 27 days between 1995 and 2009 due to warming temperatures (Takaro et al., 2013). In addition, increases in the atmospheric concentrations of CO_2 can affect the reproductive processes of plants, which will increase the production of pollen (Taylor et al., 2007; Shea et al., 2008; Ariano et al., 2010; Ariano et al., 2015; Bjerg et al., 2016). Moreover, there is evidence that higher growing temperatures and increased CO_2 emissions can increase the allergenicity (ability to induce an allergic response) of pollen, and climate change will enhance this effect (Beggs, 2004; Stach et al., 2007). Climate change will also affect regional changes in meteorological variables (e.g., humidity, precipitation, and temperature) linked to pollen dispersal and deposition (D'Amato et al., 2015). These regional climate changes will also affect plant distribution, as species that could not survive in previously hostile environments can potentially thrive because of changes in temperature and precipitation (Stach et al., 2007). Changes in the dispersal patterns of aeroallergens, longer pollen seasons, greater production of pollen grains, and increased allergenicity will lead to changes in human exposure and possibly in sensitization of individuals to allergens (Breton et al., 2006; Reid & Gamble, 2009; Bonofiglio et al., 2013). The magnitude of the impact on aeroallergens and related health effects depends, in part, on the effectiveness of adaptation actions. Adaptation strategies can help to reduce exposure to aeroallergens; for instance, research conducted in the province of Quebec showed that reducing pollen-producing plants in an area could reduce exposure to aeroallergens (Demers & Gosselin, 2019).

5.9.2 Health Effects of Changes in Aeroallergens Under Climate Change Scenarios

Approximately 20% to 25% of the Canadian population is affected by allergic rhinitis, which is most commonly due to pollen allergy (Vaitla & Drewe, 2011; Keith et al., 2012). Asthma affects between 12% to 25% of Canadian children and is estimated to affect about 3 million Canadians overall (Asher et al., 2006; Gershon et al., 2010; Ismaila et al., 2013; Sierra-Heredia et al., 2018). About two-thirds of asthmatic individuals are allergic to aeroallergens, which act as triggers for asthma exacerbations (Lafeuille et al., 2013). A number of studies in Canada have evaluated the health effects of day-to-day fluctuations in aeroallergens. Ambient aeroallergens, including tree pollen, grass pollen, ragweed pollen, and fungal spores, have been associated with increased risk of asthma-related and allergic rhinitis-related emergency department visits and hospitalizations in cities across Canada (Dales et al., 2000; Cakmak et al., 2002; Dales et al., 2004; Dales et al., 2008b; Heguy et al., 2008), as well as an increased risk of myocardial infarction among the elderly (Weichenthal et al., 2016), and earlier delivery among term pregnancies (Lavigne et al., 2017). In addition, high aeroallergen counts during the gestational period have been associated with increased risk of atopic disease in the child later in life (Lowe et al., 2012). As climate change continues to intensify, it is anticipated that allergy sufferers will experience increased exposure to aeroallergens in Canada. The incidence and prevalence of respiratory allergies and asthma are therefore projected to increase, which will be associated with an increase in health care expenses to treat these conditions (Sierra-Heredia et al., 2018).

5.9.3 Adaptation

For individuals suffering from seasonal aeroallergens, aeroallergen alert systems that provide knowledge of current or forecast pollen levels may aid in efforts to choose the right medications to treat symptoms (Lougheed et al., 2010; D'Amato et al., 2015). Health care providers can discuss optimal therapies for allergic rhinitis with their patients. For example, there is evidence that efficacies of prescribed and over-the-counter medications for allergic rhinitis are enhanced if they are taken consistently or before symptom onset (Kim et al., 2008; Keith et al., 2012). In addition, reminders and warnings that the aeroallergen season is approaching may allow Canadians to ensure that they have visited a health care provider, refilled prescriptions, and begun taking preventive medications according to their management plan (Johnston et al., 2018).

An aeroallergen alert system is also a beneficial communication strategy to advise people at risk to control their exposure when aeroallergen levels are high (Sierra-Heredia et al., 2018). In Canada, daily aeroallergen forecasts are provided by Aerobiology Research Laboratories (ARL) through the Weather Network (Aerobiology Research Laboratories, 2019a; The Weather Network, 2019). A free application provided by ARL can also be downloaded on a smart phone in order to obtain forecasts of pollen and fungal spores (Aerobiology Research Laboratories, 2019b). Other potential risk-mitigation strategies at a population level include the greening of cities with trees and species that minimize allergenicity for people exposed (Fuertes et al., 2016; Carinanos et al., 2017; Fong et al., 2018). Strategies in the province of Quebec have shown that engaging various partners at the municipal level in synchronizing their ragweed-control actions can reduce exposure to ragweed pollen (Demers & Gosselin, 2019). This includes mowing regularly, applying low-impact herbicides, and planting a competitive plant cover to prevent the spread of ragweed.

5.9.4 Key Uncertainties

Although several studies have identified health effects associated with exposure to aeroallergens, key uncertainties remain. A better understanding of the spatial coverage of aeroallergens and interactions with air pollutants and urban green spaces is needed to provide information that at-risk populations can use to modify their exposures and to shape urban greening initiatives (D'Amato et al., 2015; Sierra-Heredia et al., 2018). Additional information on the health benefits and effectiveness of adaptation strategies, such as the use of alert systems and aeroallergen control actions, is also needed. Further studies examining long-term impacts of climate change on aeroallergens would help to explain regional differences and increase understanding of the key characteristics of climate change on pollen load and seasonal length (Ziska et al., 2019). Projections of the impacts of climate change on future levels of aeroallergens are also required.

5.10 Conclusion

5.10.1 Climate Change and Air Quality Health Impacts in Canada

Climate change and air quality are intimately linked; a warming climate can worsen air pollution (i.e., the climate penalty), and some air pollutants, including ozone and components of $PM_{2.5}$, can affect the climate and enhance warming. GHGs and air pollutants are also derived from common sources linked to fossil fuel combustion; therefore, strategies to address one may have important co-benefits of reducing emissions of the other.

Air pollution is the leading environmental cause of death in Canada, contributing to an estimated 15,300 deaths annually, along with many non-fatal outcomes (Health Canada, 2021). Recent research indicates that the adverse effects of air pollution extend beyond cardiorespiratory impacts, linking exposure to varied outcomes such as diabetes, dementia, and reproductive health. Even small increases in exposure to air pollution are associated with an increased risk of adverse health impacts. The broad range of adverse health effects caused by air pollution, along with the ubiquitous and involuntary nature of air pollution exposure, highlight the importance of air quality management as a key public health issue.

Studies that have quantified the climate penalty in terms of air pollution health impacts under various climate projections have focused primarily on ozone, with some addressing $PM_{2.5}$ as well. Analyses undertaken for this assessment estimate hundreds of excess annual deaths due to air pollution in 2050, associated with the impact of climate change on air pollution, primarily in Ontario and Quebec, with a net social value of $2.7 billion. Similarly, studies in the United States have reported hundreds to thousands of excess annual air pollution deaths later this century associated with a warming climate.

Importantly, many studies have reported the considerable potential air pollution health co-benefits of pursuing climate change mitigation strategies that target either long-lived climate pollutants (e.g., CO_2) or short-lived climate forcers (e.g., black carbon or methane). Although climate benefits of near-term GHG emission reductions may only be realized in the mid- to long-term and generally occur globally, the public health benefits of the associated reductions in air pollutant emissions would be realized immediately and locally where the emission reductions are implemented. In addition, these benefits can have a large social value, which can offset a portion of the GHG mitigation costs. Incorporating air quality health co-benefits into climate change mitigation policy provides additional justification for pursuing more stringent or accelerated reduction measures to address climate change. Joint consideration of options to reduce both GHG and air pollution emissions would contribute to the strategic development of policies that optimize both and help to avoid unintended negative consequences, such as inadvertent increases in air pollution emissions from implementation of GHG mitigation strategies.

Wildfires have become an important source of air pollution in Canada, and increasing wildfire emissions due to climate change represent one of the most significant risks to air quality. It is expected that wildfire frequency, severity, and distribution will change under a warming climate, resulting in higher emissions, and the evidence suggests that some of the current increased wildfire activity can already be attributed to climate change. Exposure to wildfire smoke affects respiratory health, while the evidence for cardiovascular outcomes remains inconclusive. A Canadian analysis estimates 620 to 2700 annual deaths attributable to wildfire emissions during the 2013 to 2018 period, with an annual net social value ranging from $4.7 billion to $21 billion, as well as many non-fatal cardiorespiratory effects. Overall, the greatest impacts were noted in British Columbia and Alberta, although during the 2013 fire season the greatest impacts were reported in Ontario and Quebec. Although the highest air pollution concentrations are expected to occur closer to the fires, smoke plumes can spread over vast areas of the country, affecting population centres far from the sources. For example, 20% to 30% of the Canadian population was exposed to average wildfire $PM_{2.5}$ concentrations of 1 $\mu g/m^3$ or more during the May to September season in 2014, 2015, 2017, and 2018. The current air pollution health impacts of wildfire smoke are expected to increase under climate change, making this an important public health issue.

Canadians spend 90% of their time indoors, and climate change is expected to impact indoor air quality in a variety of ways. It is anticipated that climate change will include more frequent extreme weather events, which raise the risk of health effects from mould due to flooding, smoke from wildfires, and indoor air pollution from the inappropriate use of combustion sources, among other impacts. In addition, increased home airtightness associated with improved energy efficiency requires sufficient ventilation to ensure pollutants do not accumulate indoors. Conversely, under conditions of poor outdoor air pollution, impacts on indoor air quality can be mitigated through increased home airtightness and the use of air conditioning and filtration. Economic resources are typically needed to take protective measures, such as increasing home airtightness and ventilation. Thus, low-income households and socially disadvantaged communities may face challenges with implementing such measures.

Ambient levels of aeroallergens, including pollen and fungal spores, have been associated with increased risk of asthma-related and allergic rhinitis-related hospital visits in Canada, as well as other adverse effects. Warming temperatures, changing weather conditions, and rising atmospheric CO_2 levels result in increases in pollen counts, season length, and allergenicity, as well as changes to species distribution. Asthma affects about 3 million Canadians, including 12% to 25% of children, while approximately one-quarter of Canadians suffer from allergic rhinitis. Both of these conditions are commonly triggered by aeroallergens. The incidence and prevalence of asthma and allergic rhinitis are projected to increase with climate change.

5.10.2 Populations at Higher Risk

The scientific evidence indicates that multiple subpopulations are at increased risk from the adverse health effects of air pollution and aeroallergens and would therefore likely be more affected by worsening outdoor air pollution, wildfire smoke, indoor air pollution, and airborne allergen concentrations resulting from a changing climate.

Some groups are more susceptible to air pollutants due to age (children and seniors), pre-existing disease (e.g., asthma, COPD, or heart disease) or genetic predispositions. Growing evidence of air pollution effects on reproductive outcomes, diabetes, and progression of cardiac disease, among other health effects, suggests that a large proportion of the population may be at elevated risk. Other groups may be at higher risk due to elevated exposures because of where they live or how much time they spend outdoors, including for occupational purposes. A recent study estimates that one-third of the Canadian population has a least one risk factor making them at greater risk than the general population to the adverse effects of ozone and $PM_{2.5}$, highlighting the need for risk communication and interventions to manage air quality to target both highly exposed and more susceptible populations (Stieb et al., 2019).

Key populations at higher risk from the increasing pollen exposure and allergenicity expected under climate change include those with asthma and allergic rhinitis, and recent evidence suggests that other populations, including those with heart disease, may also be affected.

Importantly, in Canada, Indigenous populations bear a disproportionately higher burden of respiratory diseases, including asthma and COPD, than the general population, increasing their susceptibility to outdoor air pollution, wildfire smoke, indoor air pollution, and aeroallergens. Multiple, compounding health inequities (see Chapter 9: Climate Change and Health Equity) may further contribute to higher vulnerability. Indigenous Peoples living in remote communities may also be more likely to experience high levels of wildfire smoke.

5.10.3 Adaptation

A key goal of air quality management in Canada is continuous improvement, through reductions in air pollutant emissions across multiple sectors (CCME, 2019). However, additional programs have been developed to address the health risks of air pollution by reducing exposure, and these programs will become increasingly important to counterbalance the expected negative impacts of climate change.

The AQHI serves as a health-protection tool designed to help Canadians make decisions to protect their health by limiting short-term exposure to outdoor air pollution. The AQHI reports current and forecast conditions daily for communities across Canada and provides specific advice related to air quality levels associated with low, moderate, high, and very high health risks, including for susceptible individuals. In addition, local authorities use air quality alerts or advisories under high-risk conditions. The AQHI does not measure effects from heat, and there is currently only limited integration of heat and air quality messages for the public in Canada. The Government of Canada also provides forecasts of wildfire $PM_{2.5}$ levels for the next 48 hours, which include mapping of predicted smoke paths, and fire smoke air quality alerts are issued when necessary. The British Columbia Centre for Disease Control has implemented an alternative version of the

AQHI during the wildfire season to better reflect risks during wildfire smoke events, and multiple provinces and territories have information for protecting community health from wildfire smoke.

Enhancing urban green space can have multiple co-benefits for health, including direct positive health impacts as well as a role in attenuating exposure to air pollution and contributing to the reduction of the urban heat island effect.

Adaptation strategies to address indoor air quality in Canada under a changing climate will require a multi-faceted risk management approach that incorporates control of indoor sources of air pollutants, adequate ventilation under increasing home airtightness, and air filtration. Flood-prevention strategies would help to mitigate mould caused by water damage, and increased use of CO detection would help to prevent deaths from CO poisoning during power outages. In addition, targeting actions to rural, geographically affected, or low-income communities can help to protect populations at higher risk.

Daily aeroallergen forecasts, which can aid in reducing exposure to aeroallergens and optimization of pharmacotherapy for allergy sufferers, are currently provided in Canada by ARL through the Weather Network. Community-based adaptation strategies to reduce pollen levels could include urban greening with low-allergenicity species. A project in Quebec has shown that municipal-level partnerships may be effective in coordinating multiple control strategies to address ragweed, a common allergen species.

5.10.4 Knowledge Gaps

Several important knowledge gaps remain in understanding of how the health of Canadians will be affected by air quality under a changing climate. Integrated modelling of climate change and air quality, including the effects of climate parameters on $PM_{2.5}$ levels, is needed to improve understanding of both the population health impacts associated with the climate penalty and the potentially large air quality health co-benefits of GHG mitigation measures. The impact of changing climatic conditions on biogenic emissions needs to be better understood and incorporated into air quality models. In addition, an overall synthesis and comparison of the potential air quality co-benefits of multiple IPCC climate mitigation pathways remain difficult because studies to date have used different approaches, assumptions, and modelling methodologies. Aligning future study methodologies would provide improved information for pursuing climate change mitigation policies.

Improved capacity to model wildfire smoke exposure and to understand interactions between climate and wildfire risk is required to inform projections of wildfire smoke health impacts under climate change. Improved understanding of the range of adverse health effects associated with air pollution exposure, including whether the health effects of wildfire smoke are different from those of ambient air pollution, would provide a more comprehensive assessment of the population health impacts and identify populations at higher risk.

Although it is well known that Canadians spend 90% of their time indoors, characterizing pollutant exposures indoors and the associated health risks is challenging. Changing climate conditions, as well as mitigation and adaptation measures, increase the complexity of this problem. There is a need for research that supports the assessment of the health implications of changing environmental conditions and which can guide healthy building design, energy saving, ventilation, and material selection.

Recent research suggests that the spectrum of health impacts associated with airborne allergen exposure may go beyond respiratory outcomes, which may help to identify new susceptible groups. A better understanding of aeroallergen distribution and interactions with air pollutants and green space would help to inform urban greening initiatives and adaptation measures to protect populations at higher risk.

Finally, further research is required to inform the development of effective adaptation and risk mitigation strategies for wildfire smoke, indoor air quality, and aeroallergens under a changing climate to better protect the health of Canadians.

5.11 References

Abelsohn, A., & Stieb, D. M. (2011). Health effects of outdoor air pollution: approach to counseling patients using the Air Quality Health Index. *Canadian family physician, 57*(8), 881–e287.

Adelson, N. (2005). The embodiment of inequity: health disparities in aboriginal Canada. *Canadian journal of public health, 96*(2), S45–S61. <https://doi.org/10.1007/BF03403702>

Adetona, O., Reinhardt, T.E., Domitrovich, J., Broyles, G., Adetona, A. M., Kleinman, M.T., Ottmar, R. D., & Naeher, L. P. (2016). Review of the health effects of wildland fire smoke on wildland firefighters and the public. *Inhalation Toxicology, 28*(3), 95-139. doi:10.3109/08958378.2016.1145771

Aerobiology Research Laboratories. (2019a). Retrieved from <https://www.pollenexperts.ca/what-we-do/>

Aerobiology Research Laboratories. (2019b). *Personalized pollen reports.* Retrieved from <https://www.pollenexperts.ca/get-involved/#gettheapps>

Alam, S., Lang, J. J., Drucker, A.M., Gotay, C., Kozloff, N., Mate, K., Patten, S.B., Orpana, H. M., Afshin, A., & Cahill, L. E. (2019). Assessment of the burden of diseases and injuries attributable to risk factors in Canada from 1990 to 2016: an analysis of the Global Burden of Disease Study. *Canadian Medical Association Journal Open*, *7*(1), E140-E148. doi:10.9778/cmajo.20180137

Albert-Green, A., Dean, C. B., Martell, D. L., & Woolford, D. G. (2013). A methodology for investigating trends in changes in the timing of the fire season with applications to lightning-caused forest fires in Alberta and Ontario, Canada. *Canadian Journal of Forest Research, 43*(1), 39-45. doi:10.1139/cjfr-2011-0432

Alberta Health Services. (2019). *Frequently Asked Questions: Wildfires / Air Quality due to Wildfire Smoke.* Government of Alberta. Retrieved from <https://www.albertahealthservices.ca/assets/wf/eph/wf-eph-wildfire-smoke-health.pdf>

Amann, M., Klimont, Z., & Wagner F. (2013). Regional and global emissions of air pollutants: Recent trends and future scenarios. *Annual Review of Environment and Resources, 38,* 31-55. doi:10.1146/annurev-environ-052912-173303

Amiro, B. D., Cantin, A., Flannigan, M. D., & de Groot, W. (2009). Future Emissions from Canadian Boreal Forest Fires. *Canadian Journal of Forest Research, 39*, 383-395. doi:10.1139/X08-154

Anderson, V. (2016). A Harmonized Heat Warning and Information System for Ontario (HWIS) Standard Operating Practice. Toronto, ON: Ontario Ministry of Health and Long-term Care. Retrieved from <http://www.health.gov.on.ca/en/common/ministry/publications/reports/heat_warning_information_system/heat_warning_information_system.aspx>

Ariano, R., Berra, D., Chiodini, E., Ortolani, V., Cremonte, L. G., Mazzarello, M. G., Galdi E., Calosso C., & Ciprandi, G. (2015). Ragweed allergy: Pollen count and sensitization and allergy prevalence in two Italian allergy centers. *Allergy & Rhinology (Providence, R.I.), 6*(3), 177-183. doi:10.2500/ar.2015.6.0141

Ariano, R., Canonica, G. W., & Passalacqua, G. (2010). Possible role of climate changes in variations in pollen seasons and allergic sensitizations during 27 years. *Annals of Allergy, Asthma & Immunology: Official Publication of the American College of Allergy, Asthma, & Immunology, 104*(3), 215-222. doi:10.1016/j.anai.2009.12.005

Asher, M. I., Montefort, S., Bjorksten, B., Lai, C. K., Strachan, D. P., Weiland, S. K., Williams, H., & ISAAC Phase Three Study Group. (2006). Worldwide time trends in the prevalence of symptoms of asthma, allergic rhinoconjunctivitis, and eczema in childhood: ISAAC phases one and three repeat multicountry cross-sectional surveys. *Lancet (London, England), 368*(9537), 733-743. <https://doi.org/10.1016/S0140-6736(06)69283-0>

Baldauf, R. (2016): Recommendations for Constructing Roadside Vegetation Barriers to Improve Near-Road Air Quality, U.S. Environmental Protection Agency, Washington, DC, EPA/600/R-16/072. <https://cfpub.epa.gov/si/si_public_record_report.cfm?Lab=NRMRL&dirEntryId=321772&simpleSearch=1&searchAll=Recommendations+for+constructing+roadside+vegetation+barriers+to+improve+near+road+air+quality> [November 2020]

Barn, P. (2014). *Evidence review: Home and community clean air shelters to protect public health during wildfire smoke events (C. Elliott, & K. Rideout, Eds).* BC Centre for Disease Control. Retrieved from <http://www.bccdc.ca/resourcegallery/Documents/Guidelines%20and%20Forms/Guidelines%20and%20Manuals/Health-Environment/WFSG_EvidenceReview_CleanAirShelters_FINAL_v3_edstrs.pdf>

Barn, P., Larson, T., Noullett, M., Kennedy, S., Copes R., & Brauer, M. (2008). Infiltration of forest fire and residential wood smoke: an evaluation of air cleaner effectiveness. *Journal of Exposure Science and Environmental Epidemiology, 18*(5), 503-511. doi:10.1038/sj.jes.7500640

Barnes, E. A., Fiore, A. M., & Horowitz, L. W. (2016). Detection of trends in surface ozone in the presence of climate variability. *Journal of Geophysical Research: Atmospheres, 121*(10), 6112 – 6129, doi:10.1002/2015jd024397

BC Centre for Disease Control (BC CDC). (2014). Wildfire Smoke Response Planning. Retrieved from <http://www.bccdc.ca/health-professionals/professional-resources/wildfire-smoke-response-planning>

BC Centre for Disease Control (BC CDC). (2019). The burning reality of wildfire smoke and its public health impacts. Retrieved from <http://www.bccdc.ca/about/news-stories/stories/the-burning-reality-of-wildfire-smoke-and-its-public-health-impacts>

Beggs, C. B., Noakes, C. J., Sleigh, P. A., Fletcher, L. A., & Siddiqi, K. (2003). The transmission of tuberculosis in confined spaces: an analytical review of alternative epidemiological models. *The International Journal of Tuberculosis and Lung Disease, 7*(11), 1015-26.

Beggs, P. J. (2004). Impacts of climate change on aeroallergens: Past and future. Clinical and Experimental Allergy. *Journal of the British Society for Allergy and Clinical Immunology, 34*(10), 1507-1513. doi:10.1111/j.1365-2222.2004.02061.x

Bell M. L., Goldberg R., Hogrefe C., Kinney P. L., Knowlton K., Lynn B., Rosenthal J., Rosenzweig C., & Patz, J. A. (2007). Climate change, ambient ozone, and health in 50 US cities. *Climatic Change*, 82, 61–76. doi: 10.1007/s10584-006-9166-7

Benmarhnia, T., Mathlouthi, F., & Smargiassi, A. (2013). Health impacts of particles from forest fires. Institut National de Santé Publique du Québec. Retrieved from <https://www.inspq.qc.ca/pdf/publications/1793_Health_Impacts_Forest_Fires.pdf>

Berry, P., McBean, G., & Seguin, J. (2008). Vulnerabilities to natural hazards and extreme weather. In J. Seguin (Ed.), Human Health in a Changing Climate: A Canadian Assessment of Vulnerabilities and Adaptive Capacity (pp. 43-112). Ottawa, ON: Health Canada. <http://publications.gc.ca/collections/collection_2008/hc-sc/H128-1-08-528E.pdf>

Bjerg, A., Ekerljung, L., Eriksson, J., Näslund, J., Sjölander, S., Rönmark, E., Dahl, Å., Holmberg, K., Wennergren, G., Torén, K., Borres, M. P., Lötvall, J., & Lundbäck, B. (2016). Increase in pollen sensitization in Swedish adults and protective effect of keeping animals in childhood. *Clinical and Experimental Allergy: Journal of the British Society for Allergy and Clinical Immunology*, *46*(10), 1328–1336. <https://doi.org/10.1111/cea.12757>

Black, C., Tesfaigzi, Y., Bassein, J. A., & Miller, L. A. (2017). Wildfire smoke exposure and human health: significant gaps in research for a growing public health issue. *Environmental Toxicology and Pharmacology, 55,* 186-195. doi:10.1016/j.etap.2017.08.022

Bond, T. C., Doherty, S. J., Fahey, D. W., Forster, P. M., Berntsen, T., DeAngelo, B. J., Flanner, M. G., Ghan, S., Kärcher, B., Koch, D., Kinne, S., Kondo, Y., Quinn, P. K., Sarofim, M. C., Schultz, M. G., Schulz, M., Venkataraman, C., Zhang, H., Zhang, S., Bellouin, N.,… Zender, C. S. (2013). Bounding the role of black carbon in the climate system: A scientific assessment. *Journal of Geophysical Research: Atmospheres, 118,* 5380–5552, doi:10.1002/jgrd.50171.

Bone, A., Murray, V., Myers, I., Dengel, A., & Crump, D. (2010). Will drivers for home energy efficiency harm occupant health? *Perspectives in Public Health, 130*(5), 233-238. doi:10.1177/1757913910369092

Bonofiglio, T., Orlandi, F., Ruga, L., Romano, B., & Fornaciari, M. (2013). Climate change impact on the olive pollen season in Mediterranean areas of Italy: Air quality in late spring from an allergenic point of view. *Environmental Monitoring and Assessment, 185*(1), 877-890. doi:10.1007/s10661-012-2598-9

Boucher, O., Randall D., Artaxo P., Bretherton C., Feingold G., Forster P., Kerminen V.-M., Kondo Y., Liao H., Lohmann U., Rasch P., Satheesh S. K., Sherwood S., Stevens B., & Zhang X. Y. (2013). Clouds and Aerosols. In T. F. Stocker, D. Qin, G.-K. Plattner, M. Tignor, S. K. Allen, J. Boschung, A. Nauels, Y. Xia, V. Bex, & P. M. Midgley (Eds.), Climate Change 2013: The Physical Science Basis. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change (pp. 571-658). Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA, doi:10.1017/CBO9781107415324.016

Boulanger, Y., Gauthier S., & Burton P. J. (2014). A refinement of models projecting future Canadian fire regimes using homogeneous fire regime zones. *Canadian Journal of Forest Research, 44*, 365 – 376, doi:/10.1139/cjfr-2013-0372.

Breton, M. C., Garneau, M., Fortier, I., Guay, F., & Louis, J. (2006). Relationship between climate, pollen concentrations of ambrosia and medical consultations for allergic rhinitis in Montreal, 1994-2002. *The Science of the Total Environment, 370*(1), 39-50. <https://doi.org/10.1016/j.scitotenv.2006.05.022>

Bush, E., & Lemmen, D. S. (Eds.). (2019). Canada's Changing Climate Report. Ottawa, ON: Government of Canada. Retrieved from <https://changingclimate.ca/CCCR2019/>

Cakmak, S., Dales, R. E., Burnett, R. T., Judek, S., Coates, F., & Brook, J. R. (2002). Effect of airborne allergens on emergency visits by children for conjunctivitis and rhinitis. *Lancet (London, England), 359*(9310), 947-948. <https://doi.org/10.1016/S0140-6736(02)08045-5>

Camalier, L., Cox, W., & Dolwick P. (2007). The effects of meteorology on ozone in urban areas and their use in assessing ozone trends. *Atmospheric Environment, 41*(33), 7127–7137. doi:10.1016/j.atmosenv.2007.04.061.

Cames, M., & Helmers, E. (2013). Critical evaluation of the European diesel car boom – global comparison, environmental effects and various national strategies. *Environmental Sciences Europe, 25*, 15. doi:10.1186/2190-4715-25-15

Canadian Council of Ministers of the Environment (CCME). (2014). Resources: AQMS. Retrieved from <https://www.ccme.ca/en/resources/air/aqms.html>

Canadian Council of Ministers of the Environment (CCME). (2019). *Guidance Document on Air Zone Management*. Retrieved from <https://www.ccme.ca/files/Resources/air/Guidance%20Document%20on%20Air%20Zone%20Management.pdf>

Canadian Electricity Association (CEA). (2018). *Climate Change adaptation.* Retrieved from <https://electricity.ca/lead/protecting-our-environment/climate-change-adaptation/>

Canadian Mortgage and Housing Corporation (CMHC). (1998). Field Survey of Heat Recovery Ventilation Systems Final Report. Ottawa, ON: Canada Mortgage and Housing Corporation Research Division. Retrieved from <http://publications.gc.ca/collections/collection_2011/schl-cmhc/nh18-1/NH18-1-90-1998-eng.pdf>

Cariñanos, P., Casares-Porcel, M., Díaz de la Guardia, C., Aira, M. J., Belmonte, J., Boi, M., Elvira-Rendueles, B., De Linares, C., Fernández-Rodriguez, S., Maya-Manzano, J. M., Pérez-Badía, R., Rodriguez-de la Cruz, D., Rodríguez-Rajo, F. J., Rojo-Úbeda, J., Romero-Zarco, C., Sánchez-Reyes, E., Sánchez-Sánchez, J., Tormo-Molina, R., & Vega Maray, A. M. (2017). Assessing allergenicity in urban parks: A nature-based solution to reduce the impact on public health. *Environmental Research*, *155*, 219–227. <https://doi.org/10.1016/j.envres.2017.02.015>

Carrière, G. M., Garner, R., & Sanmartin, C. (2017). Housing conditions and respiratory hospitalizations among First Nations People in Canada. *Health Reports (Catalogue no. 82-003-X).* Ottawa, ON: Statistics Canada. Retrieved from <https://www150.statcan.gc.ca/n1/pub/82-003-x/2017004/article/14789-eng.pdf>

Cascio, W. E. (2018). Wildland fire smoke and human health. *Science of the Total Environment, 624*, 586-595. doi:10.1007/s10393-004-0149-8

Chand, D., Wood, R., Anderson, T. L., Satheesh, S. K., & Charlson, R. J. (2009). Satellite-derived direct radiative effect of aerosols dependent on cloud cover. *Nature Geoscience, 2,* 181 -184. doi: 10.1038/NGEO437

Chen, H., Burnett, R. T., Kwong, J. C., Villeneuve, P. J., Goldberg, M. S., Brook, R. D., van Donkelaar, A., Jerrett, M., Martin, R. V., Brook, J. R., & Copes, R. (2013a). Risk of incident diabetes in relation to long-term exposure to fine particulate matter in Ontario, Canada. *Environmental Health Perspectives, 121*(7), 804-810. doi: 10.1289/ehp.1205958

Chen, H., Li, Q., Kaufman, J. S., Wang, J., Copes, R., Su, Y., & Benmarhnia, T. (2018). Effect of air quality alerts on human health: a regression discontinuity analysis in Toronto, Canada. *The Lancet Planetary Health, 2*(1), e19-e26. doi:10.1016/S2542-5196(17)30185-7

Chen, R., Wang, X., Meng, X., Hua, J., Zhou, Z., Chen, B., & Kan, H. (2013b). Communicating air pollution-related health risks to the public: An application of the Air Quality Health Index in Shanghai, China. *Environment international, 51*, 168-173. doi.org/10.1016/j.envint.2012.11.008

Clark, N. A., Allen, R. W., Hystad, P., Wallace, L., Dell, S.D., Foty, R., Dabek-Zlotorzynska, E., Evans, G., & Wheeler, A. J. (2010). Exploring variation and predictors of residential fine particulate matter infiltration. *International Journal of Environmental Research and Public Health, 7*(8), 3211-3224. doi: 10.3390/ijerph7083211

Climate and Clean Air Coalition (CCAC). (2020). Why we need to act now. Retrieved from <ccacoalition.org/en/content/why-we-need-act-now>

Cooper, O. R., Parrish D. D., Ziemke J., Balashov N. V., Cupeiro M., Galbally I. E., Gilge S., Horowitz L., Jensen N. R., Lamarque J.-F.,. Naik V, Oltmans S. J., Schwab J., Shindell D. T., Thompson A. M., Thouret V., Wang Y., & Zbinden R. M. (2014). Global distribution and trends of tropospheric ozone: An observation-based review. *Elementa: Science of the Anthropocene, 2,* 000029. doi:/10.12952/journal.elementa.000029.

Coumou, D., Di Capua G., Vavrus S., Wang L., & Wang S. (2018). The influence of Arctic amplification on mid-latitude summer circulation. *Nature Communications, 9*(1), 2959. doi: 10.1038/s41467-018-05256-8.

Crouse, D. L., Pinault, L., Balram, A., Brauer, M., Burnett, R. T., Martin, R. V., van Donkelaar, A., Villeneuve, P. J., & Weichenthal, S. (2019). Complex relationships between greenness, air pollution, and mortality in a population-based Canadian cohort. *Environment International, 128*, 292-300. doi.org/10.1016/j.envint.2019.04.047

Dales, R., Liu, L., Wheeler, A. J., & Gilbert, N. L. (2008a). Quality of indoor residential air and health. *Canadian Medical Association journal = Journal de l'Association Médicale Canadienne*, *179*(2), 147–152. <https://doi.org/10.1503/cmaj.070359>

Dales, R. E., Cakmak, S., Burnett, R. T., Judek, S., Coates, F., & Brook, J. R. (2000). Influence of ambient fungal spores on emergency visits for asthma to a regional children's hospital. *American Journal of Respiratory and Critical Care Medicine*, *162*(6), 2087–2090. <https://doi.org/10.1164/ajrccm.162.6.2001020>

Dales, R. E., Cakmak, S., Judek, S., & Coates, F. (2008b). Tree pollen and hospitalization for asthma in urban Canada. *International Archives of Allergy and Immunology, 146*(3), 241-247. doi:10.1159/000116360

Dales, R. E., Cakmak, S., Judek, S., Dann, T., Coates, F., Brook, J. R., & Burnett, R. T. (2004). Influence of outdoor aeroallergens on hospitalization for asthma in Canada. *The Journal of Allergy and Clinical Immunology, 113*(2), 303-306. doi:10.1016/j.jaci.2003.11.016

D'Amato, G., Bergmann, K. C., Cecchi, L., Annesi-Maesano, I., Sanduzzi, A., Liccardi, G., Vitale, C., Stanziola, A., & D'Amato, M. (2014). Climate change and air pollution: Effects on pollen allergy and other allergic respiratory diseases. *Allergo Journal International, 23*(1), 17-23. doi:10.1007/s40629-014-0003-7

D'Amato, G., Holgate, S. T., Pawankar, R., Ledford, D. K., Cecchi, L., Al-Ahmad, M., Al-Enezi, F., Al-Muhsen, S., Ansotegui, I., Baena-Cagnani, C. E., Baker, D. J., Bayram, H., Bergmann, K. C., Boulet, L.-P., buters, J. T. M., D'amato, M., dorsano, S., Douwes, J., Finlay, S. E., Garrasi, D., …. Annesi-Maesano, I. (2015). Meteorological conditions, climate change, new emerging factors, and asthma and related allergic disorders. A statement of the world allergy organization. *The World Allergy Organization Journal, 8*(1), 25. doi:10.1186/s40413-015-0073-0

D'Amato, G., Vitale, C., Lanza, M., Molino, A., & D'Amato, M. (2016). Climate change, air pollution, and allergic respiratory diseases: An update. *Current Opinion in Allergy and Clinical Immunology, 16*(5), 434-440. doi:10.1097/ACI.0000000000000301

Das, L., & Kovesi, T. A. (2015). Bronchiectasis in children from Qikiqtani (Baffin) Region, Nunavut, Canada. *Annals of the American Thoracic Society*, *12*(1), 96–100. <https://doi.org/10.1513/AnnalsATS.201406-257OC>

Dawson, J. P., Adams P. J., & Pandis S. N. (2007). Sensitivity of $PM_{2.5}$ to climate in the Eastern US: a modeling case study. *Atmospheric Chemistry and Physics, 7*, 4295 – 4309. doi:10.5194/acp-7-4295-2007.

Day, M. C., & Pandis S. N. (2011). Predicted changes in summertime organic aerosol concentrations due to increased temperatures. *Atmospheric Environment, 45*(36), 6546 – 6556. doi:10.1016/j.atmosenv.2011.08.028

Demers, I., & Gosselin, P. (2019). At-a-glance -Pollens, climate and allergies: Quebec initiatives. *Health promotion and chronic disease prevention in Canada, 39*(4), 136-141. <https://doi.org/10.24095/hpcdp.39.4.05>

Deser, C., Knutti, R., Solomon S., & Phillips A. S. (2012). Communication of the role of natural variability in future North American climate. *Nature Climate Change, 2,* 775 – 779. doi: 10.1038/NCLIMATE1562.

Dodd, W., Howard, C., Rose, C., Scott, C., Scott, P., Cunsolo, A., & Orbinski, J. (2018a). The summer of smoke: Ecosocial and health impacts of a record wildfire season in the Northwest Territories, Canada. *The Lancet Global health, 6*(Suppl.2), S30. doi:10.1016/S2214-109X(18)30159-1

Dodd, W., Scott, P., Howard, C., Scott, C., Rose, C., Cunsolo, A., & Orbinski, J. (2018b). Lived experience of a record wildfire season in the Northwest Territories, Canada. *Canadian Journal of Public Health, 109*(3), 327-337. <https://doi.org/10.17269/s41997-018-0070-5>

Du, X., Chen, R., Meng, X., Liu, C., Niu, Y., Wang, W., Li, S., Kan, H., & Zhou, M. (2020). The establishment of National Air Quality Health Index in China. *Environment International*, *138*, 105594-105600. doi.org/10.1016/j.envint.2020.105594

Ebi, K. L., & McGregor, G. (2008). Climate change, tropospheric ozone and particulate matter, and health impacts. *Environment Health Perspectives, 116*(11), 1449-1455. doi:10.1289/ehp.11463

Elliott, C. T., Henderson, S. B., & Wan, V. (2013). Time series analysis of fine particulate matter and asthma reliever dispensations in populations affected by forest fires. *Environmental Health, 12,* 11. doi:10.1186/1476-069X-12-11.

Environment and Climate Change Canada (ECCC). (2020). A Healthy Environment and Healthy Economy: Canada's strengthened climate plan to create jobs and support people, communities and the planet. Gatineau, QC: Government of Canada. Retrieved from <https://www.canada.ca/content/dam/eccc/documents/pdf/climate-change/climate-plan/healthy_environment_healthy_economy_plan.pdf>

European Academies' Science Advisory Council. (2019). *The* imperative of climate action to protect human health in Europe. German National Academy of Sciences. Retrieved from <https://easac.eu/publications/details/the-imperative-of-climate-action-to-protect-human-health-in-europe/>

European Collaborative Action (ECA). (2013). Harmonisation Framework for Health Based Evaluation of Indoor Emissions from Construction Products in the European Union Using the EU-LCI Concept. Luxembourg: Publications Office of the European Union. Retrieved from <http://publications.jrc.ec.europa.eu/repository/handle/JRC83683<

Fan, J., Wang Y., Rosenfeld D., & Liu, X. (2016). Review of aerosol-cloud interactions: mechanisms, significance and challenges. *Journal of the Atmospheric Sciences, 73*(11), 4221 – 4252. doi:10.1175/JAS-D-16-0037.1, 2016.

Fang, Y., Mauzerall, D. J., Liu, J., Fiore, A. M., & Horowitz, L. W. (2013). Impacts of 21st century climate change on global air pollution-related premature mortality. *Climatic Change, 121,* 239–253. doi: 10.1007/s10584-013-0847-8

Fann, N., Alman, B., Broome, R. A., Morgan, G. G., Johnston, F. H., Pouliot, G., & Rappold, A. G. (2018). The health impacts and economic value of wildland fire episodes in the U.S.: 2018-2012. *Science of the Total Environment, 610-611,* 802-809. doi:10.1016/j.scitotenv.2017.08.024

Fann, N., Brennan, T., Dolwick, P., Gamble, J. L., Ilacqua, V., Kolb, L., Notle, C.G., Spero, T. L., & Ziska, L. (2016). Air Quality Impacts. In Crimmins, A., J. Balbus, J. L. Gamble, C. B. Beard, J. E. Bell, D. Dodgen, R. J. Eisen, N. Fann, M. D. Hawkins, S. C. Herring, L. Jantarasami, D.M. Mills, S. Saha, M. C. Sarofim, J. Trtanj, & L. Ziska (Eds.), *The Impacts of Climate Change on Human Health in the United States: A Scientific Assessment* (pp. 69-98). Washington, DC: US Global Change Research Program.

Fann, N., Kim, S.-Y., Olives, C., & Sheppard, L. (2017). Estimated changes in life expectancy and adult mortality resulting from declining $PM_{2.5}$ exposures in the contiguous United States: 1980–2010. *Environmental Health Perspectives, 125*(9), 097003. <https://doi.org/10.1289/EHP507>

Fann, N., Nolte, C.G., Dolwick, P., Spero, T.L., Brown, A. C., Phillips, S., & Anenberg, S. (2015). The geographic distribution and economic value of climate change-related ozone health impacts in the United States in 2030. *Journal of the Air & Waste Management Association, 65*(5), 570–580. doi:10.1080/10962247.2014.996270

Faustman, E. M., Silbernagel, S.M., Fenske, R. A., Burbacher, T. M., & Ponce, R. A. (2000). Mechanisms underlying Children's susceptibility to environmental toxicants. *Environmental Health Perspectives, 108*(Suppl.1), 13-21. doi:10.1289/ehp.00108s113

Fiore, A. M., Naik, V., & Leibensperger, E. M. (2015). Air quality and climate connections. *Journal of the Air & Waste Management Association, 65(*6), 645 – 685. doi:10.1080/10962247.2015.1040526

Flannigan, M. D., Cantin, A. S., de Groot, W. J., Wotton, M., Newbery, A., & Gowman, L. M. (2013). Global wildland fire season severity in the 21st century. *Forest Ecology and Management, 294*, 54 – 61. doi:10.1016/j.foreco.2012.10.022

Flannigan, M. D., Krawchuk, M. A., de Groot, W. J., Wotton, B. M., & Gowman, L. M. (2009). Implications of changing climate for global wildland fire. *International Journal of Wildland Fire, 18*(5), 483-507. doi: 10.1071/WF08187

Flannigan, M. D., Logan, K. A., Amiro, B. D., Skinner, W. R., & Stocks, B. J. (2005). Future area burned in Canada. *Climatic Change, 72*, 1-16. doi:10.1007/s10584-005-5935-y

Flannigan, M. D., Wotton, M., Marshall, G., de Groot, W. J., Johnston, J., Jurko, N., & Cantin, A. S. (2016). Fuel moisture sensitivity to temperature and precipitation: Climate change implications. *Climatic Change, 134*, 59-71. doi:10.1007/s10584-015-1521-0.

Fong, K. C., Hart, J. E., & James, P. (2018). A review of epidemiologic studies on greenness and health: Updated literature through 2017. *Current Environmental Health Reports, 5*(1), 77-87. doi:10.1007/s40572-018-0179-y

Ford, B., Val Martin, M., Zelasky, S. E., Fischer, E. V., Anenberg, S. C., Heald, C. L., & Pierce, J. R. (2018). Future fire implications of smoke concentrations, visibility, and health in the contiguous United States. *GeoHealth, 2*(8), 229-247. doi: 10.1029/2018GH000144.

Forster, P., & Shine, K. P. (1997). Radiative forcing and temperature trends from stratospheric ozone changes. *Journal of Geophysical Research Atmospheres, 102,* 10841–10855. doi:10.1029/96JD03510

Fu, P., Guo, X., Cheung, F. M. H., & Yung, K. K. L. (2019). The association between $PM_{2.5}$ exposure and neurological disorders: A systematic review and meta-analysis. *The Science of the Total Environment, 655*, 1240-1248. doi: 10.1016/j.scitotenv.2018.11.218

Fuertes, E., Markevych, I., Bowatte, G., Gruzieva, O., Gehring, U., Becker, A., Berdel, D., von Berg, A., Bergstrom, A., Brauer, M., Brunekreef, B., Bruske, I., Carlesten, C., Chan-Yeung, M., Dharmage, S.C., Hoffmann, B., Klumper, C., Koppelman, G. H., Kozyrskyi, A.,...Heinrich, J. (2016). Residential greenness is differentially associated with childhood allergic rhinitis and aeroallergen sensitization in seven birth cohorts. *Allergy, 71*(10), 1461-1471. doi:10.1111/all.12915

Gai, Y., Minet, L., Posen, I.D., Smargiassi, A., Tétrault, L.-F., & Hatzopoulou, M. (2020). Health and climate benefits of electric vehicle deployment in Greater Toronto and Hamilton Area. *Environmental Pollution, 265*, 114983. doi.org/10.1016/j.envpol.2020.114983

Gao, Y., Fu, J. S., Drake, J. B., Lamarque, J.-F., & Liu, Y. (2013). The impact of emission and climate change on ozone in the United States under representative concentration pathways (RCPs). *Atmospheric Chemistry and Physics, 13*(4), 9607-9621. doi:10.5194/acp-13-9607-2013

Garcia-Menendez, F., Saari, R. K., Monier, E., & Selin, N. E. (2015). U.S. Air quality and health benefits from avoided climate change under greenhouse gas mitigation. *Environmental Science & Technology, 49*(13), 7580-7588. doi: 10.1021/acs.est.5b01324

Gauderman, J. W., McConnell, R. O., Gilliland, F., London, S., Thomas, D., Avol, E., Vora, H., Berhane, K., Rappaport, E. B., Lurmann, F., Margolis, H. G. (2000). Association between air pollution and lung function growth in southern California children. *American journal of respiratory and critical care medicine. 162*(4), 1383-90. <https://doi.org/10.1164/ajrccm.162.4.9909096>

Gershon, A. S., Guan, J., Wang, C., & To, T. (2010). Trends in asthma prevalence and incidence in Ontario, Canada, 1996-2005: A population study. *American Journal of Epidemiology, 172*(6), 728-736. doi:10.1093/aje/kwq189

Gerson, A. S., Khan, S., Klein-Geltink, J., Wilton, D., To, T., Crighton, E. J., Pigeau, L., MacQuarrie, J., Allard, Y., Russell, S. J. & Henry, D. A. (2014). Asthma and chronic obstructive pulmonary disease (COPD) prevalence and health service use in Ontario Métis: A population-based cohort study. *PLoS One, 9*(4), e95899. doi: org/10.1371/journal.pone.0095899

Gillett, N. P., Weaver, A. J., Zwiers, F. W., & Flannigan, M. D. (2004). Detecting the effect of climate change on Canadian forest fires. *Geophysical Research Letters, 31*(18), L18211. doi:10.1029/2004GL020876

Giorgi, F., & Gutowski, W. J. (2015). Regional dynamical downscaling and the CORDEX initiative. *Annual Review of Environment and Resources, 40*(1), 467–490. doi: 10.1146/annurev-environ-102014-021217

Government of Canada. (2012). Reduction of Carbon Dioxide Emissions from Coal-fired Generation of Electricity Regulations (SOR/2012-167). *Canada Gazette Part II, 146(19). Retrieved from* <http://www.gazette.gc.ca/rp-pr/p2/2012/2012-09-12/html/sor-dors167-eng.html>

Government of Canada. (2015). *Understanding Air Quality Health Index messages.* Retrieved from <https://www.canada.ca/en/environment-climate-change/services/air-quality-health-index/understanding-messages.html>

Government of Canada. (2018). Regulations Amending the Reduction of Carbon Dioxide Emissions from Coal-fired Generation of Electricity Regulations (SOR/2018-263). *Canada Gazette Part II, 152*(25). Retrieved from <http://gazette.gc.ca/rp-pr/p2/2018/2018-12-12/html/sor-dors263-eng.html>

Government of Canada. (2019a). *Air Quality Health Index.* Retrieved from <https://www.canada.ca/en/environment-climate-change/services/air-quality-health-index.html>

Government of Canada. (2019b). *How Info-Smog works.* Retrieved from <https://www.canada.ca/en/environment-climate-change/services/info-smog/how-it-works.html>

Government of Canada. (2019c). *Canada's Wildfire Smoke Prediction System (FireWork) Daily smoke forecast maps from early April to late October.* Retrieved from <https://weather.gc.ca/firework/index_e.html>

Government of Canada. (2019d). *Wildfire smoke and air quality.* Retrieved from <https://www.canada.ca/en/environment-climate-change/services/air-quality-health-index/wildfire-smoke.html>

Groot, E., Caturay, A., Khan, Y., & Copes, R. (2019). A systematic review of the health impacts of occupational exposure to wildland fires. *International Journal of Occupational Medicine and Environmental Health, 32*(2), 121-40. <https://doi.org/10.13075/ijomeh.1896.01326>

Haines, A. (2017). Health co-benefits of climate action. *The Lancet Planetary Health, 1*(1), e4-5. <https://doi.org/10.1016/S2542-5196(17)30003-7>

Hamlin, T., & Gusdorf, J. (1997). *Airtightness and energy efficiency of new conventional and R-2000 housing in Canada.* Ottawa, ON: Government of Canada. Retrieved from <http://publications.gc.ca/site/eng/73407/publication.html>

Hanes, C., Wang, X., Jain, P., Parisien, M.-A., Little, J., & Flannigan, M. (2019). Fire-regime changes in Canada over the last half century. *Canadian Journal of Forest Research, 49,* 256–269. doi:10.1139/cjfr-2018-0293.

Hartling, L., Brison, R., & Pickett, W. (1998). Cluster of Unintentional Carbon Monoxide Poisonings Presenting to the Emergency Departments in Kingston, Ontario during 'Ice Storm 98'. *Canadian Journal of Public Health, 89*(6), 388-390. <https://doi.org/10.1007/BF03404080>

Haywood, J. M., & Shine, K.P. (1995). The effect of anthropogenic *sulfate and soot aerosol on the clear sky planetary radiation budget. Geophysical Research Letters, 22*(5), 603 – 606. doi:10.1029/95GL00075

Health Canada. (2007). *Residential Indoor Air Quality Guideline: Moulds.* Ottawa, ON: Government of Canada. Retrieved from <https://www.canada.ca/content/dam/canada/health-canada/migration/healthy-canadians/publications/healthy-living-vie-saine/mould-moisissure/alt/mould-moisissures-eng.pdf>

Health Canada. (2008). *Human Health in a Changing Climate: A Canadian Assessment of Vulnerabilities and Adaptive Capacity (J. Seguin, Ed.).* Ottawa, ON: Government of Canada. Retrieved from <http://publications.gc.ca/collections/collection_2008/hc-sc/H128-1-08-528E.pdf>

Health Canada. (2013). *Canadian Smog Science Assessment - Volume 2: Health Effects.* Ottawa, ON: Government of Canada. Retrieved from <http://publications.gc.ca/collections/collection_2014/sc-hc/En88-5-2-2013-eng.pdf>

Health Canada. (2014). *A Statistical Profile on the Health of First Nations in Canada: Determinants of Health, 2006 to 2010.* Ottawa, ON: Government of Canada. Retrieved from <https://www.sac-isc.gc.ca/eng/1585414580249/1585414609942>

Health Canada. (2016). *Human Health Risk Assessment for Ambient Nitrogen Dioxide.* Ottawa, ON: Government of Canada. Retrieved from <http://publications.gc.ca/site/eng/9.816718/publication.html>

Health Canada. (2018). *Ventilation and the Indoor Environment.* Ottawa, ON: Government of Canada. Retrieved from <https://www.canada.ca/content/dam/hc-sc/documents/services/publications/healthy-living/ventilation-indoor-environment/ventilation-indoor-environment.pdf>

Health Canada. (2020). *Guidance for Cleaner Air Spaces during Wildfire Smoke Events.* Ottawa, ON: Government of Canada. Retrieved from <http://publications.gc.ca/site/eng/9.891164/publication.html>

Health Canada. (2021). *Health Impacts of Air Pollution in Canada: Estimates of morbidity and premature mortality outcomes, 2021 Report.* Ottawa, ON: Government of Canada.

Heguy, L., Garneau, M., Goldberg, M. S., Raphoz, M., Guay, F., & Valois, M. F. (2008). Associations between grass and weed pollen and emergency department visits for asthma among children in Montreal. *Environmental Research, 106*(2), 203-211. doi:S0013-9351(07)00223-X

Henderson, S. B., Brauer, M., MacNab, Y. C., & Kennedy, S. M. (2011). Three measures of forest fire smoke exposure and their associations with respiratory and cardiovascular health outcomes in a population-based cohort. *Environmental Health Perspectives, 119*(9), 1266-1271. doi:10.1289/ehp.1002288

Hernberg, S., Sripaiboonkij, P., Quansah, R., Jaakkola, J.J., & Jaakkola, M. S. (2014). Indoor molds and lung function in healthy *adults. Respiratory medicine, 108*(5), 677-684. doi:10.1016/j.rmed.2014.03.004

Heroux, M.-E., Clark, N., Van Ryswyk, K., Mallick, R., Gilbert, N., Harrison, I., Rispler, K., Wang, D., Anastassopoulos, A., Guay, M., MacNeill, M., & Wheeler, A. (2010). Predictors of indoor air concentrations in smoking and non-Smoking residences. *International Journal of Environmental Research and Public Health, 7*(8), 3080-3099. doi:10.3390/ijerph7083080

Hogrefe, C., Lynn, B., Civerolo, K., Ku, J.-Y., Rosenthal, J., Rosenzweig, C., Goldberg, R., Gaffin, S., Knowlton, K. & Kinney, P. L. (2004). Simulating changes in regional air pollution over the eastern United States due to changes in global and regional climate and emissions. *Journal of Geophysical Research: Atmospheres, 109*(22), D22301, doi:10.1029/2004JD004690

Horton, D. E., Skinner, C. B., Singh, D. & Diffenbaugh, N. S. (2014). Occurrence and persistence of future atmospheric stagnation events. *Nature Climate Change, 4*, 698-703. doi:10.1038/NCLIMATE2272, 2014.

Huang, S., Xiong, J., Cai, C., Xu, W., & Zhang, Y. (2016). Influence of humidity on the initial emittable concentration of formaldehyde and hexaldehyde in building materials: experimental observation and correlation. *Scientific Reports, 6*, 23388. doi: 10.1038/srep23388

Hulin, M., Simoni, M., Veigi, G., & Annesi-Maesano, I. (2012). Respiratory health and indoor air pollutants based on quantitative exposure assessments. *The European Respiratory Journal, 40*(4), 1033-1045. doi: 10.1183/09031936.00159011

Hystad, P. U., Setton, E. M., Allen, R. W., Keller, P. C., & Brauer, M. (2009). Modeling residential fine particulate matter infiltration for exposure assessment. *Journal of Exposure Science & Environmental Epidemiology, 19*(6), 570-579. doi:10.1038/jes.2008.45

Indigenous Services Canada (ISC). (2018). *Preventing and Managing Chronic Disease in First Nations Communities: A guidance framework.* Ottawa, ON: Indigenous Services Canada. Retrieved from <http://publications.gc.ca/collections/collection_2018/aanc-inac/H34-313-1-2017-eng.pdf>

Institute for Health Metrics and Evaluation (IHME). (2019). *Global Burden of Disease Study 2017*. Retrieved from <http://www.healthdata.org/sites/default/files/files/policy_report/2019/GBD_2017_Booklet.pdf>

Institute of Medicine (IOM). (2011). *Climate Change, the Indoor Environment, and Health*. Washington, DC: The National Academies Press.

Insurance Bureau of Canada (IBC). (2013). *Canada inundated by severe weather in 2013: Insurance companies pay out record-breaking $3.2 billion to policyholders.* Retrieved from <http://www.ibc.ca/yt/resources/media-centre/media-releases/canada-inundated-by-severe-weather-in-2013-insurance-companies-pay-out-record-breaking-$3-2-billion-to-policyholders>

Intergovernmental Panel on Climate Change (IPCC). (2014). Synthesis Report. In C.B. Field, V.R. Barros, D.J. Dokken, K. J. Mach, M. D. Mastrandrea, T. E. Bilir, M. Chatterjee, K. L. Ebi, Y. O. Estrada, R. C. Genova, B. Girma, E. S. Kissel, A. N. Levy, S. MacCracken, P. R. Mastrandrea, & L. L. White (Eds.), *Climate Change 2014* (pp.151). Contribution of Working Groups I, II and III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge, United Kingdom: Cambridge University Press.

Intergovernmental Panel on Climate Change (IPCC). (2018). Summary for Policymakers. In V. Masson-Delmotte, P. Zhai, H.-O. Pörtner, D. Roberts, J. Skea, P.R. Shukla, A. Pirani, W. Moufouma-Okia, C. Péan, R. Pidcock, S. Connors, J. B. R. Matthews, Y. Chen, X. Zhou, M.I. Gomis, E. Lonnoy, T. Maycock, M. Tignor, & T. Waterfield (Eds.), *Global Warming of 1.5°C*. Cambridge, United Kingdom: Cambridge University Press. Retrieved from <https://www.ipcc.ch/sr15/chapter/spm/>

International Agency for Research on Cancer (IARC). (2016). *Outdoor Air Pollution: IARC Monographs on the Evaluation of Carcinogenic Risks to Humans. Lyon, France. Retrieved from* <https://monographs.iarc.fr/wp-content/uploads/2018/06/mono109.pdf>

Inuit Tapiriit Kanatami (ITK). (2014). *Social Determinants of Inuit Health in Canada. Retrieved from* <https://www.itk.ca/wp-content/uploads/2016/07/ITK_Social_Determinants_Report.pdf>

Ismaila, A. S., Sayani, A. P., Marin, M., & Su, Z. (2013). Clinical, economic, and humanistic burden of asthma in Canada: A systematic review. *BMC Pulmonary Medicine*, 13, 70. doi:10.1186/1471-2466-13-70

Jacobson, M. Z. (2001). Strong radiative heating due to the mixing state of black carbon in atmospheric aerosols. *Nature, 409*(6821), 695 – 697. doi: 10.1038/35055518

Jiang, Z., McDonald, B. C., Worden, H., Worden, J. R., Miyazaki, K., Qug, Z., Henze, D. K., Jones, D. B. A., Arellano, A. F., Fischer, E. V., Zhu, L., & Boersma, K. F. (2018). Unexpected slowdown of US pollutant emission reduction in the past decade. *Proceedings of the National Academy of Sciences of the United States of America, 115*(20), 5099 – 5104. doi:10.1073/pnas.1801191115, 2018.

Johnston, F. H., Henderson, S. B., Chen, Y., Randerson, J.T., Marlier, M., DeFries, R. S., Kinney, P., Bowman, D. M. J. S., & Brauer, M. (2012). Estimated global mortality attributable to smoke from landscape fires. *Environmental Health Perspectives, 120*(5), 695-701. <https://doi.org/10.1289/ehp.1104422>

Johnston, F. H., Wheeler, A. J., Williamson, G. J., Campbell, S. L., Jones, P. J., Koolhof, I. S., Lucani, C., Cooling, N.B., & Bowman, D.M.J.S. (2018). Using smartphone technology to reduce health impacts from atmospheric environmental hazards. *Environmental Research Letters, 13*(4), 044019. doi:10.1088/1748-9326/aab1e6

Kasoar, M., Voulgarakis, A., Lamarque, J.-F., Shindell, D. T., Bellouin, N., Collins, W. J., Faluvegi, G., & Tsigaridis, K. (2016). Regional and global temperature response to anthropogenic SO_2 emissions from China in three climate models. *Atmospheric Chemistry and Physics, 16*(15), 9785 – 9804. doi:10.5194/acp-16-9785-2016

Kearney, J., Wallace, L., MacNeill, M., Héroux, M.-E., Kindzierski, W., & Wheeler, A. (2014). Residential infiltration of fine and ultrafine particles in Edmonton. *Atmospheric Environment, 94*, 793-805. doi:10.1016/j.atmosenv.2014.05.020

Keefe, A. (2014). *Evidence review: Filtration in institutional settings during wildfire smoke events (C. Elliott, & K. Rideout, Eds).* BC Centre for Disease Control. Retrieved from <http://www.bccdc.ca/resourcegallery/Documents/Guidelines%20and%20Forms/Guidelines%20and%20Manuals/Health-Environment/WFSG_EvidenceReview_FiltrationinInstitutions_FINAL_v3_edstrs.pdf>

Keith, P. K., Desrosiers, M., Laister, T., Schellenberg, R. R., & Waserman, S. (2012). The burden of allergic rhinitis (AR) in Canada: Perspectives of physicians and patients. *Allergy, Asthma, and Clinical Immunology : Official Journal of the Canadian Society of Allergy and Clinical Immunology, 8*(1), 7-1492-8-7. doi:10.1186/1710-1492-8-7

Kelly, J., Makar, P. A., & Plummer, D. A. (2012). Projections of mid-century summer air-quality for North America: effects of changes in climate and precursor emissions. *Atmospheric Chemistry and Physics, 12*(12), 5367 – 5390. doi:10.5194/acp-12-5367-2012

Kim, H., Bouchard, J., & Renzi, P. M. (2008). The link between allergic rhinitis and asthma: A role for antileukotrienes? *Canadian Respiratory Journal, 15*(2), 91-98. doi:10.1155/2008/416095

Kirchmeier-Young, M. C., Gillett, N. P., Zwiers, F. W., Cannon, A. J., & Anslow, F. S. (2019). Attribution of the influence of human-induced climate change on an extreme fire season. *Earth's Future, 7*, 2-10. <https://doi.org/10.1029/2018EF001050>

Kirchmeier-Young, M. C., Zwiers, F. W., Gillett, N. P., & Cannon, A. J. (2017). Attributing extreme fire risk in Western Canada to human emissions. *Climatic Change, 144*, 365-379. doi:10.1007/s10584-017-2030-0

Kondo, M. C., De Roos, A. J., White, L. S., Heilman, W. E., Mockrin, M. H., Gross-Davis, C. A., & Burstyn, I. (2019). Meta-analysis of heterogeneity in the effects of wildfire smoke exposure on respiratory health in North America. *International Journal of Environmental Research and Public Health, 16*(6), 960. <https://doi.org/10.3390/ijerph16060960>

Korpi, A., Pasanen, A.-L. & Pasanen, P. (1998). Volatile Compounds Originating from Mixed Microbial cultures on building materials under various humidity conditions. *Applied and Environmental Microbiology, 64*(8), 2914-2919. <https://doi.org/10.1128/AEM.64.8.2914-2919.1998>

Kovesi, T., Zaloum, C., Stocco, C., Fugler, D., Dales, R.E., Ni, A., Barrowman, N., Gilbert, N. L., & Miler, J. D. (2009). Heat recovery ventilators prevent respiratory disorders in Inuit children. *Indoor Air, 19*(6), 489-499. doi:10.1111/j.1600-0668.2009.00615.x

Kovesi, T. (2012). Respiratory disease in Canadian First Nations and Inuit children. *Paediatrics & Child Health, 17*(7), 376-380. doi:10.1093/pch/17.7.376

Lafeuille, M. H., Gravel, J., Figliomeni, M., Zhang, J., & Lefebvre, P. (2013). Burden of illness of patients with allergic asthma versus non-allergic asthma. *The Journal of Asthma : Official Journal of the Association for the Care of Asthma, 50*(8), 900-907. doi:10.3109/02770903.2013.810244

Lam, Y. F., Fu, J. S., Wu, S., & Mickley, L. J. (2011). Impacts of future climate change and effects of biogenic emissions on surface ozone and particulate matter concentrations in the United States. *Atmospheric Chemistry and Physics, 11*(10), 4789 – 4806. doi:10.5194/acp-11-4789-2011

Lavigne, E., Gasparrini, A., Stieb, D. M., Chen, H., Yasseen, A. S., 3rd, Crighton, E., To, T., Weichenthal, S., Villeneuve, P. J., Cakmak, S., Coates, F., & Walker, M. (2017). Maternal Exposure to Aeroallergens and the Risk of Early Delivery. *Epidemiology (Cambridge, Mass.)*, *28*(1), 107–115. <https://doi.org/10.1097/EDE.0000000000000573>

Lavigne, É., Burnett, R. T., Stieb, D. M., Evans, G. J., Godri Pollitt, K. J., Chen, H., van Rijswijk, D., & Weichenthal, S. (2018). Fine Particulate Air Pollution and Adverse Birth Outcomes: Effect Modification by Regional Nonvolatile Oxidative Potential. *Environmental Health Perspectives*, *126*(7), 077012. <https://doi.org/10.1289/EHP2535>

Leech, J. A., Nelson, W. C., Burnett, R. T., Aaron, S., & Raizenne, M. E. (2002). It's about time: a comparison of Canadian and American time-activity patterns. *Journal of Exposure Analysis and Environmental Epidemiology*, *12*(6), 427–432. <https://doi.org/10.1038/sj.jea.7500244>

Leech, J. A., Raizenne, M., & Gusdorf, J. (2004). Health in occupants of energy efficient new homes. *Indoor Air*, *14*(3), 169–173. <https://doi.org/10.1111/j.1600-0668.2004.00212.x>

Leibensperger, E. M., Mickley, L. J., & Jacob, D. J. (2008). Sensitivity of US air quality to mid-latitude cyclone frequency and implications of 1980–2006 climate change. *Atmospheric Chemistry and Physics*, *8*(23), 7075–7086. doi: 10.5194/acp-8-7075-2008

Liu, D., Whitehead, J., Alfarra, M. R., Reyes-Villegas, E., Spracklen, D. V., Reddington, C. L., Kong, S., Williams, P. I., Ting, Y.-C., Haslett, S., Taylor, J. W., Flynn, M. J., Morgan, W. T., McFiggans, G., Coe *H., & Allan J. D. (2017)*. Black-carbon absorption enhancement in the atmosphere determined by particle mixing state. *Nature Geoscience, 10*, 184 – 191. doi:10.1038/NGEO2901

Liu, J. C., Mickely, L. J., Sulprizio, M. P., Yue, X., Peng, R. D., Dominici, F., & Bell, M. L. (2016). Future respiratory hospital admissions from wildfire smoke under climate change in the Western US. *Environmental Research Letters, 11*(12), 124018. doi:10.1088/1748-9326/11/12/124018.

Liu, J. C., Pereira, G., Uhl, S. A., Bravo, M. A., & Bell, M. L. (2015). A systematic review of the physical health impacts from non-occupational exposure to wildfire smoke. *Environmental Research*, *136*, 120–132. <https://doi.org/10.1016/j.envres.2014.10.015>

Liu, J. C., Zheng, Y., Li, Z., Flynn, C., & Cribb, M. (2012). Seasonal variations of aerosol optical properties, vertical distribution and associated radiative effects in the Yangtze Delta region of China. *Journal of Geophysical Research: Atmospheres, 117*(D16), D00K38. doi:10.1029/2011JD016490

Lougheed, M. D., Lemière, C., Dell, S. D., Ducharme, F. M., Fitzgerald, J. M., Leigh, R., Licskai, C., Rowe, B. H., Bowie, D., Becker, A., Boulet, L. P., & Canadian Thoracic Society Asthma Committee (2010). Canadian Thoracic Society Asthma Management Continuum--2010 Consensus Summary for children six years of age and over, and adults. *Canadian Respiratory Journal*, *17*(1), 15–24. <https://doi.org/10.1155/2010/827281>

Lowe, A. J., Olsson, D., Braback, L., & Forsberg, B. (2012). Pollen exposure in pregnancy and infancy and risk of asthma hospitalisation - a register based cohort study. *Allergy, Asthma, and Clinical Immunology : Official Journal of the Canadian Society of Allergy and Clinical Immunology, 8*(1), 17. doi:10.1186/1710-1492-8-17

MacNeill, M., Kearney, J., Wallace, L., Gibson, M., Héroux, M. E., Kuchta, J., Guernsey, J. R., & Wheeler, A. J. (2014). Quantifying the contribution of ambient and indoor-generated fine particles to indoor air in residential environments. *Indoor Air*, *24*(4), 362–375. <https://doi.org/10.1111/ina.12084>

MacNeill, M., Wallace, L., Kearney, J., Allen, R. W., Van Ryswyk, K., Judek, S., Xu, X., & Wheeler, A. (2012). Factors influencing variability in the infiltration of $PM_{2.5}$ mass and its components. *Atmospheric Environment, 61,* 518-532. doi: 10.1016/j.atmosenv.2012.07.005

Maguet, S. (2018). *Public Health Responses to Wildfire Smoke Events*. BC Centre for Disease Control and National Collaborating Centre for Environmental Health. Retrieved from <https://ncceh.ca/sites/default/files/Responding%20to%20Wildfire%20Smoke%20Events%20EN.pdf>

Maidment, C. D., Jones, C. R., Webb, T.L., Hathway, E. A., & Gilbertson, J. M. (2014). The impact of household energy efficiency measures on health: A meta-analysis. *Energy Policy, 65*, 583-593. doi: 10.1016/j.enpol.2013.10.054

Manitoba Health. (2019). *Health Effects of Smoke Exposure due to Wildland Fires. Government of Manitoba*. Retrieved from <https://www.gov.mb.ca/health/publichealth/environmentalhealth/smoke.html>

Markandya, A., Sampedro, J., Smith, S. J., Van Dingenen, R., Pizarro-Irizar, C., Arto, I., & González-Eguino, M. (2018). Health co-benefits from air pollution and mitigation costs of the Paris Agreement: a modelling study. *The Lancet Planet Health, 2*(3), e126-e133. doi: 10.1016/S2542-5196(18)30029-9

Matz, C. J., Stieb, D. M., Davis, K., Egyed, M., Rose, A., Chou, B., & Brion, O. (2014). Effects of age, season, gender and urban-rural status on time-activity: Canadian Human Activity Pattern Survey 2 (CHAPS 2). *International Journal of Environmental Research and Public Health*, *11*(2), 2108–2124. <https://doi.org/10.3390/ijerph110202108>

Matz, C. J., Egyed, M., Xi, G., Racine, J., Pavlovic, R., Rittmaster, R., Henderson, S. B., & Stieb, D. M. (2020). Health impact analysis of $PM_{2.5}$ from wildfire smoke in Canada (2013-2015, 2017-2018). *The Science of the Total Environment*, *725*, 138506. <https://doi.org/10.1016/j.scitotenv.2020.138506>

McCuskee, S., Kirlew, M., Kelly, L., Fewer, S., & Kovesi, T. (2014). Bronchiolitis and pneumonia requiring hospitalization in young first nations children in Northern Ontario, Canada. *The Pediatric Infectious Disease Journal*, *33*(10), 1023–1026. <https://doi.org/10.1097/INF.0000000000000361>

McLean, K. E., Yao, J., & Henderson, S. B. (2015). An Evaluation of the British Columbia Asthma Monitoring System (BCAMS) and $PM_{2.5}$ Exposure Metrics during the 2014 Forest Fire Season. *International journal of environmental research and public health*, *12*(6), 6710–6724. <https://doi.org/10.3390/ijerph120606710>

Mearns, L. O., Sain, S., Leung, L. R., Bukovsky, M. S., McGinnis, S., Biner, S., Caya, D., Arritt, R. W., Gutowski, W., Takle, E., Snyder, M., Jones, R. G., Nunes, A. M. B., Tucker, S., Herzmann, D., McDaniel, L., & Sloan, L. (2013). Climate change projections of the North American Regional Climate Change Assessment Program (NARCCAP). *Climate Change, 120*, 965 – 975. doi:10.1007/s10584-013-0831-3.

Mehiriz, K., & Gosselin, P. (2019). Evaluation of the Impacts of a Phone Warning and Advising System for Individuals Vulnerable to Smog. Evidence from a Randomized Controlled Trial Study in Canada. *International Journal of Environmental Research and Public Health*, *16*(10), 1817. <https://doi.org/10.3390/ijerph16101817>

MetroVancouver. (2018). *Climate 2050 Strategic Framework. Retrieved from* <http://www.metrovancouver.org/climate2050>

Mickley, L. J., Jacob, D. J., Field, B. D., & Rind, D. (2004). Effects of future climate change on regional air pollution episodes in the United States. *Geophysical Research Letters, 31*(24), L24103. doi:10.1029/2004GL021216

Mills, D., Jones, R., Wobus, C., Ekstrom, J., Jantarasami, L., St Juliana, A., & Crimmins, A. (2018). Projecting Age-Stratified Risk of Exposure to Inland Flooding and Wildfire Smoke in the United States under Two Climate Scenarios. *Environmental Health Perspectives*, *126*(4), 047007. <https://doi.org/10.1289/EHP2594>

Ministère de l'Environnement et de la Lutte contre les changements climatiques. (2012). *Pan d'action 2013-2020 sur les changements climatiques*. Gouvernement du Québec. Retrieved from <http://www.environnement.gouv.qc.ca/changements/plan_action/pacc2020.pdf>

Ministère de l'Environnement et de la Lutte contre les changements climatiques. (2019). *Feux de forêt.* Gouvernement du Québec. Retrieved from <https://www.environnement.gouv.qc.ca/air/feux-foret/index.htm>

Munioz-Alpizar, R., Pavlovic, R., Moran, M.D., Chen, J., Gravel, S., Henderson, S.B., Menard, S., Racine, J., Duhamel, A., Gilbert, S., Beaulieu, P.-A., Davignon, D., Cousineau, S., & Bouchet, V. (2017). Multi-year (2013-2016) $PM_{2.5}$ wildfire pollution exposure over North America as determined from operation air quality forecasts. *Atmosphere, 8*(9), 179. doi: 10.3390/atmos8090179

Myhre, G., Shindell, D., Bréon, F.-M., Collins, W., Fuglestvedt, J., Huang, J., Koch, D., Lamarque, J.-F., Lee, D., Mendoza, D., Nakajima, T., Robock, A., Stephens, G., Takemura, T., & Zhang, H. (2013). Anthropogenic and Natural Radiative Forcing. *In* T. F. Stocker, D. Qin, G.-K. Plattner, M. Tignor, S. K. Allen, J. Boschung, A. Nauels, Y. Xia, V. Bex & P. M. Midgley, Eds., *Climate Change 2013: The Physical Science Basis (pp. 659-740).* Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge, United Kingdom and New York, NY, USA: Cambridge University Press. doi:10.1017/CBO9781107415324.018

Naeher, L. P., Brauer, M., Lipsett, M., Zelikoff, J. T., Simpson, C. D., Koenig, J. Q., & Smith, K. R. (2007). Woodsmoke health effects: a review. *Inhalation Toxicology*, *19*(1), 67–106. <https://doi.org/10.1080/08958370600985875>

Nakicenovic, N., Alcamo, J., Davis, G., de Vries, B., Fenhann, J., Gaffin, S., Gregory, K., Grübler, A., Jung, T.Y., Kram, T., La Rovere, E.L., Michaelis, L., Mori, S., Morita, T., Pepper, W., Pitcher, H., Price, L., Riahi, K., Roehrl, A., Rogner, H.-H., Sankovski, A., Schlesinger, M., Shukla, P., Smith, S., Swart, R., van Rooijen, S., Victor, N., & Dadi, Z. (2000). Special report on emissions scenarios. In N. Nakicenovic, & R. Swart, Eds., *Emissions Scenarios (pp. 599). Contribution of Working Group III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change.* Cambridge, United Kingdom and New York, NY, USA: Cambridge University Press.

National Collaborating Centre for Aboriginal Health (NCCAH). *(2017). Housing as a Social Determinant of First Nations, Inuit, and Métis Health.* Prince George, BC. Retrieved from <https://www.ccnsa-nccah.ca/docs/determinants/FS-Housing-SDOH2017-EN.pdf>

Natural Resources Canada (NRCan). (2016). *Energy Efficiency Trends in Canada.* Ottawa, ON: Natural Resources Canada, Office of Energy Efficiency.

Natural Resources Canada (NRCan). (2018). *Energy and Greenhouse Gas Emissions (GHGs). Retrieved from* <https://www.nrcan.gc.ca/energy/facts/energy-ghgs/20063>

Nazaroff, W., & Weschler, C. (2004). Cleaning products and air fresheners: Exposure to primary and secondary air pollutants. *Atmospheric Environment, 38*(18), 2841-2865. doi:10.1016/j.atmosenv.2004.02.040

Nolte, C. G., Gilliland, A. B., Hogrefe, C., & Mickley, L. J. (2008). Linking global to regional models to assess future climate impacts on surface ozone levels in the United States. *Journal of Geophysical Research: Atmospheres, 113*(D14), D14307. doi:10.1029/2007JD008497

Northwest Territories Health and Social Services (NT HSS). (2016a). *Wildfire Smoke and Your Health Air quality Self-assessment Method. Government of Northwest Territories.* Retrieved from <https://www.hss.gov.nt.ca/sites/hss/files/wildfire-smoke-health.pdf>

Northwest Territories Health and Social Services (NT HSS). (2016b). *Smoke Exposure from Wildfire: Guidelines for Protecting Community Health and Wellbeing. Government of the Northwest Territories.* Retrieved from <https://www.hss.gov.nt.ca/sites/hss/files/smoke-exposure-wildfire-guidelines.pdf>

Northwest Territories Health and Social Services (NT HSS). (2018). *Wildfire smoke information in the Northwest Territories. Government of the Northwest Territories.* Retrieved from <https://www.hss.gov.nt.ca/sites/hss/files/resources/smoke-warning-aug2018.pdf>

Nova Scotia Department of Health and Wellness. (2018). *Emergencies and Extreme Weather Events – Wildfires. Province of Nova Scotia.* Retrieved from <https://novascotia.ca/dhw/environmental/wildfires.asp>

Oakes, M., Baxter, L., & Long, T. C. (2014). Evaluating the application of multipollutant exposure metrics in air pollution health studies. *Environment International*, *69*, 90–99. <https://doi.org/10.1016/j.envint.2014.03.030>

Ontario Ministry of Environment Conservation and Parks (OMECP). (2010). *AQHI categories and health messages. Government of Ontario*. Retrieved from <http://www.airqualityontario.com/aqhi/health_messages.php>

Ontario Ministry of Health and Long term Care (OHLTC). (n.d.). *Emergency Management Forest Fire Smoke and Your Health. Government of Ontario.* Retrieved from <http://www.health.gov.on.ca/en/public/programs/emu/fire_mn.aspx>

Parekh, A., Roux, L., & Gallant, P. (2007, July). *Thermal and air leakage characteristics of Canadian housing [Conference session]*. The 11th Canadian Conference on Building Science and Technology, Banff, Alberta, Canada.

Park, S., Allen, R.J., & Lim, C. H. (2020). A likely increase in fine particulate matter and premature mortality under future climate change. *Air Quality, Atmosphere & Health, 13*(7), 143-151. doi.org/10.1007/s11869-019-00785-7

Partanen, A.-I., Landry, J.-S., & Matthews, H. D. (2018). Climate and health implications of future aerosol emission scenarios. *Environmental Research Letters, 13(*2), 024028. doi:10.1088/1748-9326/aaa511

Pavlovic, R., Chen, J., Anderson, K., Moran, M. D., Beaulieu, P. A., Davignon, D., & Cousineau, S. (2016). The FireWork air quality forecast system with near-real-time biomass burning emissions: Recent developments and evaluation of performance for the 2015 North American wildfire season. *Journal of the Air & Waste Management Association, 66(*9), 819-41. doi:10.1080/10962247.2016.1158214

Peters, D. R., Schnell, J. L., Kinney, P. L., Naik, V., & Horton, D. E. (2020). Public health and climate benefits and trade-offs of U.S. vehicle electrification. *GeoHealth, 4*(10), e2020GH000275. <https://doi.org/10.1029/2020GH000275>

Pfister, G. G., Walters, S., Lamarque, J.-F., Fast, J., Barth, M. C., Wong, J., Done, J., Holland, G. & Bruyère, C. L. (2014). Projections of future summertime ozone over the U.S. *Journal of Geophysical Research: Atmospheres, 119*(9), 5559–5582. doi:10.1002/2013JD020932

Pinault, L., Crouse, D., Jerrett, M., Brauer, M., & Tjepkema, M. (2016a). Spatial associations between socioeconomic groups and NO_2 air pollution exposure within three large Canadian cities. *Environmental Research*, *147*, 373–382. <https://doi.org/10.1016/j.envres.2016.02.033>

Pinault, L., Crouse, D., Jerrett, M., Brauer, M., & Tjepkema, M. (2016b). Socioeconomic differences in nitrogen dioxide ambient air pollution exposure among children in the three largest Canadian cities. *Health Reports*, *27*(7), 3–9.

Pinault, L., van Donkelaar, A., & Martin, R. V. (2017). Exposure to fine particulate matter air pollution in Canada. *Health Reports*, *28*(3), 9–16.

Potera, C. (2011). Climate change impacts indoor environment. *Environmental health perspectives*, *119*(9), a382. <https://doi.org/10.1289/ehp.119-a382>

Poulin, P., Levasseur, M.-E., & Huppé, V. (2016). *Mesures d'adaptation pour une saine qualité de l'air intérieur dans un contexte de changements climatiques : revue de la littérature.* Institut National de Santé Publique du Québec. Retrieved from <https://www.inspq.qc.ca/publications/2194>

Prather, M. J., Holmes, C. D., & Hsu, J. (2012). Reactive greenhouse gas scenarios: Systematic exploration of uncertainties and the role of atmospheric chemistry. *Geophysical Research Letters, 39*(9), L09803. doi:10.1029/2012GL051440

Public Health Agency of Canada (PHAC). (2018). *Key Health Inequalities in Canada: A National Portrait.* Ottawa, ON. Retrieved from <https://www.canada.ca/content/dam/phac-aspc/documents/services/publications/science-research/key-health-inequalities-canada-national-portrait-executive-summary/hir-full-report-eng.pdf>

Radisic, S., Newbold, K. B., Eyles, J., & Williams, A. (2016). Factors influencing health behaviours in response to the air quality health index: A cross-sectional study in Hamilton, Canada. *Environmental Health Review, 59*(1), 17-29. doi: 10.5864/d2016-002

Ramanathan, R., Chung, C., Kim, D., Bettge, T., Buja, L., Kiehl, J. T., Washington, W. M., Fu, Q., Sikka, D. R., & Wild, M. (2005). Atmospheric brown clouds: Impacts on South Asian climate and hydrological cycle. *Proceedings of the National Academy of Sciences of the United States of America, 102*(15), 5326 – 5333. doi: 10.1073/pnas.0500656102

Ravishankara, A. R., Dawson, J. P., & Winner, D. A. (2012). New Directions: Adapting air quality management to climate change: A must for planning. *Atmospheric Environment, 50*, 387 – 389. doi:10.1016/j.atmosenv.2011.12.048

Reading, C., & Halseth, R. (2013). *Pathways to improving well-being for Indigenous peoples: how living conditions decide health.* Prince George, BC: National Collaborating Centre for Aboriginal Health. Retrieved from <https://www.ccnsa-nccah.ca/docs/determinants/RPT-PathwaysWellBeing-Reading-Halseth-EN.pdf>

Reid, C. E., Brauer, M., Johnston, F. H., Jerrett, M., Balmes, J. R., & Elliott, C. T. (2016). Critical Review of Health Impacts of Wildfire Smoke Exposure. *Environmental Health Perspectives*, *124*(9), 1334–1343. <https://doi.org/10.1289/ehp.1409277>

Reid, C. E., & Gamble, J. L. (2009). Aeroallergens, allergic disease, and climate change: impacts and adaptation. *EcoHealth*, *6*(3), 458–470. <https://doi.org/10.1007/s10393-009-0261-x>

Reisen, F., Duran, S. M., Flannigan, M. D., Elliott, C., & Rideout, K. (2015). Wildfire smoke and public health risk. *International Journal of Wildland Fire, 24*(8), 1029-1044. doi: 10.1071/WF15034

Rice, M. B., Thurston, G. D., Balmes, J. R., & Pinkerton, K. E. (2014). Climate change. A global threat to cardiopulmonary health. *American Journal of Respiratory and Critical Care Medicine*, *189*(5), 512–519. <https://doi.org/10.1164/rccm.201310-1924PP>

Rittmaster, R., Adamowicz, W. L., Amiro, B., & Pelletier, R. T. (2006). Economic analysis of health effects from forest fires. *Canadian Journal of Forest Research, 36*(4), 868-877. doi: 10.1139/x05-293

Rittmaster, R., Adamowicz, W. L., Amiro, B., & Pelletier, R. T. (2008). Erratum: Economic analysis of health effects from forest fires. *Canadian Journal of Forest Research, 38*(4), 908. doi:10.1139/X08-905

Romero-Lankao, P., Smith, J. B., Davidson, D. J., Diffenbaugh, N. S., Kinney, P. L., Kirshen, P., Kovacs, P., & Villers Ruiz, L. (2014). North America. In V. R. Barros, C. B. Field, D. J. Dokken, M. D. Mastrandrea, K. J. Mach, T. E. Bilir, M. Chatterjee, K. L. Ebi, Y. O. Estrada, R. C. Genova, B. Girma, E. S. Kissel, A. N. Levy, S. MacCracken, P. R. Mastrandrea, & L. L.White (Eds.), *Climate Change 2014: Impacts, Adaptation, and Vulnerability. Part B: Regional Aspects. Contribution of Working Group II to the Fifth Assessment, Report of the Intergovernmental Panel on Climate Change.* Cambridge, United Kingdom and New York, NY, USA: Cambridge University Press.

Romps, D. M., Seeley, J. T., Vollaro, D., & Molinari, J. (2014). Projected increase in lightning strikes in the United States due to global warming. *Science, 346*(6211), 851–854. doi: 10.1126/science.1259100

Saari, R. K., Selin, N. E., Rausch, S., & Thompson, T. M. (2015). A self-consistent method to assess air quality co-benefits from U.S. climate policies. *Journal of the Air and Waste Management Association (1995)*, *65*(1), 74–89. <https://doi.org/10.1080/10962247.2014.959139>

Saari, R. K., Mei, Y., Monier, E., & Garcia-Menendez, F. (2019). Effect of health-related uncertainty and natural variability on health impacts and cobenefits of climate policy. *Environmental science and technology*, *53*(3), 1098–1108. <https://doi.org/10.1021/acs.est.8b05094>

Santé Montréal. (2017). *Info-smog.* City of Québec. Retrieved from <https://santemontreal.qc.ca/en/public/advice-and-prevention/info-smog/>

Saskatchewan Environment Public Health and Safety. (n.d.). *Wildfire Smoke and Air Quality.* Government of Saskatchewan. Retrieved from <https://www.saskatchewan.ca/residents/environment-public-health-and-safety/wildfire-in-saskatchewan/fire-smoke-and-air-quality>

Saunois, M., Bousquet, P., Poulter, B., Peregon, A., Ciais, P., Canadell, J. G., Dlugokencky, E. J., Etiope, G., Bastviken, D., Houweling, S., Janssens-Maenhout, G., Tubiello, F. N., Castaldi, S., Jackson, R. B., Alexe, M., Arora, V. K., Beerling, D. J., Bergamaschi, P., Blake, D. R., Brailsford, G., …Zhu, Q. (2016). The global methane budget 2000 – 2012. *Earth System Science Data Discussions, 8*(2), 697–751. doi:10.5194/essd-8-697-2016

Schnell, J. L., & Prather, M. (2017). Co-occurrence of extremes in surface ozone, particulate matter, and temperature over eastern North America. *Proceedings of the National Academy of Sciences of the United States of America, 114*(11), 2854 – 2859. doi:10.1073/pnas.1614453114, 2017.

Scout Environmental. (2019). Air Health Check. Retrieved from <https://scoutenvironmental.com/air-health-check/>

Seguel, J. M., Merrill, R., Seguel, D., & Campagna, A. C. (2016). Indoor Air Quality. *American journal of lifestyle medicine*, *11*(4), 284–295. <https://doi.org/10.1177/1559827616653343>

Seinfeld, J. H., Bretherton, C., Carslaw, K. S., Coe, H., DeMott, P. J., Dunlea, E. J., Feingold, G., Ghan, S., Guenther, A. B., Kahn, R., Kraucunas, I., Kreidenweis, S. M., Molina, M. J., Nenes, A., Penner, J. E., Prather, K. A., Ramanathan, V., …Wood, R. (2016). Improving our fundamental understanding of the role of aerosol−cloud interactions in the climate system. *Proceedings of the National Academy of Sciences of the United States of America, 113*(21), 5781 – 5790. doi:10.1073/pnas.1514043113

Selin, N. E., Wu, S., Nam, K. M., Reilly, J. M., Paltsev, S., Prinn, R. G., & Webster, M. D. (2009). Global health and economic impacts of future ozone pollution. *Environmental Research Letters, 4*(4), 044014. doi:10.1088/1748-9326/4/4/044014

Sharpe, R. A., Thornton, C. R., Nikolaou, V., & Osborne, N. J. (2015). Higher energy efficient homes are associated with increased risk of doctor diagnosed asthma in a UK subpopulation. *Environment International*, *75*, 234–244. <https://doi.org/10.1016/j.envint.2014.11.017>

Shea, K. M., Truckner, R. T., Weber, R. W., & Peden, D. B. (2008). Climate change and allergic disease. *The Journal of Allergy and Clinical Immunology*, *122*(3), 443–455. <https://doi.org/10.1016/j.jaci.2008.06.032>

Shindell, D. T., Faluvegi, G., Rotstayn, L., & Milly, G. (2015). Spatial patterns of radiative forcing and surface temperature response. *Journal of Geophysical Research: Atmospheres, 120(11),* 5385–5403. doi:10.1002/2014JD022752

Shindell, D. T., Faluvegi, G., Seltzer, K., & Shindell C. (2018). Quantified, localized health benefits of accelerated carbon dioxide emission reductions. *Nature Climate Change, 8*(4), 291-295. doi:10.1038/s41558-018-0108-y

Shindell, D. T., Kuylenstierna, J. C. I., Vignati, E., van Dingenen, R., Amann, M., Klimont, Z., Anenberg, S. C., Muller, N., Janssens-Maenhout, G., Raes, F., Schwartz, J., Faluvegi, G., Pozzoli, L., Kupiainen, K., Höglund-Isaksson, L., Emberson, L., Streets, D., Ramanathan, V., ….Fowler, D. (2012). Simultaneously mitigating near-term climate change and improving human health and food security. *Science, 335*(6065), 183-189. doi:10.1126/science.1210026

Shindell, D. T., Lee, Y., & Faluvegi, G. (2016). Climate and health impacts of US emissions reductions consistent with 2ºC. *Nature Climate Change, 6,* 503-507. doi: 10.1038/nclimate2935

Sierra-Heredia, C., North, M., Brook, J., Daly, C., Ellis, A. K., Henderson, D., Henderson, S. B., Lavigne, É., & Takaro, T. K. (2018). Aeroallergens in Canada: Distribution, Public Health Impacts, and Opportunities for Prevention. *International Journal of Environmental Research and Public Health*, *15*(8), 1577. <https://doi.org/10.3390/ijerph15081577>

Silva, R. A., West, J. J., Lamarque, J. F., Shindell, D. T., Collins, W. J., Dalsoren, S., Faluvegi, G., Folberth, G., Horowitz, L. W., Nagashima, T., Naik, V., Rumbold, S. T., Sudo, K., Takemura, T., Bergmann, D., Cameron-Smith, P., Cionni, I., Doherty, R. M., Eyring, V., Josse, B., … Zeng, G. (2016). The effect of future ambient air pollution on human premature mortality to 2100 using output from the ACCMIP model ensemble. *Atmospheric Chemistry and Physics*, *16*(15), 9847–9862. <https://doi.org/10.5194/acp-16-9847-2016>

Silva, R. A., West, J. J., Lamarque, J. F., Shindell, D. T., Collins, W. J., Faluvegi, G., Folberth, G. A., Horowitz, L. W., Nagashima, T., Naik, V., Rumbold, S. T., Sudo, K., Takemura, T., Bergmann, D., Cameron-Smith, P., Doherty, R. M., Josse, B., MacKenzie, I. A., Stevenson, D. S., & Zeng, G. (2017). FUTURE GLOBAL MORTALITY FROM CHANGES IN AIR POLLUTION ATTRIBUTABLE TO CLIMATE CHANGE. *Nature climate change*, *7*(9), 647–651. <https://doi.org/10.1038/nclimate3354>

Singh, R., Walsh, P., & Mazza, C. (2019). Sustainable Housing: Understanding the Barriers to Adopting Net Zero Energy Homes in Ontario, Canada. *Sustainability, 11*(22), 6236-6257. doi:10.3390/su11226236

Skiles, S. M., Flanner, M., Cook, J. M., Dumont, M., & Painter, T. H. (2018). Radiative forcing by light-absorbing particles in snow: *Nature Climate Change, 8*(11), 964 – 971. doi:10.1038/s41558-018-0296-5

Smith, K. R., Woodward, A., Campbell-Lendrum, D., Chadee, D. D., Honda, Y., Liu, Q., Olwoch, J. M., Revich, B., & Sauerborn, R. (2014). Human health: impacts, adaptation, and co-benefits. In C. B. Field, V. R. Barros, D. J. Dokken, K. J. Mach, M. D. Mastrandrea, T. E. Bilir, M. Chatterjee, K. L. Ebi, Y. O. Estrada, R. C. Genova, B. Girma, E. S. Kissel, A. N. Levy, S. MacCracken, P. R. Mastrandrea, & L. L. White (Eds.), *Climate Change 2014: Impacts, adaptation, and vulnerability. Part A: Global and sectoral aspects (pp. 709-754). Contribution of Working Groups I, II and III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change.* Cambridge, United Kingdom: Cambridge University Press.

Solomon, G. M., Hjelmroos-Koski, M., Rotkin-Ellman, M., & Hammond, S. K. (2006). Airborne mold and endotoxin concentrations in New Orleans, Louisiana, after flooding, October through November 2005. *Environmental Health Perspectives*, *114*(9), 1381–1386. <https://doi.org/10.1289/ehp.9198>

Spurr, K., Pendergast, N., & MacDonald, S. (2014). Assessing the use of the Air Quality Health Index by vulnerable populations in a 'low-risk' region: A pilot study. *Canadian Journal of Respiratory Therapy : CJRT = Revue Canadienne de la Therapie Respiratoire : RCTR*, *50*(2), 45–49.

Stach, A., García-Mozo, H., Prieto-Baena, J. C., Czarnecka-Operacz, M., Jenerowicz, D., Silny, W., & Galán, C. (2007). Prevalence of Artemisia species pollinosis in western Poland: impact of climate change on aerobiological trends, 1995-2004. *Journal of Investigational Allergology & Clinical Immunology*, *17*(1), 39–47

Statistics Canada. (2015). *Aboriginal Peoples: Fact Sheet for Canada*. Retrieved from <https://www150.statcan.gc.ca/n1/pub/89-656-x/89-656-x2015001-eng.htm>

Statistics Canada. (2017). *The housing conditions of Aboriginal people in Canada Census of Population 2016.* Ottawa ON: Government of Canada. Retrieved from <https://www12.statcan.gc.ca/census-recensement/2016/as-sa/98-200-x/2016021/98-200-x2016021-eng.pdf>

Steiner, A. L., Tonse, S., Cohen, R. C., Goldstein, A. H., & Harley, R. A. (2006). Influence of future climate and emissions on regional air quality in California. *Journal of Geophysical Research: Atmospheres, 111*(D18), D18303. doi:10.1029/2005JD006935

Stevenson, D. S., Young, P. J., Naik, V., Lamarque, J.-F., Shindell, D. T., Voulgarakis, A., Skeie, R. B., Dalsoren, S. B., Myhre, G., Berntsen, T. K., Folberth, G. A., Rumbold, S. T., Collins, W. J., MacKenzie, I. A., Doherty, R. M., Zeng, G., van Noije, T. P. C., Strunk, A., Bergmann, D., Cameron-Smith, P., …Archibald, A. (2013). Tropospheric ozone changes, radiative forcing and attribution to emissions in the Atmospheric Chemistry and Climate Model Intercomparison Project (ACCMIP). *Atmospheric Chemistry and Physics, 13*(6), 3063-3085. doi:10.5194/acp-13-3063-2013

Stieb, D. M., Burnett, R. T., Smith-Doiron, M., Brion, O., Shin, H. H., & Economou, V. (2008). A new multipollutant, no-threshold air quality health index based on short-term associations observed in daily time-series analyses. *Journal of the Air & Waste Management Association (1995)*, *58*(3), 435–450. <https://doi.org/10.3155/1047-3289.58.3.435>

Stieb, D. M., Chen, L., Beckerman, B. S., Jerrett, M., Crouse, D. L., Omariba, D. W., Peters, P. A., van Donkelaar, A., Martin, R. V., Burnett, R. T., Gilbert, N. L., Tjepkema, M., Liu, S., & Dugandzic, R. M. (2016). Associations of Pregnancy Outcomes and $PM_{2.5}$ in a National Canadian Study. *Environmental Health Perspectives*, *124*(2), 243–249. <https://doi.org/10.1289/ehp.1408995>

Stieb, D. M., Judek, S., van Donkelaar, A., Martin, R. V., Brand, K., Shin, H. H., Burnett, R. T., & Smith-Doiron, M. H. (2015). Estimated public health impacts of changes in concentrations of fine particle air pollution in Canada, 2000 to 2011. *Canadian Journal of Public Health = Revue Canadienne de Sante Publique*, *106*(6), e362–e368. <https://doi.org/10.17269/cjph.106.4983>

Stieb, D. M., Yao, J., Henderson, S. B., Pinault, L., Smith-Doiron, M. H., Robichaud, A., van Donkelaar, A., Martin, R. V., Ménard, R., & Brook, J. R. (2019). Variability in ambient ozone and fine particle concentrations and population susceptibility among Canadian health regions. *Canadian Journal of Public Health = Revue Canadienne de Sante Publique*, *110*(2), 149–158. <https://doi.org/10.17269/s41997-018-0169-8>

Stohl, A., Aamaas, B., Amann, M., Baker, L.H., Bellouin, N., Berntsen, T.K., Boucher, O., Cherian, R., Collins, W., Daskalakis, N., Dusinska, M., Eckhardt, S., Fuglestvedt, J. S., Harju, M., Heyes, C., Hodnebrog, Ø., Hao, J., Im, U., Kanakidou, M., Klimont, Z., Kupiainen, K., …. Zhu, T. (2015). Evaluating the climate and air quality impacts of short-lived pollutants. *Atmospheric Chemistry and Physics, 15*(18), 10529 – 10566 doi:10.5194/acp-15-10529-2015, 2015.

Stowell, J. D., Kim, Y.-M., Gao, Y., Fu, J. S., Chang, H. H., & Liu, Y. (2017). The impact of climate change and emissions control on future ozone levels: Implications for human health. *Environmental International, 108*, p. 41-50. doi:10.1016/j.envint.2017.08.001

Tagaris, E., Liao, K.-J., DeLucia, A. J., Deck, L., Amar, P. & Russell, A. G. (2009). Potential impact of climate change on air pollution-related human health effects. *Environmental Science and Technology, 43*(13), 4979–4988. doi: 10.1021/es803650w

Tagaris, E., Manomaiphiboon, K., Liao, K.-J., Leung, L. R., Woo, J.-H., He, S., Amar, P., & Russell, A. G. (2007). Impacts of global climate change and emissions on regional ozone and fine particulate matter concentrations over the United States. *Journal Geophysical Research: Atmospheres, 112*(D14), D14312. doi:10.1029/2006JD008262.

Tai, A. P. K., Mickley, L. J., & Jacob, D. J. (2010). Correlations between fine particulate matter ($PM_{2.5}$) and meteorological *variables in the United States: Implications for the sensitivity of* $PM_{2.5}$ *to climate change. Atmospheric Environment, 44*(32), 3976-3984. doi:10.1016/j.atmosenv.2010.06.060

Tai, A. P. K., Mickley, L. J., & Jacob, D. J. (2012). Impact of 2000–2050 climate change on fine particulate matter ($PM_{2.5}$) air quality inferred from a multi-model analysis of meteorological modes. *Atmospheric Chemistry and Physics, 12*(23), 11329-11337. doi:10.5194/acp-12-11329-2012

Takaro, T. K., Knowlton, K., & Balmes, J. R. (2013). Climate change and respiratory health: current evidence and knowledge gaps. *Expert Review of Respiratory Medicine*, *7*(4), 349–361. <https://doi.org/10.1586/17476348.2013.814367>

Tan, X., Chen, S., Gan, T. Y., Liu, B., & Chen, X. (2019). Dynamic and thermodynamic changes conducive to the increased occurrence of extreme fire weather over western Canada under possible anthropogenic climate change. *Agricultural and Forest Meteorology, 265*, 269-279. doi:10.1016/j.agrformet.2018.11.026

Taylor, P. E., Jacobson, K. W., House, J. M., & Glovsky, M. M. (2007). Links between pollen, atopy and the asthma epidemic. *International Archives of Allergy and Immunology*, *144*(2), 162–170. <https://doi.org/10.1159/000103230>

The Weather Network. (2019). *Pollen report list. Retrieved from* <https://www.theweathernetwork.com/forecasts/pollen/list>

Thompson, T. M., Rausch, S., Saari, R. K., & Selin, N. E. (2014). A systems approach to evaluating the air quality co-benefits of US carbon policies. *Nature Climate Change, 4,* 917-923. doi: 10.1038/nclimate2342

Thompson, T. M., Rausch, S., Saari, R. K., & Selin, N. E. (2016). Air quality co-benefits of subnational carbon policies. *Journal of the Air & Waste Management Association (1995)*, *66*(10), 988–1002. <https://doi.org/10.1080/10962247.2016.1192071>

To, T., Dell, S., Tassoudji, M., & Wang, C. (2009). Health outcomes in low-income children with current asthma in Canada. *Chronic Diseases in Canada*, *29*(2), 49–55.

Traidl-Hoffmann, C., Kasche, A., Menzel, A., Jakob, T., Thiel, M., Ring, J., & Behrendt, H. (2003). Impact of pollen on human health: more than allergen carriers? *International Archives of Allergy and Immunology*, *131*(1), 1–13. <https://doi.org/10.1159/000070428>

Trail, M., Tsimpidi, A. P., Liu, P., Tsigaridis, K., Rudokas, J., Miller, P., Nenes, A., Hu, Y., & Russell, A. G. (2014). Sensitivity of air quality to potential future climate change and emissions in the United States and major cities. *Atmospheric Environment, 94,* 552 – 563, doi:10.1016/j.atmosenv.2014.05.079

Turner, M. C., Jerrett, M., Pope, C. A., 3rd, Krewski, D., Gapstur, S. M., Diver, W. R., Beckerman, B. S., Marshall, J. D., Su, J., Crouse, D. L., & Burnett, R. T. (2016). Long-Term Ozone Exposure and Mortality in a Large Prospective Study. *American Journal of Respiratory and Critical Care Medicine*, *193*(10), 1134–1142. <https://doi.org/10.1164/rccm.201508-1633OC>

Turnock, S. T., Smith, S., & O'Connor, F. M. (2019). The impact of climate mitigation measures on near term climate forcers. *Environmental Research Letters, 14*(10), 104013. doi: 10.1088/1748-9326/ab4222

Twohig-Bennett, C., & Jones, A. (2018). The health benefits of the great outdoors: A systematic review and meta-analysis of greenspace exposure and health outcomes. *Environmental Research*, *166*, 628–637. <https://doi.org/10.1016/j.envres.2018.06.030>

United National Framework Convention on Climate Change (UNFCCC). (2015). *The Paris Agreement.* Retrieved from <https://unfccc.int/process-and-meetings/the-paris-agreement/the-paris-agreement>

United States Centers for Disease Control and Prevention (US CDC). (2020). *COVID-19 considerations for cleaner air shelters and cleaner air spaces to protect the public from wildfire Smoke.* Atlanta, GA. Retrieved from <https://www.cdc.gov/coronavirus/2019-ncov/php/cleaner-air-shelters.html>

United States Environmental Protection Agency (US EPA). (2016). *Wildfire Smoke: A guide for public health officials.* Retrieved from <https://www.airnow.gov/publications/wildfire-smoke-guide/wildfire-smoke-a-guide-for-public-health-officials/>

United States Environmental Protection Agency (US EPA). (2019). *Integrated Science Assessment for Particulate Matter.* Washington, DC. Retrieved from <https://epa.gov/isa/integrated-science-assessment-particulate-matter>

United States Environmental Protection Agency (US EPA). (2020a). *Integrated Science Assessment for Ozone and Related Photochemical Oxidants*. Washington, DC. Retrieved from <https://cfpub.epa.gov/ncea/isa/recordisplay.cfm?deid=348522>

United States Environmental Protection Agency (US EPA). (2020b). *Integrated Science Assessments (ISAs)*. Retrieved from <www.epa.gov/isa>

Vachon, J., Gallant, V., & Siu, W. (2018). Tuberculosis in Canada, 2016. *Canada Communicable Disease Report = Releve des Maladies Transmissibles au Canada*, *44*(3-4), 75–81. <https://doi.org/10.14745/ccdr.v44i34a01>

Vaitla, P. M., & Drewe, E. (2011). Identifying the culprit allergen in seasonal allergic rhinitis. *The Practitioner*, *255*(1740), 27–2.

Val Martin, M., Heald, C. L., Lamarque, J.-F., Tilmes, S., Emmons, L. K., & Schichtel, B. A. (2015). How emissions, climate, and land use change will impact mid-century air quality over the United States: a focus on effects at national parks. *Atmospheric Chemistry and Physics, 15*(5), 2805–2823. doi:10.5194/acp-15-2805-2015

Vandyck, T., Keramidas, K., Kitous, A., Spadaro, J. V., Van Dingenen, R., Holland, M., & Saveyn, B. (2018). Air quality co-benefits for human health and agriculture counterbalance costs to meet Paris Agreement pledges. *Nature Communications*, *9*(1), 4939. <https://doi.org/10.1038/s41467-018-06885-9>

Van Vuuren, D. P., & Carter, T. R. (2014). Climate and socio-economic scenarios for climate change research and assessment: reconciling the new with the old. *Climate change, 122*(3), 415-429. doi:10.1007/s10584-013-0974-2

Van Vuuren, D.P., Edmonds, J., Kainuma, M., Riahi, K., Thomson, A., Hibbard, K., Hurtt, G. C., Kram, T., Krey, V., Lamarque, J., Masui, T., Meinshausen, M., Nakicenovic, N., Smith, S. J., & Rose, S. K. (2011). The representative concentration pathways: an overview. *Climatic Change, 109*(1), 5-31. doi:10.1007/s10584-011-0148-z

Viegi, G., Maio, S., Simoni, M., & Annesi-Maesano, I. (2009). The epidemiological link between ageing and respiratory diseases. In V. Bellia and R. Incalzi (Eds.), Respiratory Diseases in Elderly (p. 1–17).

Wallace, L. (1996). Indoor particles: a review. *Journal of the Air & Waste Management Association (1995), 46*(2), 98–126. <https://doi.org/10.1080/10473289.1996.10467451>

Wang, X., Parisien, M.-A., Taylor, S. W., Candau, J.-N., Stralberg, D., Marshall, G. A., Little, J. M., & Flannigan, M. D. (2017). Projected changes in daily fire spread across Canada over the next century. *Environmental Research Letters, 12*(2), 025005. doi:10.1088/1748-9326/aa5835

Warren, F. J., & Lemmen, D. S. (Eds.). (2014). Canada in a Changing Climate: Sector Perspectives on Impacts and Adaptation. Ottawa, ON: Government of Canada.

Weichenthal, S., Lavigne, E., Villeneuve, P. J., & Reeves, F. (2016). Airborne Pollen Concentrations and Emergency Room Visits for Myocardial Infarction: A Multicity Case-Crossover Study in Ontario, Canada. *American Journal of Epidemiology, 183*(7), 613–621. <https://doi.org/10.1093/aje/kwv252>

Wen, X. J., Balluz, L., & Mokdad, A. (2009). Association between media alerts of air quality index and change of outdoor activity among adult asthma in six states, BRFSS, 2005. *Journal of Community Health, 34*(1), 40–46. <https://doi.org/10.1007/s10900-008-9126-4>

Weschler, C. J. (2006). Ozone's impact on public health: contributions from indoor exposures to ozone and products of ozone-initiated chemistry. *Environmental Health Perspectives, 114*(10), 1489–1496. <https://doi.org/10.1289/ehp.9256>

Weschler, C.J. (2009). Changes in indoor pollutants since the 1950s. *Atmospheric Environment, 43*(1), 153–169. doi: 10.1016/j.atmosenv.2008.09.044

West, J. J., Fiore, A. M., & Horowitz, L. W. (2012). Scenarios of methane emission reductions to 2030: abatement costs and co-benefits to ozone air quality and human mortality. *Climatic Change, 114*(3-4), 441–461. doi:10.1007/s10584-012-0426-4

West, J. J., Smith, S. J., Silva, R. A., Naik, V., Zhang, Y., Adelan, Z., Fry, M. M., Anenberg, S., Horowitz, L. W., & Lamarque, J. F. (2013). Co-benefits of mitigating global greenhouse gas emissions for future air quality and human health. *Nature Climate Change, 3*(10), 885-889. doi: 10.1038/nclimate2009

West, J. J., Szopa, S., & Hauglustaine, D. A. (2007). Human mortality effects of future concentrations of tropospheric ozone. Comptes Rendus Geoscience, 339(11), 775-783. doi:10.1016/j.crte.2007.08.005

Wheeler, A. J., Gibson, M. D., MacNeill, M., Ward, T. J., Wallace, L. A., Kuchta, J., Seaboyer, M., Dabek-Zlotorzynska, E., Guernsey, J. R., & Stieb, D. M. (2014). Impacts of air cleaners on indoor air quality in residences impacted by wood smoke. *Environmental Science & Technology, 48*(20), 12157–12163. <https://doi.org/10.1021/es503144h>

Wolkoff, P. (1998). Impact of air velocity, temperature, humidity, and air on long-term VOC emissions from building products. Atmospheric Environment, 32(14–15), 2659-2668. doi: 10.1016/S1352-2310(97)00402-0

World Health Organization (WHO). (2020). Ambient (outdoor) air pollution. Retrieved from <www.who.int/mediacentre/factsheets/fs313/en/>

Wotton, B. M., Flannigan, M. D., & Marshall, G. (2017). Potential climate change impacts on fire intensity and key wildfire suppression thresholds in Canada. Environmental Research Letters, 12(9), 095003. doi:10.1088/1748-9326/aa7e6e

Wright, G. S., & Klingenberg, K. (2015). Climate-specific passive building standards. Golden, CO: National Renewable Energy Lab (NREL). doi:10.2172/1215274

Wu, S., Mickley, L. J., Jacob, D. J., Rind, D., & Streets, D. G. (2008a). Effects of 2000 – 2050 changes in climate and emissions on global tropospheric ozone and the policy-relevant background surface ozone in the United States. *Journal of Geophysical Research: Atmospheres, 113*(D18), D18312. doi:10.1029/2007JD009639

Wu, S., Mickley, L. J., Leibensperger, E. M., Jacob, D. J., Rind, D., & Streets, D. G. (2008b). Effects of 2000-2050 change on ozone air quality in the United States. *Journal of Geophysical Research: Atmospheres, 113*(D6), D06302. doi:10.1029/2007JD008917

Xiong, J., Wei, W., Huang, S., & Zhang, Y. (2013). Association between the emission rate and temperature for chemical pollutants in building materials: general correlation and understanding. *Environmental Science and Technology*, *47*(15), 8540–8547. <https://doi.org/10.1021/es401173d>

Yahya, K., Campbell, P., & Zhang Y. (2017). Decadal application of WRF/chem for regional air quality and climate modeling over the U.S. under the representative concentration pathways scenarios. Part 2: Current vs. future simulations. *Atmospheric Environment, 152*, 584 – 604. doi:10.1016/j.atmosenv.2016.12.028

Yao, J., Stieb, D. M., Taylor, E., & Henderson, S. B. (2020). Assessment of the Air Quality Health Index (AQHI) and four alternate AQHI-Plus amendments for wildfire seasons in British Columbia. *Canadian Journal of Public Health = Revue Canadienne de Sante Publique*, *111*(1), 96–106. <https://doi.org/10.17269/s41997-019-00237-w>

Youssouf, H., Liousse, C., Roblou, L., Assamoi, E. M., Salonen, R. O., Maesano, C., Banerjee, S., & Annesi-Maesano, I. (2014). Non-accidental health impacts of wildfire smoke. *International Journal of Environmental Research and Public Health*, *11*(11), 11772–11804. <https://doi.org/10.3390/ijerph111111772>

Yue, X., Mickley, L. J., Logan, J. A., & Kaplan, J. O. (2013). Ensemble projections of wildfire activity and carbonaceous aerosol concentrations over the western United States in the mid-21st century. *Atmospheric Environment, 77*, 767-780. doi:10.1016/j.atmosenv.2013.06.003

Yukon Health and Social Services. (2019). *Wildfire Smoke.* Government of Yukon. Retrieved from <http://www.hss.gov.yk.ca/wildfiresmoke.php>

Zhang, J., & Smith, K. R. (2003). Indoor air pollution: a global health concern. *British Medical Bulletin*, *68*, 209–225. <https://doi.org/10.1093/bmb/ldg029>

Zhang, Y., Smith, S. J., Bowden, J. H., Adelman, Z., & West, J. J. (2017). Co-benefits of global, domestic, and sectoral greenhouse gas mitigation for US air quality and human health in 2050. *Environmental Research Letters, 12*(11), 114033. <https://doi.org/10.1088/1748-9326/aa8f76>

Zhang, Y., West, J. J., Mathur, R., Xing, J., Hogrefe, C., Roselle, S. J., Bash, J. O., Pleim, J. E., Gan, C. M., & Wong, D. C. (2018). Long-term trends in the ambient $PM_{2.5}$- and O3-related mortality burdens in the United States under emission reductions from 1990 to 2010. *Atmospheric Chemistry and Physics*, *18*(20), 15003–15016. <https://doi.org/10.5194/acp-18-15003-2018>

Zickfeld, K., & Herrington, T. (2015). The time lag between a carbon dioxide emission and maximum warming increases with the size of the emission. *Environment Research Letters, 10*(3), 031001. doi:10.1088/1748-9326/10/3/031001

Ziska, L. H., & Beggs, P. J. (2012). Anthropogenic climate change and allergen exposure: The role of plant biology. *The Journal of Allergy and Clinical Immunology, 129*(1), 27-32. doi:10.1016/j.jaci.2011.10.032

Ziska, L. H., Makra, L., Harry, S. K., Bruffaerts, N., Hendrickx, M., Coates, F., Saarto, A., Thibaudon, M., Oliver, G., Damialis, A., Charalampopoulos, A., Vokou, D., Heidmarsson, S., Gudjohnsen, E., Vonini, M., Oh, J. W., Sullivan, K., Ford, L., Brooks, G. D., Myszkowska, D., …Crimmins, A. R. (2019). Temperature-related changes in airborne allergenic pollen abundance and seasonality across the northern hemisphere: A retrospective data analysis. *The Lancet Planetary Health, 3*(3), e124-e131. doi:S2542-5196(19)30015-4