University of Alberta

Economic Analysis of the Human Health Effects from Forest Fires

by

Robyn Rittmaster ©

A thesis submitted to the Faculty of Graduate Studies and Research in partial fulfillment of the requirements for the degree of *Master of Science*

in

Agricultural and Resource Economics

Department of Rural Economy

Edmonton, Alberta

Fall 2004

Your file Votre référence
ISBN: 0-612-95842-6
Our file Notre référence
ISBN: 0-612-95842-6

A JOURNEY OF A THOUSAND MILES BEGINS WITH ONE SINGLE STEP

- JUDY DIDUCK

I am not sure who said this quote originally, but the person must have been writing their thesis. I would like to thank my supervisors Vic Adamowicz and Peter Boxall for making the journey as smooth as possible and the Sustainable Forest Management Network for their research funding. Also, I would like to thank the students of the Rural Economy Department for making everyday enjoyable.

A final thank you to the University of Alberta Panda's hockey team. What a run. I still get shivers. Looking back it wasn't much of a journey, I think we were always at our destination.

TABLE OF CONTENTS

List of Tables

List of Figures

1.0 INTRODUCTION

1.1 Study Background

Recent large and destructive fires have highlighted the substantial costs that are incurred when fighting fires. The 1998 fire season in Alberta cost the provincial government $242 million in fire fighting resources and burned approximately 726,000 hectares of forests (Nash et al, 1999). During the 2001 Chisholm Fire, in northern Alberta, approximately $10 million was spent over seven days in an attempt to suppress the fire (Chisholm Fire Review Committee, 2001). While the monetary costs of fighting fires are often highlighted, the impacts of wildfire smoke on human health are generally ignored in fire management decisions. Wildfire management priorities often include human life and communities (in the immediate vicinity of the fire), timber, infrastructure and environmental values, but seldom include consideration of the impacts off-site from the fire.

Epidemiological studies have linked increases in particulate matter, similar to that released from forest fires, to premature mortality and morbidity effects including cardiovascular and respiratory related illnesses (Sampson et al., 2000). Increases in emergency room visits and restricted activities days have also been attributed to increases in pollution levels (Stieb et al., 2002). Therefore, concerns arising from wildfire should include health effects arising from the smoke impacts.

The purpose of this research is to model smoke dispersion from forest fires in Alberta and use the model to examine the potential economic impacts of adverse health effects. Actual images of smoke plumes are used along with weather variables to predict dispersion behaviour. The health impacts of smoke on populations in Alberta are estimated using the Air Quality Valuation Model 3.0 (AQVM 3.0). The model takes into account economic valuation estimates, Alberta census data and recent epidemiological studies exploring the adverse effects of pollution. By measuring the impacts in economic terms this analysis will provide insights into the magnitude of health effects relative to other fire impacts and will provide information to fire managers about the potential for health impacts from specific fires.

1.2 Problem Identification

Current fire behaviour models are limited because they do not evaluate long-distance smoke dispersion. Additionally, there are no smoke dispersion models that integrate health and economic values into the analysis. Over large populations these health effects can be substantial and should be considered during all forest management decisions. Generally, wildfire management considers human health, but only in immediate proximity to the fire where evacuations are considered. Little research has been done to quantify the health-related costs of wildfire smoke and few attempts have been made to integrate the information into fire management decisions. In response to record setting fire fighting expenditures in 1998, the Land and Forest Service commissioned a review of the Forest Protection Program in Alberta (Nash, 1999). However, without an accurate estimate of the health costs associated with forest fires a benefit-cost analysis of expenditures will not be complete. This research provides some of the necessary analysis to model long-distance smoke dispersion and the related adverse health impacts.

1.3 Problem Overview

This research is part of a larger initiative by the Alberta government through the Sustainable Forest Management Network to identify and map the values at risk from forest fires. The map of the values at risk will be used in resource allocation decisions and the positioning of fire fighting resources in an effort to reduce the values lost to forest fires. Health-related costs are one component of the map; others include recreation values at risk and timber values at risk. Combining these values and organizing them based on specific areas gives an overall assessment of the values at risk in various regions of Alberta. With information outlining the areas that are at risk, priorities can be set as to the amount of fire suppression needed in each region.

An additional aspect of this research is that it can be a part of an assessment of return on investment for fire management. Questions about the effectiveness of fire management can be assessed in a return on investment framework. However, such a framework will require measurements of the values affected by fire management, including human health effects.

This research adds to the information needed to create the values at risk map by analyzing three key questions: 1) How important is smoke and the associated human health impacts compared to other values at risk? 2) Are the human health impacts of smoke more of a concern in some areas relative to others? 3) Can a model of smoke and health impacts be built to help fire managers in resource allocation decisions?

1.4 Study Approach

The overall objective of this thesis is to develop an approach to evaluate the economic costs related to the health effects from forest fires. Chapter two provides an overview of the province of Alberta and describes relevant fire-specific policies and important background information. Chapter three presents literature reviews on the topics of health effects, health valuation and smoke dispersion modeling. In chapter four, a simple smoke dispersion model is developed. This dispersion model focuses on the emissions and concentrations of fine particulates arising from fire. Fine particulates or $PM_{2.5}$ (referring to particulate matter 2.5 microns in size or smaller) are only one potential source of health effects from fire. However, they are identified as being among the most important in generating health impacts. The model uses inputs for fire and weather variables and predicts concentration levels of $PM_{2.5}$ at varying distances from a fire. The predicted concentrations of particulate matter are presented in Chapter five as well as a sensitivity analysis of concentration levels. Chapter six applies the smoke dispersion model and the AQVM 3.0 to examine the health impacts of the 2001 Chisholm Fire, a large fire in central Alberta. This case study is chosen because air quality monitoring data are available and will be used to compare the results from the smoke dispersion model with measures based on actual monitoring stations. Chapter six also presents the economic values of these health effects so that a common measurement unit (dollars) can be used to provide estimates of health values at risk in comparison to other values such as timber values. Chapter seven provides a discussion and conclusions of this research.

2.0 BACKGROUND

2.1 Alberta at a Glance

The population of Alberta is approximately 3.1 million with 60 percent of individuals living in the two largest cities of Calgary and Edmonton (StatsCan, 2004). Alberta's boundaries include 661,000 km^2 of land. Forested land is characterized by boreal spruce and mixedwood conifer in the north and grassland areas in the southern part of the province (NRCan, 2004). Approximately 59 percent of Alberta is forest with the province containing 2.2 billion cubic meters of growing stock (Alberta Economic Development, 2004). The majority of the forests in Alberta are publicly owned (87 percent) with most of the forested land under forest management agreements. Annual harvest totals about 27.4 million cubic meters, contributing to 20,000 direct jobs and generating $2.8 billion in exports (Canadian Forest Service, 2004). Recreation in Alberta is worth approximately $1.4 billion (1994CN) annually and is an important part of the Alberta economy (Dobson and Thompson, 1996). One and a half percent of Alberta's gross domestic product and two percent of all employment results from recreation-related activities (Dobson and Thompson, 1996). Thus, forests are an essential component of life in Alberta; issues relating to forest management are important and imperative to the quality of life in the province.

2.2 Recent Fire History in Alberta

The boreal forest is characterized by frequent fire events. In the past 50 years over 75 percent of the boreal forest in Alberta has burned (SRD, 2004a). Most of the forest is closed canopied forest consisting of conifers, and few segments of old growth exist. Canopies are close to the ground with dead lower branches. This along with low moisture content contributes to high fire intensities and crown fires (Johnson et al., 2001). Generally, fire frequency fits an exponential model with the number of fire occurrences increasing with time (Johnson et al. 2001). Fires mostly occur between April and September with 90% of the area burned in May and June (Johnson et al. 2001).

Most fires in Alberta are small. Less than two percent of fires in the province become larger than 200 hectares. However, this two percent accounts for approximately 97 percent of the total area burned (Amiro et al., 2001). Between 1961 and 1993, 87 percent of fires were ignited by lightning and only 8.4 percent of the fires were attributed to human activities. The losses due to fires are partly related to weather patterns (Cumming, 2001; Bessie and Johnson 1995). Between 1990 and 1995 the average temperature in which fires occurred was 27°C (SRD, 2004b). In comparison, the average temperature in July ranges from 20°C in the south to 14°C in northern Alberta (Government of Alberta, 2004). Li et al., (2000) predicted that climate change will cause these extreme fire conditions to increase creating an environment favourable to catastrophic wildfires.

In the last quarter century the most significant losses from forest fires in Alberta were during the 1998 fire season which was followed by a near record setting 1999 fire season (Nash et al., 1999). The number of fires (1,697) was well above the 10 year average of 912 fires and the total area burned was almost seven times higher than the 10 year average. Costs of fighting large forest fires can reach one million dollars a day. During the 1998 fire season $242 million were spent on forest protection with the largest expenditure related to fighting fires (Nash et al., 1999).

In response to the large expenditures in the 1998 fire season, the Land and Forest Service commissioned a review of the forest protection policies in Alberta. The review concluded that while fire suppression is important, fire seasons like the one experienced in 1998, force fire managers to make important resource allocation decisions. Further, while recognizing Alberta's limited suppression resources, not all fires need to be serviced with the same level of protection. Therefore, the review recommended a priority rating system for forested areas based on values at risk (Nash et al., 1999).

2.3 Current Fire Policy

Current forest policy is guided by two main Acts. The Forest Act and Prairie Protection Act is responsible for fire control, prevention and education. Regulations that attempt to sustain the forest as a resource fall under the Forest Act (SRD, 2004c). Alberta's department of Sustainable Resource Development (SRD) mission is to ensure the sustained contribution to Albertans from Alberta's public and natural resources. To execute this mission, one of SRD's core business goals is to prevent and suppress all wildfires in the forest protection areas (Cardinal, 2004). Communities that are located outside of the forest protection areas are responsible for their own fire control and suppression policies. Fire suppression in Alberta's national parks is the responsibility of the federal government (Cordy Tymstra, personal communication January 2004).

The overall goal of fire management is to reduce the impacts fire has on the residents of Alberta (Chisholm Fire Review Committee, 2001). Upon detection, fires are evaluated and given a priority rating based on the values at risk. Risk of loss of human lives is the first consideration when deciding to control a fire followed by community values, watersheds, natural resources and infrastructure. It is questionable whether health impacts are considered in fire policy since policies do not typically mention smoke effects.

The policy for fighting wildfires in Alberta is to initially attack all fires. A decision is then made, based on the values at risk, to either let the fire burn or attempt to control it. In 2001, 96% of all fires were reported within five minutes and 92% of the fires were suppressed before they reached the size of four hectares (Cardinal, 2004). While there is a time objective to respond to all fires the amount of suppression may vary. In northern Alberta, if the fire is located in the ecological forest management zone, it will be attacked initially but will be allowed to burn if no significant values are at risk. Forests are an essential component to Alberta and its landscape. It is important to note that wildfires have important ecological functions and thus their complete removal would not be desirable. Fire management involves a balancing of the benefits and costs of fire suppressions activities.

Fire suppression effort and costs have been increasing since the 1950's (Schneider, 2002). Improved technology has allowed for the suppression of larger, more remote fires, but the price of suppression is high. An air tanker costs $12,000 per flight. A helicopter costs an average of $1,250 per hour (Cardinal, 2004). Forest managers today are responsible for suppressing wildfires as cost-effectively as possible. The forest industry, through a forest protection fee, pays the Alberta government about $6 million per year towards fire management costs including fire suppression (Cardinal, 2004). The government is therefore responsible for providing protection to forest-related industries from wildfires.

3.0 LITERATURE REVIEWS

3.1 REVIEW OF HEALTH EFFECTS RELATED TO PARTICULATE MATTER INCREASES

Forest fires release particulate matter containing the biological material that has been burned (Ward, 1999). Particulate matter released from wood smoke is harmful to human health because toxic organic compounds are inhaled and absorbed through the lungs (Sampson, 2000). Fine particulate matter ($PM_{2.5}$) is defined as any atmospheric material with a diameter equal to or smaller than 2.5 microns in aerodynamic diameter (μm) (Green, 2002). When presented in terms of concentration levels $PM_{2.5}$ is termed micrograms per cubic meter of air ($\mu g/m^3$). The form of the particulate can range from liquid to dust and contain pollutants such as carbon, sulfate and gas. While this research focuses on particulate matter with a 2.5μm diameter, it is also plausible that larger sizes of particulate (PM_{10}) will have associated adverse health impacts. However, for simplicity we focus on the health effects related to fine particulate matter only.

Ideally, increases in $PM_{2.5}$ from forest fires would be available to analyze, however few studies have examined the health effects resulting from forest fires directly (Ward, 1999). The few studies that examine this issue are related to prescribed burns and not wildfires. Consequently, we rely on the air pollution literature that examines the health effects due to increases in $PM_{2.5}$. The Canadian standard for $PM_{2.5}$ is $30\mu g/m^3$ averaged over a 24-hour period (CCME, 2004). Estimates of the increase in mortality related to increases in particulate matter are provided below in Table 3.1. It should be noted that only the positive associations are reported. While some studies have shown negative associations these were not common in the literature. One possibility is a publication bias on the part of researchers since negative associations may not have supported their original hypothesis. Additionally, it is hard to find a reasonable explanation for a result that shows increases in particulate matter reduce the probability of death (Heuss, 1999 and Steenland, 1997).

Studies examining the correlation between increases in particulate matter and health effects can be divided into two categories; time-series and cohort tracking. Time series is the most prevalent. These studies examine daily particulate matter levels and correlate them with adverse health effects (mortality, hospital admissions, etc.) on the day of or the few days after the increase in particulate matter. The other type of analysis tracks cohorts of individuals and assesses their response to changes in air quality (Royal Society of Canada, 2001). These studies tend to generate higher impacts because of the potential for chronic exposure and because the heterogeneity in the characteristics of the individuals is accounted for.

Since 1996, there have been over 800 studies from around the world focusing on the health effects related to increases in particulate matter (American Lung Association, 2001). This next section presents a relevant subset of the recent epidemiological research from North America and concludes with a section describing some of the limitations of the research.

3.1.1 Time Series Studies of Mortality and Morbidity Increases Related to $PM_{2.5}$

Daily time-series studies focus on short-term increases in particulate matter and the associated acute mortality and morbidity outcomes. Due to the short-term nature of most time-series studies long-term exposure and chronic health effects are not measurable. Therefore, the findings presented in this section, and in Table 3.1, are likely an underestimate of the true health effects from long-term particulate matter increases. Samet et al. (2000) found a 0.5 percent increase in mortality for every $10\mu g/m^3$ increase in PM_{10}. Deaths were attributed to heart and lung disease. Morbidity outcomes, for the same study, included a one percent increase in hospital admissions for cardiovascular disease and a two percent increase in admissions for pneumonia and cardiovascular obstructive pulmonary disease (American Lung Association, 2001). Lippman et al. (2000) found positive effects of $PM_{2.5}$ on all-case mortality and for pneumonia hospital admissions, cardiovascular related admissions, and incidence of stroke.

3.1.2 Cohort Studies of Mortality

Cohort studies not only account for acute health effects, but also take into account the increases in chronic health outcomes related to increases in particulate levels.[1] The leading research in the analysis of the correlation between mortality and particulate matter comes from the Health Effects Institute (American Lung Association, 2001). Recently, the Health Effects Institute confirmed results from previous cohort studies that found increases in mortality rates for individuals in cities with high levels of particulate matter. Sensitivity analysis was completed that controlled for various statistical methods and confounding variables such as health status, weather variables and other pollutants.

Pope et al. (2002) examined health data that followed 500,000 Americans for 16 years. They concluded that $PM_{2.5}$ was correlated with increased risk of lung cancer mortality and cardiopulmonary mortality. The researchers controlled for individual-level factors including smoking, body mass, diet, occupation and many other factors.

[1] Cohort studies as defined by the International Society for Pharmacoeconomics and Outcomes research in 1998 are studies in which the experiences of a defined group (the cohort) are followed over time to link patient and or clinical management characteristics with various health outcomes.

Table 3.1. Results from recent studies examining the human health effects related to increases in particulate matter concentrations.

Source	Pollution Level	Health Effect	Notable Results (95% Confidence Interval)	Study Type
Sullivan et al., 2003	13.8μg/m^3 increase in $PM_{2.5}$	Cardiac arrest for smokers with preexisting heart disease	29% increase (0.06-0.55)	Time series
Schwartz et al., 2002	10μg/m^3 increase in $PM_{2.5}$	All-case mortality	1.5% increase (CI not reported)	Time series
Dominici et al., 2002	For a 10μg/m^3 increase in PM_{10} for one day	All-case mortality	0.48% increase (0.05-0.92)	Time series
Peters et al., 2001	For a 25μg/m^3 increase in PM_{25} 2 hr before admission	Myocardial infraction	1.5 RR[a] (1.1-2.0)	Time series
Levy et al., 2001	For a 19.3μg/m^3 increases in PM_{10}	Cardiac arrest	0.87 (0.74-1.0)	Time series
Limppman et al., 2000	For a 36μg/m^3 increase in $PM_{2.5}$	All-case mortality (measured other morbidity outcomes)	RR 0.045 (-0.09 -.0102%)	Time series
Samet et al., 2000	For a 10μg/m^3 increase in PM_{10} for one day	All-case mortality	0.50%	Time series
Daniels et al., 2000	For a 10μg/m^3 increase in PM_{10} for one day	Respiratory and cardiovascular related mortality	0.69% increase (0.40-0.98)	Time series
Schwartz, 2000	For a 10μg/m^3 increase in PM_{10} for one day	All-case mortality	0.67% increase (0.52-0.81%)	Time series
Health Canada (AQVM 1999)	For a 1μg/m^3 increase in yearly $PM_{2.5}$	Mortality	2.14 extra deaths per million population	
Abbey et al., 1999	For a 100μg/m^3 increase in PM_{10} for 43 days	Cardiovascular related mortality	18% increase	Time series
Schwartz et al., 1996	For a 1μg/m^3 increase in yearly PM_{10}	Mortality	12.1 extra deaths per million population	
Pope et al., 1995	For a 10μg/m^3 increase in PM_{10}	Mortality	3.5% increase	Cross sectional

[a] relative risk

3.1.3 Extensions to Current Health Effects Research

There are concerns in the concentration-response literature regarding the analysis of threshold levels (Levy, 2003 and Deck, 2001). Current literature suggests that there is little evidence of a minimum threshold level at which particulate matter does not affect health (Daniels et al., 2000; Schwartz and Zanobetti, 2002; Chestnut et al., 1999; and Rabl, 1999). Daniels et al. (2000) do not find evidence of a threshold level while Burnett et al. (1995) demonstrate significant health effects for particulate levels at their lowest measurable amount of $25\mu g/m^3$. Recent research by Schwartz and Zanobetti (2002, 1996) suggested the minimum dose-response level may be as low as $2\mu g/m^3$. Regardless of the true minimum threshold level, researchers have found that there are health benefits to reducing particulate matter below the Canada Wide Standard of $30\mu g/m^3$ (Pandey and Nathwani, 2003).

The shape of the concentration-response function is also uncertain (Deck, 2001; Schwartz and Zanobetti, 2002 and Schwartz et al., 2002). It may be that the relationship is not linear, as assumed by most researchers for ease of measurement. A maximum level at which adverse health effects taper or increase substantially has not been determined either. An additional concern put forth by the Royal Society of Canada (2001) is that the frequency and variation of particulate matter levels may play an important role in determining the extent of pollution related health problems. The Royal Society of Canada questions the extent as to which peaks in concentration levels, would be more harmful than just a rise in the annual average. This is important when comparing results of health effects studies from different locations. Each region may not only have a different baseline level of particulate matter, but also different patterns to the pollution. Another confounding factor is the makeup of the particulate matter itself (Deck, 2001). Particulate matter contains a mixture of combustion particles, dust, nitrates and sulfate, therefore it may be that the particulate matter in one study area is more harmful than particulates from another area of interest (Rabl, 1999).

It is also likely that there are separate concentration-response functions for chronic and acute effects (Leksell and Rabl, 2001 and Nevalainen and Pekkanen 1998). Time series studies only measure health outcomes for the days surrounding an event but long-term health changes are not observed. A complete analysis would include cohort studies that examine the increase in health effects from a population exposed to high levels of particulate matter and compare them over the same timeframe to a population not exposed to similar levels of particulate matter.

An important component of the literature that is being explored is the concentration-response relationship for sub-groups of the population who are at risk (Levy 2003; Coyle, 2003 and Deck 2001). For example, individuals who have asthma or a preexisting heart condition will likely be more at risk than a healthier population (Sullivan, 2003; Goldberg et al., 2000 and Zanobetti et al., 2002). Additionally, the elderly and young children may be affected differently by particulate matter and ozone than a different age group (Pandey and Nathwani, 2003). Delfino et al. (1997) found a positive association between $PM_{2.5}$ and emergency room visits in the elderly, while Gauderman et al. (2000) discovered decreased lung function in children who experienced increased levels of $PM_{2.5}$. Relating the concentration-response function to socio-demographic variables will also help policy makers in their measurement of the health effects of pollution in a specific region.

3.1.4 Concerns Surrounding Current Health Effects Research

While there seems to be over-whelming evidence relating increases in $PM_{2.5}$ to adverse health outcomes, a few researchers have questioned the validity of this evidence. Two notable articles illustrate these concerns. The most recent, Koop and Tole (2004) critique the statistical techniques used in time-series analysis. They provide evidence that positive correlations occur only in a few selected models while the bulk of estimates show negative or no correlation between particulate levels and adverse health impacts. Additionally, when model uncertainty is accounted for and used when estimates are already relatively small, confidence intervals more often than not suggest that pollution has no effect on mortality.

The second notable study, by Green et al. (2002), questions the assumption that all types of particulate have the same toxicity levels. The authors point out that even oxygen particles, which do not have any apparent adverse health impacts, can be characterized as particulate matter and thus must meet air quality standards for particulate matter. While it is true that the components of the particulate probably do affect health in different ways, the particulate matter from forest fires contains pollutants that cause health problems (McKenzie et al., 1995). Additionally, the suggestion to monitor and regulate particulates based on their components is not realistic in practice. Similar to Koop and Tole, Green et al. (2002) also suggest that cohort studies be favoured over time-series analysis.

3.2 CONCENTRATION-RESPONSE FUNCTIONS AND ECONOMIC VALUATION USED IN THE AQVM 3.0

3.2.1 Background

The AQVM 3.0 was developed by Environment Canada and Health Canada to estimate the human health and welfare costs or benefits resulting from changes in air quality in Canada. The background studies used for the concentration-response functions and health outcome valuation estimates were chosen based on a comprehensive literature review by the model's developers. The AQVM 3.0 and supporting methodology was also reviewed by the Royal Society of Canada in 2001 (Royal Society of Canada, 2001). If new information becomes available with regards to concentration-response functions or health valuation estimates the changes can be incorporated relatively easily into the AQVM 3.0. However for the purpose of this thesis, the concentration-response functions and health valuation estimates will not be altered and we rely on the work by previous researchers.

The underlying methodology for determining human health impacts and related costs from changes in particulate matter levels is the damage function approach. This approach assumes that changes in particulate levels can be monitored and applied to concentration-response functions. Economic valuation techniques are then used to value the health impacts. AQVM 3.0's role in the damage function approach is to use external estimates of changes in particulate concentration levels to quantify related health effects and monetary costs. The internal concentration-response functions and estimates of health costs are explained in the following sections.

3.2.2 Concentration-Response Functions

Concentration-response functions describe the relationship between changes in $PM_{2.5}$ levels and human health impacts. Generally, these relationships are for a given change in a particulate level, for example $1\mu g/m^3$, and a specified amount of exposure time. In the case of increases in $PM_{2.5}$ in this study, only acute health effects are measured as it is assumed that the increase in concentration level is for a short period. Health endpoints available for $PM_{2.5}$ in the AQVM 3.0 model include mortality, cardiac hospital admissions, respiratory hospital admissions, emergency room visits, asthma symptom days, restricted activity days, acute respiratory symptom event and childhood bronchitis.

Asthma symptom days describe the number of days of elevated asthma symptoms for the 6% of the population that has asthma. The concentration-response functions were obtained from a survey of self-reported asthma symptoms including shortness of breath, wheezing or an increase in medication use. Productivity losses due to illness are approximated by quantifying the number of restricted activity days. Days spent in bed, days missed from work and days when activity is restricted are termed restricted activity days. An acute respiratory symptom is one of 19 symptoms including coughing, sore throat, head cold, sinus trouble, headache, flu etc. The concentration-response functions were derived from research correlating observations made in health diaries to pollution levels. Not included in the analysis are chronic illness related to $PM_{2.5}$. Long term increases in $PM_{2.5}$ are associated with chronic respiratory disease, however the illness is the result of long-term exposure and not the short-term increases that are experienced with forest fires. The concentration-response functions used in AQVM 3.0 are the result of a comprehensive literature review. Probability weights for each function are provided based on the confidence of the estimate.

Concentration-response functions are multiplied by the particulate matter increase and by the population exposed to determine the extent of a specific health outcome. For example, the low estimate of mortality risk from a $1 \mu g/m^3$ annual increase in $PM_{2.5}$ is $0.87*10^{-5}$ and if the population exposed to the increase is one million then the number of mortality cases each year is expected to increase by six. Annual concentration-response functions can be divided by 365 to calculate daily responses to changes in $PM_{2.5}$ levels. To determine the number of individuals exposed, population characteristics for each census subdivision are calculated (internal to AQVM 3.0).

There are over 800 studies describing the correlations between health outcomes and particulate matter (American Lung Association, 2001). Developers of AQVM 3.0 assessed the values from studies which were evaluated based on the quality of the research. Criteria for data quality included continuous monitoring of particulate matter levels, minimal location bias, control of seasonal and weather patterns, model specification and studies that tested a Canadian population. Table 3.2 summarizes the concentration-response functions used in the AQVM 3.0. Some of the health effects included in the model overlap. Acute respiratory symptom days include days that are also restricted activity days. To avoid double counting, the concentration-response functions have been adjusted.

Table 3.2 Concentration-response functions used in AQVM 3.0 for increases in $PM_{2.5}$ concentrations.

Health Effect	Concentration-response function	
Annual mortality risk per 1 μg/m^3 change in annual average $PM_{2.5}$ concentration. Sources: Pope et al. (1995); Schwartz et al. (1996).	Low Central High	$0.87*10^{-5}$ (22%) $2.14*10^{-5}$ (67%) $4.82*10^{-5}$ (11%)
Respiratory hospital admissions daily risk factors per 1 μg/m^3 change in daily average $PM_{2.5}$ concentration. Population 25 years old and older. Source: Burnett et al. (1995).	Low Central High	$4.13*10^{-8}$ (25%) $1.21*10^{-8}$ (50%) $1.42*10^{-8}$ (11%)
Cardiac hospital admissions daily risk per 1 μg/m^3 change in daily average $PM_{2.5}$ concentrations. Source: Burnett et al. (1995).	Low Central High	$1.00*10^{-8}$ (25%) $8.27*10^{-8}$ (50%) $12.4*10^{-8}$ (25%)
Net emergency room visits daily risk factors per 1μg/m^3 change in daily averages $PM_{2.5}$ concentrations. Source: Stieb et al. (1995).	Low Central High	$4.62*10^{-8}$ (25%) $8.27*10^{-8}$ (50%) $12.4*10^{-8}$ (25%)
Asthma symptom days daily risk factors given a 1μg/m^3 change in daily average $PM_{2.5}$ concentration. For population with asthma (6%). Sources: Whittemore and Korn (1980); Ostro et al. (1991).	Low Central High	$1.6*10^{-4}$ (33%) $2.64*10^{-4}$ (34%) $3.65*10^{-4}$ (33%)
Restricted activity days daily risk factors given a 1μg/m^3 change in daily average $PM_{2.5}$ concentration. Non-asthmatic population (94%). Sources: Ostro (1987); Ostro and Rothschild (1989).	Low Central High	$1.31*10^{-4}$ (25%) $2.50*10^{-4}$ (50%) $3.95*10^{-4}$ (25%)
Net day with acute respiratory symptom daily risk factors given a 1 μg/m^3 change in daily average $PM_{2.5}$ concentration. Sources: Krupnick et al. (1990).	Low Central High	$1.25*10^{-4}$ (25%) $2.79*10^{-4}$ (50%) $4.14*10^{-4}$ (25%)
Child acute bronchitis annual risk factors given a 1μg/m^3 change in annual average $PM_{2.5}$ concentration. Source: Dockery et al. (1996).	Low Central High	$0.62*10^{-3}$ (25%) $1.65*10\text{-}^{3}$ (50%) $2.69*10\text{-}^{3}$ (25%)

Note: Low, central and high estimates refer to ranges of estimates in the reviewed literature. Percentages are given suggesting the confidence of the estimate.
*Table extracted from Chestnut et al., 1999.

3.2.3 Estimates of Economic Values used in AQVM 3.0

Concentration-response functions, combined with particulate level increases and population information, are used to quantify health impacts. It is useful for policy makers to not only know the quantity of impacts (number of cases), but also the related economic value. AQVM 3.0 uses a combination of cost-of-illness and willingness to pay approaches for monetary evaluation. The approach selected differs by health endpoint and is selected based on the evidence from the recent literature on health valuation.

Direct medical costs and lost productivity are usually measured using the cost-of–illness approach. The approach uses wage data and hospital expenditure data to calculate total costs. Indirect costs are defined as days with restricted activity and the costs of pain and suffering. The cost-of-illness approach undervalues the true cost because it does not account for pain and suffering (Freeman, 2003). As a result, economists often use willingness to pay methods (or stated and revealed preference methods) to not only account for direct medical costs but the indirect costs of illness as well. Costs related to morbidity such as an asthma event are relatively straight forward to calculate, however the costs associated with mortality need more explanation. The approach for estimating mortality values is explained below.

3.2.4 Values Related to Mortality Risk Reductions

Although AQVM 3.0 predicts the number of lives lost to a particulate matter increases, it is not these lives per se that are being valued when estimating monetary values. Rather it is the increase in risk of death from concentration increases that is being valued. The value of a mortality risk increase is determined using revealed or stated preference methods mentioned previously. The underlying theory is that value is defined as the amount that a person would be willing to trade for something else. In this context individuals trade income for changes in their probability of death (Freeman, 2003).

3.2.5 Stated Preference Methods

Values relating to risk changes can be obtained using survey methods which elicit information about the consumer's values and preferences (Freeman, 2003). Briefly, maximum willingness-to-pay is calculated from answers to surveys that ask respondents hypothetical questions about how much income they would give up to avoid an increase in the risk of death. A survey may ask respondents their willingness-to-pay to avoid an increase in the risk of death due to increased particulate levels. When the willingness-to-pay value is added over a population and then divided by the total number of lives saved from avoiding the risk increase, a "value of a statistical life" (VSL) is obtained. For example, if 100,000 respondents are willing to pay $240 to reduce their risk of death by 0.004% (4 in 100,000) then respondents, collectively, are willing to pay $6 million per statistical life. The value does not suggest that one person would pay $6 million to avoid death or would accept $6 million as compensation for death, rather the value is a collective value over a population (Hammitt, 2000).

3.2.6 Revealed Preference Methods

An alternative framework to stated preference measures are revealed preference methods. Revealed preference methods observe actual market behaviour to evaluate a consumer's preferences for changes in risk. One example of a revealed preference method is to relate wages to increased on-the-job risk of mortality. The value of mortality risk can be obtained by comparing jobs with an increased risk of death to the wage differentials from jobs without risk. This approach assumes that workers will need to be compensated for accepting an increased risk of death at work.

Morbidity costs are estimated using cost-of-illness studies and willingness-to-pay methods. Low, central and high estimates are used to cover the range of values estimated by researchers. Table 3.3 provides the estimates associated with mortality risk reductions while Table 3.4 shows values associated with morbidity.

Table 3.3. Economic values used in AQVM 3.0 for a mortality risk reduction for specific age groups.

Age-Adjusted Values	Values (1996 CAN$ millions)		
	Low	Central	High
>65 years old	$2.3	$3.9	$7.8
<65 years old	$3.1	$5.2	$10.4
Age-weighted average	$2.4	$4.1	$8.2
Weights used in sensitivity analysis describing the confidence of the estimate	33%	50%	17%

*Table extracted from Chestnut et al., 1999

Table 3.4. Economic valuation of morbidity estimates used in the AQVM 3.0.

Health Effect	Low	Central	High	Source	Study Type
	Values (1996 CAN$ millions)				
Respiratory Hospital Admission	$3,300	$6,600	$9800	Canadian Institute for Health Information (1994)	COI[a]
Cardiac Hospital Admission	$4,200	$8,400	$12,600	Canadian Institute for Health Information (1994)	COI
Emergency Room Visit	$290	$570	$860	Rowe et al. (1986)	COI
Childhood Bronchitis	$150	$310	$460	Krupnick and Cropper (1989)	COI
Restricted Activity Day	$37	$73	$110	Loehman et al. (1979)	WTP[b]/COI
Asthma Symptom Day	$17	$46	$75	Rowe and Chestnut (1986)	WTP
Minor Restricted Activity Day	$20	$33	$57	Krupnick and Kopp (1988)	WTP
Acute Respiratory Symptom Day	$7	$15	422	Loehman et al. (1979)	WTP
Probability values associated with the morbidity estimates	33%	34%	33%		

[a]Cost of Illness studies

[b]Willingness to Pay studies

* Table extracted from Chestnut et al., 1999.

3.3 SUMMARY

The concentration-response functions used in AQVM 3.0 specify the relationship between increases in $PM_{2.5}$ and human health effects. These values along with the associated costs of each health effect are determined through a literature review of current estimates. The costs of non-fatal illness resulting from increases in $PM_{2.5}$ levels can be valued by similar methods using cost-of-illness estimates. Mortality risk increases are valued using willingness to pay measures derived from stated and revealed preference studies. Ranges for each estimate and a probability weight are provided to be used in sensitivity analyses. For a detailed discussion of the strengths and weakness of measures used in health valuation see Freeman, 2003.

The basis for the Canada Wide Standard regulating particulate levels comes from the examination of a variety of time-series and cohort studies (CCME, 2004). Much research and effort has been put into quantifying the relationship between increases in particulate levels and adverse health impacts. While some researchers have questioned the validity of the results there is over-whelming evidence showing that increases in particulate levels cause health problems. Future research will probably focus analysis on subgroups of the population and on the costs and benefits associated with attaining various particulate standards. In the context of forest fires examined in this study, the impacts from increased particulate levels are assumed to be the same as the particulate matter impacts from pollution. If there is no minimum threshold for the effects of particulate matter then populations relatively large distances away from a source may experience significant health problems.

3.4.0 REVIEW OF SMOKE DISPERSION AND FIRE BEHAVIOUR MODELS

3.4.1 Limitations of Current Smoke Dispersion and Fire Behaviour Models

An ideal smoke dispersion model will take into account not only fire specific influences, but will model long distance smoke dispersion behaviour. There are approximately 40 different fire management models (USDA, 2004a). Previously, fire behaviour models were combined with smoke dispersion models to estimate total smoke dispersion from various types of fires, however, few models are self sufficient. This section reviews current smoke dispersion and fire behaviour models that assist fire management teams.

3.4.2 Fire Behaviour Models

Fire behaviour models predict fire patterns and intensity based on fuel type and weather variables. The models described in this section can stand alone to predict particulate emissions or can be linked with other models. CONSUME is one of the most commonly used models to predict the amount of particulate matter released from a fire. While it does not focus on dispersion, it can be linked with a dispersion model. The advantages of using CONSUME are that it has an easy to use interface and it takes into account landscape and fire conditions (Ottmar et al., 1993). FOFEM (First Order Fire Effects Model) is similar to CONSUME in that it predicts initial smoke emissions, but not dispersion (Reinhardt, 1997). The focus of FOFEM is on tree mortality and other ecological effects. Another often used model is the FARSITE model. It is primarily a fire growth model, but can also estimate emission releases. Its major advantage is that it allows for a variety of landscapes and fire intensities and is linked with a GIS system to produce maps of fire growth. Users must take an official course to use the model (Finney and Andrews, 1999).

3.4.3 Smoke Dispersion Models

The major components of smoke dispersion models are weather variables and fire release information. Models used by fire managers and researchers include FIREPLUME and CALpuff. FIREPLUME calculates ground level concentration of a pollutant downwind from the source. The background for the model comes from fog-oil smoke plumes in California and not wildfire specifically. A strength of the model is that it accounts for three different release conditions; flaming, glowing and smoldering (Brown et al., 1999). One limitation of the model is that it is recommended for local and not long-distance analysis. Most other models limit analysis to 100 kilometers (Breyfogle and Ferguson, 1996). CALpuff is a dispersion-only model that is recommended by the EPA for dispersion modeling for distances greater than 50 kilometers (EPA, 2003). However, Breyfogle and Ferguson (1996) suggest that the model has not been adequately tested in wildland fires and should not be used to model smoke from wildfire events. Another limitation of this model is that it needs inputs for the amount of pollutant released and it may be difficult to locate sufficient meteorological data.

Additional models focus on prescribed burns and not wildfires. While most of the models described above take into account landscape-specific information, none have been developed and verified in Western Canada. NFSPuff is a three dimensional computer model that incorporates emissions released from a fire as well as dispersion characteristics. Weather data for the model must be obtained through the National Center for Environmental Prediction and a private vendor (Breyfogle and Ferguson, 1996) This model is one of the few models that has sufficient range to cover landscapes as large and similar to Alberta. Optimal range is less than 488 kilometers. It was initially designed for the complex terrain of the western United States (USDA, 2004a). However, the model only estimates PM_{10} emissions and not $PM_{2.5}$.

A combination fire behaviour-smoke dispersion model called BlueSky is being developed by the USDA and shows some potential for future fire management planning. Although still in testing stages, BlueSky can estimate $PM_{2.5}$ concentration levels. The model is an improvement over previous tools because it is relatively easy to use and accounts for prescribed burns and wildfires. By 2004 it is scheduled to be available for smoke dispersion analysis across the United States. The impact of smoke from Canadian wildfires on the United States has led current users in the United States to request the incorporation of Canadian wildfire data into the modeling. Currently the model lacks fuel loading estimates for Canada and thus emission concentrations cannot be estimated. However, future versions of BlueSky will likely incorporate Canadian weather data and landscape information (USDA, 2004b).

3.4.4 Conclusion

Many smoke dispersion and fire behaviour models attempt to predict the concentration of a pollutant downwind from a source. However the models have limitations either with their design, output limitations or ease of use. Additionally, few models focus on the long distance dispersion of $PM_{2.5}$ emissions. The models tend to operate at distances of less than 100 kilometers. While the model BlueSky will be a consideration in the future, the limitations of current models are key factors leading us to focus on the use of satellite images of actual smoke plumes to predict dispersion of particulate matter concentrations.

4.0 METHODS: SMOKE DISPERSION MODELING AND PARTICULATE MATTER CONCENTRATION ESTIMATES

4.1 INTRODUCTION

The difficulty in using computer-assisted dispersion models to predict smoke dispersion over long distances lead to the development of estimation based on actual satellite images of smoke plumes and associated weather data. Images examined from western Canada and the north-western United States provide an advantage when predicting dispersion in Alberta as opposed to using a general computer model that is not based on a specific location. The following sections describe the methods used to predict long distance smoke dispersion and the concentration of fine particulate matter contained in the smoke plume. This strategy involved measuring images of actual smoke plumes and developing statistical relationships between the width and length of the plume. Associated with each image are weather and fire specific variables that are believed to influence plume width. This relationship is then combined with estimations of emissions and fire intensity to predict particulate matter concentrations.

Two components are needed to estimate the level of particulate mater at a given distance. The first is an emission release term which describes the quantity of $PM_{2.5}$ that is released from the fire. The formula used for the amount of particulate matter released is shown in Equation 4.1.

$$\text{Total Emissions Released} = ROS * CONSUMPTION * EMISSION\_FACTOR \qquad (4.1)$$

ROS is the rate of spread of the burned area in hectares per minute, CONSUMPTION is the amount of fuel burned in kilograms per hectare and EMISSION_FACTOR is the amount of $PM_{2.5}$ released per unit of fuel burned.

The second component needed to measure particulate matter concentration levels is a dispersion term which describes the area in which the particulate matter, from Equation 4.1, is dispersed. Dispersion area is defined in Equation 4.2:

$$\text{Dispersion Area} = \frac{1}{WIDTH * WIND * HEIGHT} \qquad (4.2)$$

In Equation 4.2 WIDTH is a statistical analysis predicting the width of the smoke plume as a function of distance and the atmospheric dispersion index, WIND is the wind speed in meters per minute, HEIGHT is the mixing height in meters. Equation 4.3 combines the two previous equations to give total grams of $PM_{2.5}$ per cubic meter of air.

Equation 4.3: Grams of $PM_{2.5}$ per Cubic Meter of Air

$$= \frac{1}{WIDTH * WIND * HEIGHT} * ROS * CONSUMPTION * EMISSION\_FACTOR \qquad (4.3)$$

The above equation can be explained by two diagrams. Figures 4.1 and 4.2 show two cross sections of a smoke plume. Figure 4.1 shows the mixing height of the plume which defines the area that the particulate matter mixes vertically while Figure 4.2 is an overhead view showing the increase in plume width as distance increases. The particulate matter released into the plume area is defined from Equation 4.1. It is assumed that the particulate matter released from the fire is dispersed evenly throughout the area of the plume.

Figure 4.1: Mixing height

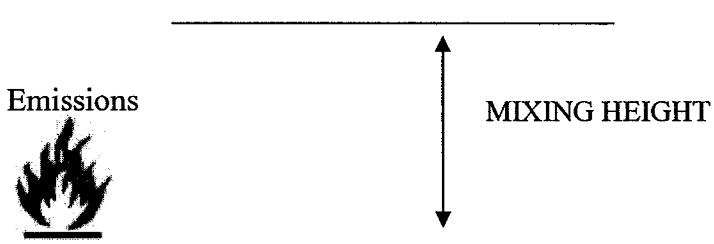
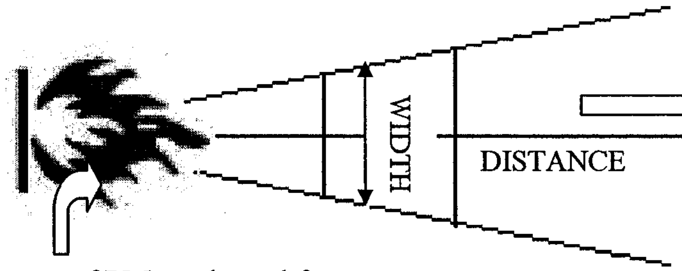


Figure 4.2: Smoke dispersion factors



$PM_{2.5}$ is typically measured in micrograms per cubic meter and thus the values calculated are reported in this fashion as well. The change in particulate matter levels can be calculated given different fire and meteorological conditions. The variances of each variable will be used to provide a sensitivity analysis of potential particulate matter concentrations.

4.2.0 DISPERSION MODELING WITH SATELLITE IMAGES

4.2.1 Image Access, Wildfire and Weather Data

The estimation of the WIDTH variable which estimates plume width at a given distance from the fire was completed by analyzing satellite images of forest fire smoke plumes. Images were obtained from NASA's Rapid Response System which provides access to Moderate Resolution Imaging Spectroradiometer (MODIS) satellite images. The MODIS images are of a one kilometer resolution and provide visible smoke plumes as well as an outline indicating thermal hotspots. The hotspots are detected by the satellite using data from the middle infrared and thermal infrared bands. While the hotspots do not indicate fire size they are a good indicator of the general location. The purpose of using the images is to develop a statistical relationship between length of the plume and width.

Weather and wildfire data which were used in the estimation of plume width and the calculation of particulate matter emissions were obtained from the Canadian Forest Service. Observations were available from meteorological weather stations located in Western Canada. The closest weather station to each fire was used. In a few cases the closest weather station could not provide data points for the requested dates and times, in which case the next closest weather station's data were substituted.

4.2.2 Plume Selection

Fire image selection comes from western Canada and the north western United States during the 2000-2003 fire seasons. Plume selection included plumes that had a continuous visible plume not interrupted by cloud or smoke from other fires. Plumes in which a direction and general shape could not be determined were not included in the analysis. Additional exclusions included plumes that had a distance of less than 1.5 centimeters on the image. The distance of 1.5 centimeters represents three measurement intervals. When converted to actual kilometers, 1.5 centimeters is between 5 and 21 kilometers depending on the scale. In general, the measurement of distance stopped either when the plume changed direction significantly or when the dispersion of smoke was too faint to measure. Approximately 60 hotspots were detected with 35 measurable plumes. From the 35 measurable plumes, weather data was available for 21 plumes

4.2.3 Scale Determination

The set of images obtained from the two satellites were not geo-referenced. Georeferencing corrects for the distortion caused when a curved object such as the earth is projected as a flat image. Without distinct coordinates, distance measurements are not possible. Thus the scale of the image was determined by comparing the distance between known landmarks on the image and the same landmarks on a map. Common landmarks included state and provincial boundaries as well as distances between rivers and lakes. Four measurements from each image were used to determine the scale: two from a north to south landmark and two from an east to west landmark.

4.2.4 Distance and Width Measurements

Distance indicators were marked on the image every five millimeters between the end of the hotspot (identified by the satellite) and the visible end of the plume. Width measurements of the plume were taken at each indicator. Difficulty arose when measuring the plume width; smoke plumes have two components. The components include one distinct white-coloured plume and beyond the obvious plume are fainter signs of smoke. A magnifying glass was used to locate the edge of the plume. Where possible the largest visible portion of the plume is used for measurement. However, in some cases, only the distinct white plume was used because of the difficulty in identifying the plume boundary.

4.3.0 PLUME DIMENSION ANALYSIS

Smoke disperses through a volume of air that is defined by width, wind speed and height. Plume width is modeled as a linear function of distance and an atmospheric dispersion index (ADI), however other variables accounting for the influence of adjacent fires and fuel flammability are also examined in the estimation of plume width. Mixing height and wind speed are part of the dispersion index, but will also be used to represent the height and length of the smoke plume. The dispersion parameters used for mixing height, wind speed and ADI were taken from daytime measurements which correspond to the daytime images used to extract width and distance data. Generally, dispersion characteristics lessen during the night and can cause particulate matter to be trapped in a small area.

4.3.1 Atmospheric Dispersion Index (ADI)

Atmospheric conditions are the main factors contributing to smoke dispersion and the estimation of plume width. The atmospheric dispersion index is used by the Canadian Forest Service to predict the ease of smoke dispersion. The index was chosen because many of the weather variables predicting dispersion were correlated. Using the ADI provided a summary measure of the weather conditions close to the time the satellite photo was taken. ADI values range from 1 to over 100. Low values represent poor dispersion characteristics. At these values smoke will stay close to the point of origin and particulate matter concentration levels will remain high. Indexes greater than 60 indicate good dispersion conditions, while values greater than 100 represent blow up conditions with crown fires likely. The ADI consists of the four weather variables listed below (NRCan, 2004):

Ceiling Height: The height at which the highest cloud layer covers at least 50% of the sky.

Pasquill-Gifford-Turner Stability Index: This stability index takes into account the amount of mixing between atmospheric layers and the temperature of each layer. Unstable conditions allow the smoke plume to mix through a larger volume creating low concentrations of particulate matter.

Mixing Height: The mixing height is the maximum distance smoke will rise. A low mixing height traps smoke near the ground. During the daytime mixing height generally increases and smoke can be dispersed vertically for several kilometers.

Transport Wind Vector: The transport wind vector accounts for wind speed. A high wind speed increases the amount of dispersion allowing for lower concentrations of particulate matter, however, high winds speeds also contribute to a larger plume and larger fire spread rate.

4.3.2 Statistical Estimation of Plume Width

Each of the 21 smoke plume images provides measures of the specific plume's distance and width which form a panel data set. The visible length of each plume varied causing the number of width measurements available per plume to range between 3 and 24, which created an unbalanced panel data set. A stratification variable was used to account for the unbalanced number of width and distance measurements across fires. Panel data taken from a variety of fires allows for flexibility in modeling the differences in the initial width measurement across fires. Equation 4.4 and Equation 4.5 summarize the statistical models. The intercept term in Equation 4.4 represents the initial width measurement which cannot be assumed to be constant across all plumes as would be required using traditional OLS regression models (Greene, 2000). Estimating the model using an OLS framework will create estimators that are not efficient. A more realistic approach is to assume that the intercept is a random variable with a normal distribution. Consequently, a random effects model is also estimated and described in Equation 4.5. The model accounts for the different intercepts across fires and the panel nature of the data. The software package LIMDEP was used to estimate both models.

An example of one specification of the OLS model is:

$$WIDTH_{it} = \alpha_1 + \beta_1 DIST_{it} + \beta_2 ADI_{it} + \varepsilon_{it} \tag{4.4}$$

where WIDTH is the width of the i^{th} plume at the t^{th} observation, DIST is the distance away from the plume's source and ADI represents each plume's atmospheric dispersion index.

An example of the random effects model is:

$$WIDTH_{it} = \alpha_1 + \beta_1 DIST_{it} + \beta_2 ADI_i + u_i + \varepsilon_{it} \tag{4.5}$$

The difference between the two models is that the random effects model decomposes the error term into u_i which is the random disturbance for the i^{th} observation while ε_{it} represents the error within fires. The distribution around the u_i term is assumed to be normal with the variance estimated as a random parameter.

With the width estimated and distance accounted for, the final variable in the plume volume is mixing height. The average daily mixing height was used as an estimate for the height of the plume and is presented in Table 4.1. After determining the volume of the plume, the next step of the analysis is to estimate the concentration of particulate matter in the plume.

4.4.0 PARTICULATE MATTER CONCENTRATION ESTIMATES

A major component of this research is the estimate of the level of particulate matter at varying distances from a fire. The amount of particulate matter released into the atmosphere is determined by the spread rate of the fire, the amount of fuel burned per meter and the amount of particulate matter released for each kilogram of burned fuel. Dispersion variables including plume width, wind speed and mixing height, are used to estimate how the released particulate matter disperses as distance increases.

4.4.1 Fire Spread Rate

The 2001, 2002 and 2003 fire seasons were analyzed to estimate fire spread rate. Changes in fire size were calculated based on data from the Alberta Government's Wildfire Data Report. The use of daytime satellite images led to the calculation of daytime measurements for spread rate. Generally, weather and atmospheric conditions contribute to a high spread rate during the day as opposed to the spread rate at night. The average rate of spread was calculated using 133 observations from 45 fires. Observations were from fires that had been given an "out of control" classification. Fires of this type were assumed to have little influence from suppression efforts. Zero values and values above 15 ha/min were removed as outliers. A final mean value of 2.13 (SD 2.77) hectares per minute is estimated for daytime conditions. Most spread rates were calculated during the first few days of the fire as the data set did not provide multiple daily measurements during the later part of each fire. The average spread rate including overnight observations is 1.77 (SD 2.54) hectares per minute.

4.4.2 Fuel Consumption

Amiro et al. (2001) estimated the mean total fuel consumption from 10,771 fires in various ecoregions. The average consumption per fire for the Boreal plains eco-region was 2.35kg/m^2 (SD 0.99), while the Prairie eco-region was 1.09kg/m^2 (SD 0.74) The Canada-wide average is 2.6kg/m^2 (SD 1.00) The estimates for the Boreal plains eco-region is used in the calculation of $PM_{2.5}$ at various distances from the fire.

4.4.3 Particulate Matter Emission Factors

The United States Environmental Protection Agency's average $PM_{2.5}$ emission factor over all fire types is 8.5 kilograms of fine particulate matter released per mega gram of fuel consumed by the fire (United States Environmental Protection Agency and Office of Air Quality Planning and Standards, 1995). Variables that affect the emission factor, but are not explicitly controlled for are type of fuel consumed and fire intensity.

4.5.4 Formula for Amount of $PM_{2.5}$ per Cubic Meter of Air

While spread rate, fuel consumption and emission factors are used to estimate the amount of pollutant, width, wind speed and mixing height determine how the pollutant disperses in the atmosphere. Width varies linearly with distance by the relationship from Equation 4.2. The daily averages from each day of the fire were used as estimates for wind speed and mixing height. Tables 4.1 and 4.2 summarize the data used to calculate the concentration of $PM_{2.5}$ at a given distance from the starting point of the plume. The formula for the concentration of particulate matter is presented in Equation 4.3.

Table 4.1. Descriptive statistics of weather variables used in the Smoke Dispersion Model associated with 21 fire plume images from Western Canada.

Statistic	***Mixing Height (m)***[a]	***Wind Speed (km/hr)***[a]	**Atmospheric Dispersion Index**[b]
Mean	1324	18	40
Standard Error	203	2	8
Median	1067	18	34
Mode	407	6	2
Standard Deviation	931	9	35
Range	3393	3e	114
Minimum	294	5	2
Maximum	3687	38	116
Count	21	21	21
Confidence Interval (95.0%)	424	4	16

[a] Weather data provided by Environment Canada, Meteorological Service of Canada and Telesat Canada
[b] ADI provided by Natural Resource Canada

Table 4.2 Descriptive statistics of variables used in the Smoke Dispersion Model which contribute to particulate matter emissions.

Statistic	***Daytime Spread Rate (ha/min)***	***Fuel Consumption (Kg/ha)***[a]	***Emission Factor $PM_{2.5}$ (g/kg)***[b]
Mean	2.128	3340	13.5
Standard Error	0.240		
Median	0.964		
Standard Deviation	2.772	990	
Range	13.615		
Minimum	0.005		
Maximum	13.620		
Count	134	1390	
Confidence Interval (95.0%)	0.474		

[a] Amiro et al., 2001
[b] Battye and Battye, 2002

4.5.0 HEALTH EFFECTS OF FOREST FIRE SMOKE

To determine the health impacts from a specific level of particulate matter concentration the Air Quality Valuation Model version 3.0 (AQVM 3.0) is used. This computer model estimates human health effects and related costs or benefits resulting from changes in particulate levels. The model uses a damage function approach and assumes emissions released from forest fires change ambient air pollution. Using concentration-response relationships from current epidemiological research, a sensitivity analysis is conducted that shows the range of health impacts due to increased levels of particulate matter. Once the range of the health impact is estimated, the economic value of each health outcome is calculated. AQVM 3.0 combines costs-of-illness approaches, willingness to pay approaches and revealed preference techniques to value the direct and indirect health costs.

AQVM 3.0 provides estimates of the health impacts based on census subdivisions and the population characteristics of each census subdivision. Included in the model are demographic characteristics of the population. Based on national averages, AQVM 3.0 also estimates the number of people in a census division that have asthma as a preexisting condition. These demographic factors for the province of Alberta are combined with the health concentration-response information to generate estimates of the impacts of air quality changes on a variety of health categories. Included in the calculation of the impacts of short-term increases in particulate matter are the following health outcomes: premature mortality, childhood bronchitis, cardiovascular related hospital admissions, respiratory hospital admissions, restricted activity days, asthma symptoms and other acute respiratory symptoms. Only acute health problems are accounted for as it is assumed that the particulate matter concentration increases from forest fires are for a short period.

4.6.0 CONCLUSION

The purpose of this section has been to outline the methodology used to determine the level of particulate matter experienced by a population downwind from a fire and to estimate the monetary health effects from the increased particulate matter. Observations from satellite images were used to obtain a statistical relationship between distance from the start of the fire and plume width. The atmospheric dispersion index also helped to explain the width of the plume. Weather data at the time of the fire and burning characteristics were used to predict the particulate matter concentration levels in the plume. The introduction of the AQVM 3.0 computer model allows for a transition between the amount of increased particulate matter and the associated health and economic impacts.

5.0 RESULTS FROM SMOKE DISPERSION AND PARTICULATE MATTER CONCENTRATION ESTIMATIONS

5.1 Introduction

This section describes the results from statistical models of the relationship between distance from the fire, atmospheric dispersion, fire specific variables and the width of smoke plumes. Additionally, in this section, estimates of particulate matter concentrations contained in hypothetical smoke plumes are generated. The models predicting plume width and $PM_{2.5}$ concentration levels, collectively are termed the Smoke Dispersion Model. Estimation of plume width is used along with mixing height and wind speed to determine the volume of the plume. Dispersed in the plume are particulate matter emissions which are released during forest fires. Emissions are estimated with a fire spread rate variable, fuel consumption variable and an emission factor. Plume width, wind speed, the atmospheric dispersion index (ADI) and mixing height are all estimated from 21 fires and corresponding weather variables. There is a degree of uncertainty as to the accuracy of the estimates of these variables. Thus a sensitivity analysis is conducted using possible ranges of the variables. Their effect on the concentration of particulate matter is analyzed at varying distances from the source. Fire specific variables including rate of spread, fuel consumption, and emission factors also are not certain and are examined with a sensitivity analysis.

5.2 Estimation of Plume Width

Satellite smoke images were used to estimate the relationship between distance from the fire and the width of the fire's smoke plume. These components are required to describe the area of the plume and particulate matter concentrations at various distances. Other possible factors affecting plume width are the influence of adjacent fires and weather conditions such as wind speed, mixing height and time of day. Two types of models were estimated. An OLS model was estimated which assumed a constant intercept term while the random effects model accounted for the variation in intercept terms among the fires. Various specifications of the model are presented in Table 5.1.

5.2.1 Specification One

The ADI variable provides a numerical index for the dispersion characteristics at the time the image of the plume was taken. In specification 1 in Table 5.1 the ADI coefficient is positive and significant indicating that width will increase as dispersion characteristics increase. The constant term in specification 1 is 5.1 which represents an initial plume width of 5.1 kilometers. A Lagrange Multiplier (LM) test was performed to test for individual effects in the constant term predicting initial width. The null hypothesis is that the classical linear regression model, with a single constant term, is appropriate for the current data set. Based on the LM test statistic of 284.23 and a chi-squared critical value of 3.84, the result is to reject the null hypothesis and conclude that the random effects panel model is appropriate (Greene, 2000). While the LM test suggests that the model has individual group effects, the Hausman test for a fixed effects constant term versus a random effects constant term was also used. The hypothesis that the fixed effects model is appropriate is rejected with a test statistic 1.45 (p-value of 0.259). The critical value from a chi-squared table with three degrees of freedom is 7.814. Therefore, based on the LM test predicting individual effects and the Hausman test which suggests these effects are uncorrelated with other variables in the model, the conclusion is that the random effects model is appropriate for this data set (Greene, 2000). Specification 1, compared to the two other models estimated, provided the best overall explanation of plume width. The variables in the model are significant and consistent with *a priori* expectations that ADI and distance contribute to plume width.

5.2.2 Specification Two

A factor contributing to plume width is the presence of adjacent fires feeding into one plume. To account for the influence of multiple fires, a dummy variable with one indicating the presence of more than one fire and zero used to represent the presence of a single fire with a single plume was used. Specification 2 in Table 5.1 shows the results of the model. The multiple plume indicator was negative and not significant. The indicator variable was correlated with the atmospheric dispersion index thus the atmospheric stability index is used to account for dispersion characteristics. The stability index ranges from 1-7 with low values representing unstable atmospheric conditions causing smoke to disperse. The negative value on the coefficient suggests plume width decreases as stability conditions stabilize and dispersion decreases. Similar to Specification 1, the LM test rejects the null hypothesis that an OLS model is appropriate. The Hausman test suggests that the random effects model, in this case, is favoured over a fixed effects model.

5.2.3 Specification Three

It is possible that fire specific variables such as fuel amount, fuel flammability and fuel moisture content will increase either the fire intensity or spread rate. Both intensity and spread rate theoretically influence plume width. Indexes provided by Natural Resources Canada cover a range of weather and fire behaviour, however most are correlated and can not be estimated together. The initial spread index combines wind speed, predicted spread rate and moisture content variables. Table 5.1 shows the results of the initial spread index's influence on plume width. Consistent with the other two specifications the LM test and the Hausman test suggest that the random effects model is appropriate. It is expected that as the initial spread index increases the width of the plume will increase, as greater amounts of fuel are released. This was not the case; the initial spread index coefficient was negative and not significant for the OLS model and positive and insignificant for the random effects model.

Table 5.1. Results of linear regressions predicting plume width (km).

Dependent Variable: Plume Width (Km)	**Specification 1: Atmospheric Dispersion Index**		**Specification 2: Multiple Plume Indicator**		**Specification 3: Initial Spread Index**	
Independent Variables	**OLS Model (t-stat)**	**Random Effects (t-stat)**	**OLS Model (t-stat)**	**Random Effects (t-stat)**	**OLS Model (t-stat)**	**Random Effects (t-stat)**
Intercept	4.01*	5.1*	16.65*	21.40*	10.79*	6.94*
	(3.28)	(2.16)	(8.38)	(4.54)	(5.48)	(1.96)
Distance from Source (km)	0.28*	0.24*	0.27*	0.21*	0.28*	0.24*
	(9.13)	(9.60)	(9.07)	(9.85)	(7.82)	(9.83)
Atmospheric Dispersion Index (ADI)	0.15*	0.16*				
	(8.31)	(3.68)				
Stability Index			-1.37*	-2.42*		
			(-2.81)	(-2.00)		
Multiple Plume Indicator			-0.48	0.826		
			(-.377)	(0.47)		
Initial Spread Index					-0.14	0.35
					(-0.13)	(1.66)
Lagrange Multiplier Test Statistic		284.23 (p-value 0.00)		543.28 (p-value 0.00)		367.15 (p-value 0.00)
Fixed vs. Random Effects (Hausman)		1.45 (p-value 0.48)		2.53 (p-value 0.47)		5.19 (p-value 0.075)
Variance [u][a]		41.39		65.33		67.07
Variance [e][b]		33.24		31.27		33.06
Variance OLS	129.96		128.36		129.96	
R^2 X and Group Effects		0.77		0.78		0.77
R^2	0.43		0.27		0.24	

* Significant results (alpha =0.05, t=1.96)

[a]Variance [u] refers to the variance surrounding the individual fires

[b]Variance [e] refers to the variance surrounding the error term in the regression

5.3.0 SENSITIVITY ANALYSIS OF EMISSION CALCULATIONS

5.3.1 Introduction

Input variables either affect the volume of the plume or the amount of particulate matter released from the fire. An increase in the plume's volume will allow the amount of particulate matter released to be dispersed over a larger area, thus lowering the concentration level. Holding the plume volume constant, an increase in the amount of emissions will increase particulate matter concentrations. The baseline level of particulate matter is the background level of $PM_{2.5}$ in the atmosphere regardless of fire or pollution conditions. The average baseline level for Edmonton in 2001 was 12.55 $\mu g/m^3$ (CASA, 2004). Therefore particulate matter concentrations presented include particulates released from the fire as well as the baseline or background level of PM. The Canada Wide Standard for particulate matter is provided to give context to the concentration levels in the sensitivity analysis. Currently, the Canada Wide Standard for $PM_{2.5}$ is $30\mu g/m^3$ averaged over a 24 hour period. Table 5.2 describes the effect of each input variable on the concentration of particulate matter downwind from a fire. Equation 4.3, explained in Chapter 4, is shown below for reference purposes.

Equation 4.3: Micrograms of $PM_{2.5}$ per Cubic Meter of Air

$$= \frac{1}{WIDTH * WIND * HEIGHT} * ROS * CONSUMPTION * EMISSION\_FACTOR$$

Function of Distance

Table 5.2. The expected effect of an increase in weather and fire related variables on particulate matter concentration levels from forest fires.

Category of Variable	Effect on $PM_{2.5}$ Concentration Level
Increase in Plume Volume Variables	
Width	Decrease
Intercept (Random Effect Parameter)	Decrease
Distance	Decrease
ADI	Decrease
Wind Speed	Decrease
Mixing Height	Decrease
Increase in Emission Variables	
Fire Spread Rate	Increase
Fuel Consumption	Increase
Emission Factor	Increase
Increase in Baseline level of $PM_{2.5}$	Increase

5.3.2 Plume Volume and Particulate Matter Concentration Sensitivity Analysis

The sensitivity analysis presented in the following sections uses average estimates of mixing height, wind speed and atmospheric dispersion from 21 fires in Western Canada. Average values are also used for emission variables. Each variable is analyzed individually or together with other variables that affect either plume width or the prediction of emissions released. The average values used and presented in Table 4.1 are as follows: mixing height, 1324 m; wind speed, 17.91 km/hr; dispersion index, 40; fire spread rate, 2.13 ha/min; fuel consumption, 2.35 kg/m^2 and an emission factor of 13.5 g/kg. We hold all values constant at their mean values while looking at the range of effects over the confidence limits.

5.3.2.1 Atmospheric Dispersion Index (ADI)

Plume width is described as a function of distance and ADI. Low ADI values indicate poor dispersion. When dispersion characteristics increase and ADI increases, the width of the plume is estimated to increase by a factor of 0.16. The average value for ADI from the 20 fires measured is 40 (CI 24, 56). Holding all else constant, the ADI's contribution to the plume's width is between 4 and 9 kilometers for every kilometer increase in distance. Figure 5.1 shows the confidence interval for ADI and the associated impact on concentration levels. The graph indicates that the particulate matter concentrations generated by the base fire are relatively low and fall below the Canada Wide Standard within 30 kilometres of the fire for the base value of ADI=40. For lower values of ADI, ADI=24 or the lower end of the confidence interval, concentration is increased at any given distance from the fire, but the magnitude of the change is relatively small. Reviewing the higher end of the confidence interval, ADI=56, shows a small increase above the baseline particulate matter level (12 μg/m^3) with concentration levels well below the Canada Wide Standard.

5.3.2.2 Wind Speed and Mixing Height

Wind speed describes how quickly the wind disperses particulate matter. An increase in wind speed will lower particulate levels. Figure 5.2 shows the base level and associated confidence interval for wind speed. In this case, the base fire falls below the Canada Wide Standard at approximately 30 kilometers. Analyzing the lower confidence interval for wind speed (10 km/hr) shows concentration levels to stay above the Canada Wide Standard for an additional 15 kilometres beyond the prediction from the base fire. Estimation using the higher confidence interval for wind speed shows little affect on concentration levels. Mixing height is similar to wind speed in that an increase in either variable will cause a linear decrease in concentration levels. Mixing height is the height at which smoke travels vertically. It is assumed that the smoke will not penetrate the maximum mixing height, if however it were to penetrate this layer concentration levels would be lower. Figure 5.3 shows the confidence interval and base case for mixing height. Results are similar to the analysis from the ADI and wind speed. The base fire case, by definition, is the same for all variables while the lower confidence interval for mixing height (900 m) causes particulate matter levels to fall below the Canada Wide Standard before 50 kilometers.

5.3.2.3 Constant Term

The random effects model described in Equation 4.2 predicts the distribution of the constant term which represents the values surrounding the starting width value at a distance of zero. The constant term affects concentration levels by increasing or decreasing the width variable in Equation 4.3. Therefore, the constant term does not affect concentration levels directly rather it is part of the plume width estimation. The standard error for the constant term is 2.36 kilometers, and its affect on concentration levels is shown in Figure 5.4. Confidence intervals surrounding the base value of 5.1

kilometers show little contribution to concentration levels.[2] The higher confidence interval (7.46 km) causes concentrations to fall below the Canada Wide Standard at 35 kilometers with the lower confidence interval (2.75 km) causing concentrations to fall below the standard at 15 kilometers.

5.3.2.4 Summary of Plume Volume and Effect on Particulate Matter Concentrations

An increase in any of the volume variables; width, wind speed or mixing height, reduces particulate levels downwind from a fire. Figure 5.5 shows a confidence interval for all the variables that affect plume volume. In this scenario particulate levels will fall below the Canada Wide Standard at a maximum distance of 95 kilometers, for the values associated with the lower confidence intervals. Examining the higher confidence intervals for each variable shows that these values have little influence on concentration levels. The levels become close to the baseline level of $12 \mu g/m^3$ fairly quickly. It is important to note that the values for ADI, wind speed and mixing height were averaged during 20 fire occurrences and are not seasonal averages. Fires are more likely to occur under specific weather conditions, thus it is likely that the variables for ADI, wind speed and mixing height are correlated. It is probable that if one of the variables is conducive to fires occurring then the other variables will also be in the range that are favourable to forest fires.

5.3.3 Sensitivity Analysis of Particulate Matter Emissions from Wildfires

Variables estimating the grams of particulate matter released from a fire include fire spread rate, kilograms of fuel consumed and the grams of particulate matter released per hectare burned. Fuel consumption and emission factors are obtained through a literature review while the fire spread rate was estimated from 45 individual fires in

---

[2] The overall variance was also tested to examine the potential contribution to concentration levels. The standard deviation for the random effects model is 6.4 km. Taking one standard deviation from either side of the mean results in negative values which is not a possible outcome for initial width estimation. Therefore only the confidence interval with regards to the intercept term was analyzed.

Alberta. An increase in any one of the three variables in this section will cause particulate matter levels to increase.

5.3.3.1 Fire Spread Rate

Figure 5.6 shows the confidence interval for fire spread rate. Fire spread rate is calculated during the daytime due to the daytime measurements of other variables and because plume images were only available during the day. Generally, fires burn quicker during the day than at night. For comparison, the average over all spread rates is added to Figure 5.6. Another variation in spread rate occurs at the start of the fire compared to when it is closer to burning out. The spread rate for this analysis was used from the first few days of a fire. Results from Figure 5.6 show that the difference between the daytime spread rate value of 2.13 ha/min and the 24-hr average spread rate value of 1.77 ha/min contributes to only a 5 kilometer difference in the distance it takes for concentration levels to fall below the Canada Wide Standard. The higher confidence interval for the daytime spread rate (2.6 ha/min) indicates that concentration levels will fall below the Canada Wide Standard at 35 kilometers while the lower confidence interval (1.65 ha/min) causes concentration levels to fall below the standard at 15 kilometers.

5.3.3.2. Fuel Consumption and $PM_{2.5}$ Emission Factors

The research by Amiro et al. (2001) was used to determine fuel consumption rates. Figure 5.7 shows the effect of average fuel consumption rates, with a 0.99 kilogram per hectare standard deviation, on particulate matter concentrations. Similar to fire spread rate, the higher value used for fuel consumption (3.34 kg/m^3) increases the distance it takes for particulate matter levels to fall below the Canada Wide Standard, but only by a distance of 15 kilometers. Emission factors estimate the kilograms of $PM_{2.5}$ released per unit of fuel consumed. Both the fuel consumption estimates and emission factor estimates are dependent on fuel type and fire intensity, which are not included in the model. Figure 5.8 shows the variations in $PM_{2.5}$ emissions depending on the range of input values for the fire spread rate and fuel consumption. The values used for the lower end of the sensitivity analysis indicate that if these were the true conditions then there is little concern that concentration levels will reach the Canada Wide Standard even at close distances. However, the values used in the higher end of the sensitivity analysis indicate concentration levels will stay above the standard for a distance of 40 kilometers.

The contribution of each variable from Equation 4.3 to particulate matter concentration levels is relatively small. There is not one variable that has a dramatic influence on concentration levels and if one variable were to deviate from the average values used in the base fire case it will not change concentration levels much. However, the incidence of more than one variable changing in a way that increases concentration levels creates concern. The next section summarizes the concern by presenting a best and worst case scenario.

5.3.4 Best and Worst Case Scenarios

While examining variables individually shows the contribution of each variable to the particulate matter concentration level, it is interesting to show the impact of changes in all the variables simultaneously. To this end a worst case and best case scenario is developed based on the confidence intervals of the input variables. The worst case scenario uses values that contribute to a small dispersion area, but high emission releases. In contrast, the best case scenario is developed using values of variables that contribute to a large dispersion area and low emission rates.

To generate the worst case scenario the following values were used: mixing height, 900m; wind speed, 10.1 km/hr; ADI, 24; spread rate, 1.65 ha/min; fuel consumption, 1.36 kg/m^3 and an emission factor of 13.5 g/kg. Additionally, to generate the best case scenario the following values were used: mixing height, 1747m; wind speed 25 km/hr; ADI, 56; spread rate, 2.60 ha/min; fuel consumption, 3.34 kg/m^3 and an emission factor of 13.5g/kg. The values for mixing height, wind speed and ADI represent lower and upper bounds of confidence intervals observed from the fires used in the statistical analysis. The confidence interval for spread rate was obtained from an average of 45 fires in Alberta, while fuel consumption and emission factor confidence intervals were obtained from current literature. At a given distance from the fire, the worst case scenario will have a much higher particulate matter concentration than the best case scenario. Figure 5.9 shows the results of the best and worst case scenarios. The scenarios provide an indication of the range of human health impacts from fires. The distance it takes for concentrations to drop to the Canada Wide Standard is between zero kilometers and 165 kilometers for the best case and worst case scenarios respectively.

Figure 5.1. Sensitivity analysis of the influence of the Atmospheric Dispersion Index (ADI) on particulate matter concentrations.

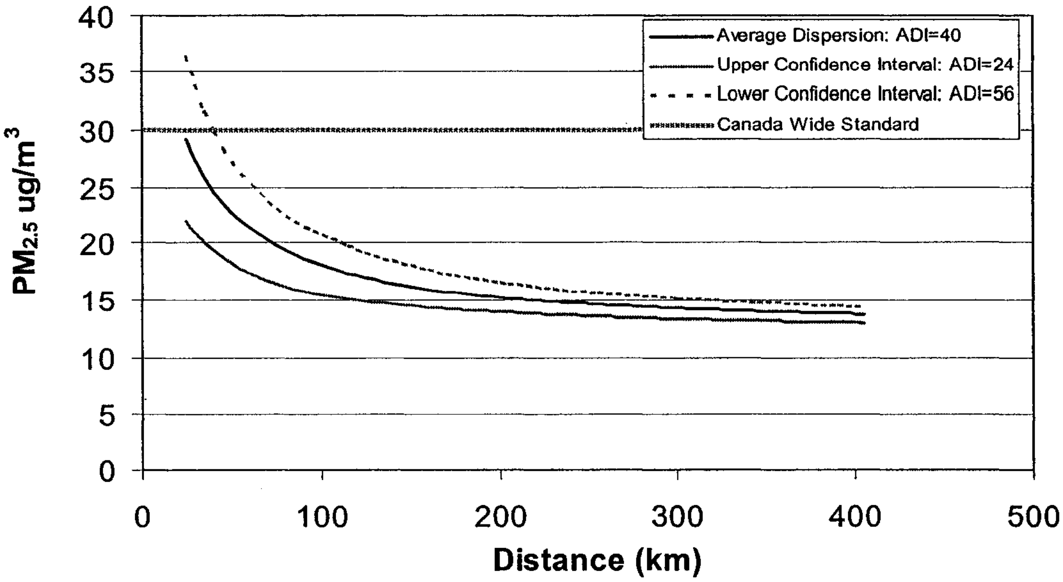


Figure 5.2. Sensitivity analysis of the influence of wind speed on particulate matter concentrations.

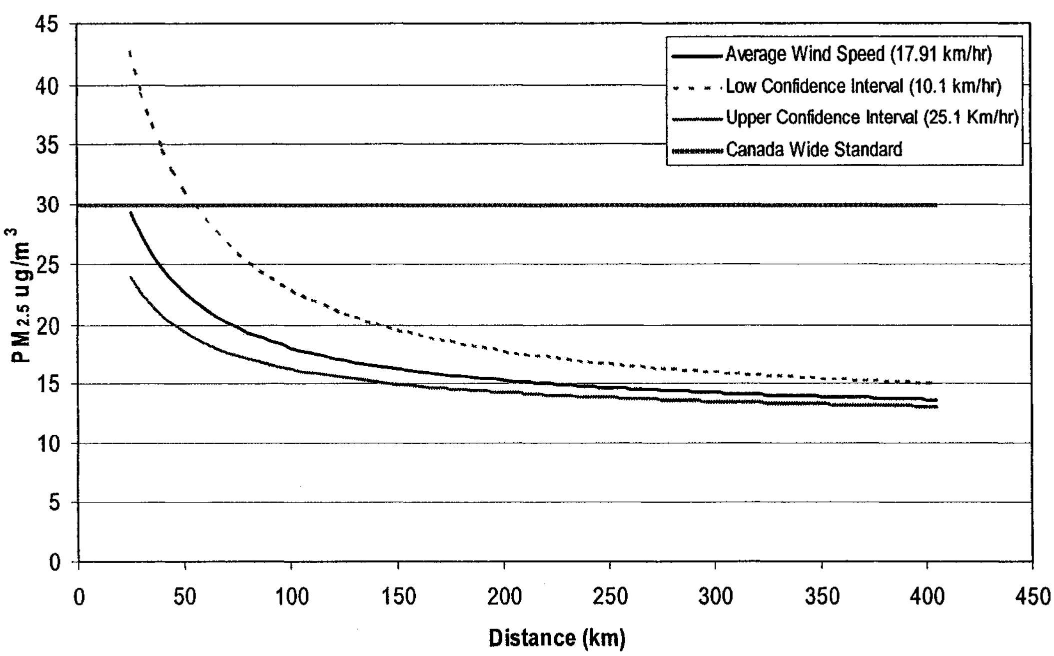


Figure 5.3. Sensitivity analysis of the influence of mixing height on particulate matter concentrations.

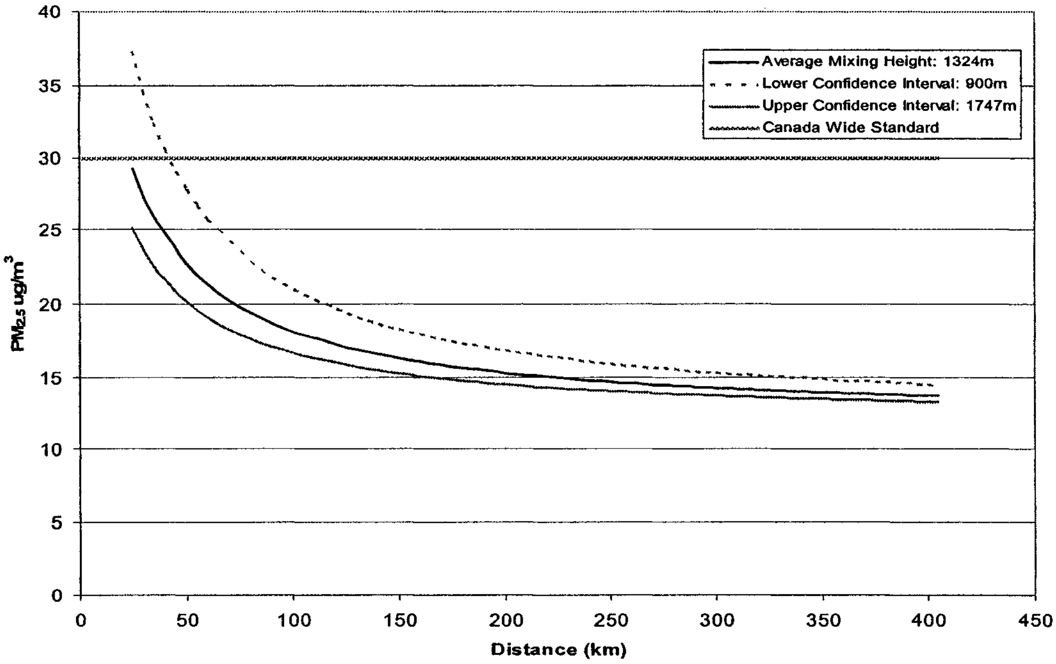


Figure 5.4. Sensitivity analysis of the influence of the variation in the constant term on particulate matter concentrations.

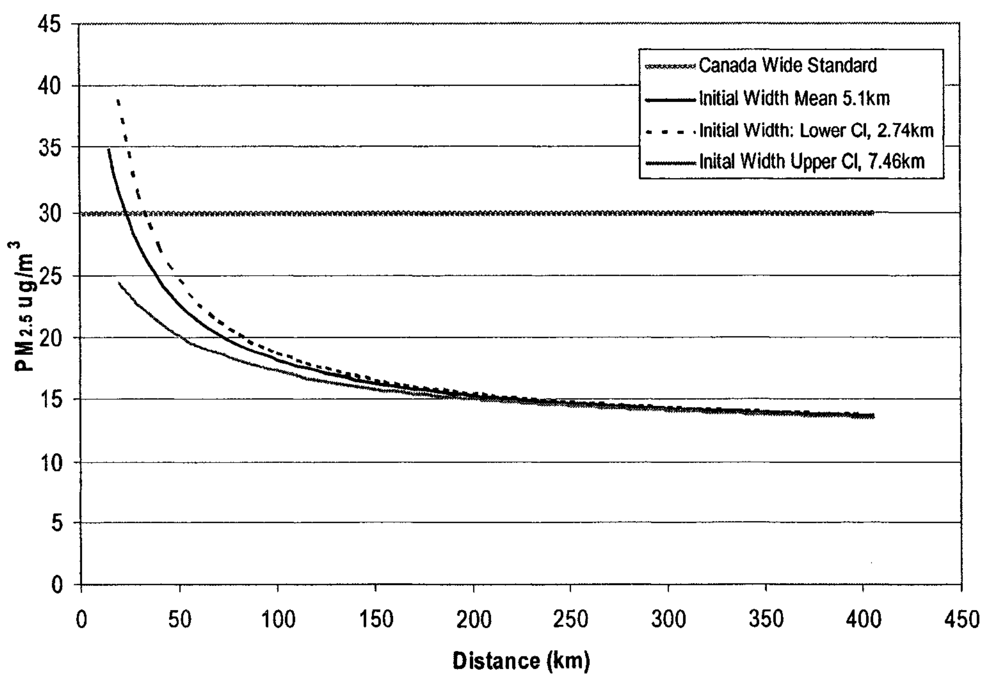


Figure 5.5. Sensitivity analysis of plume volume variables (wind speed, mixing height and ADI) on particulate matter concentrations.

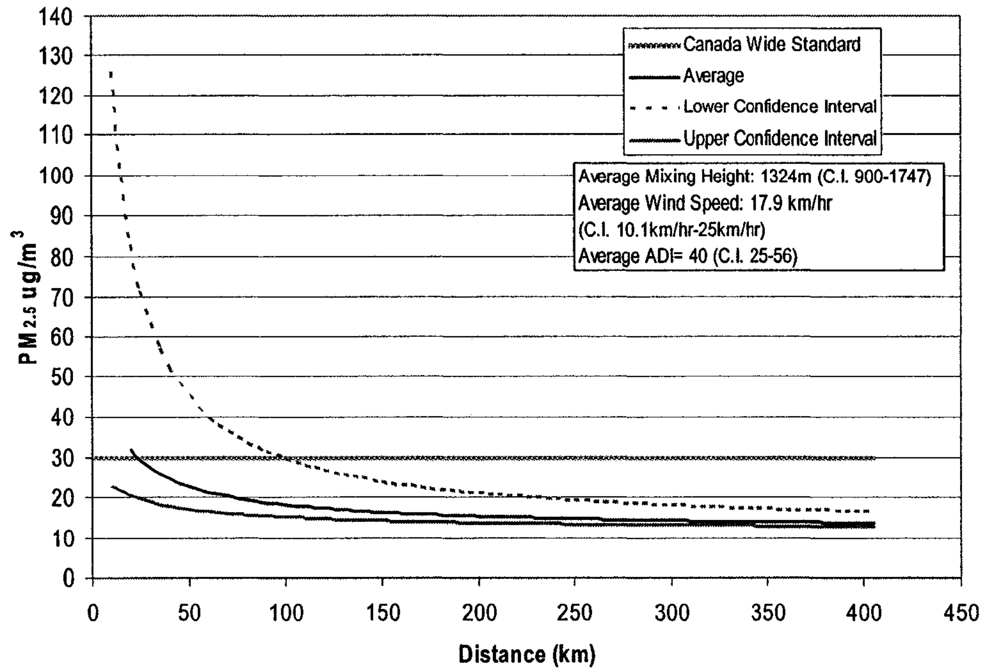


Figure 5.6 Sensitivity analysis of the influence of fire spread rate on particulate matter concentrations.

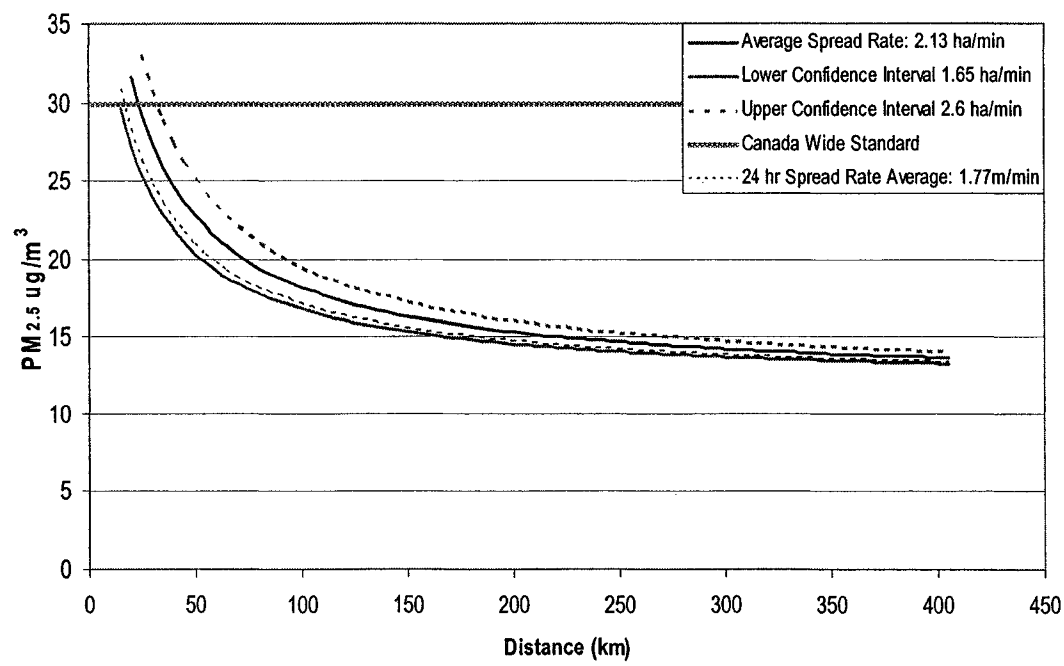


Figure 5.7 Sensitivity analysis of the influence of fuel consumption on particulate matter concentrations.

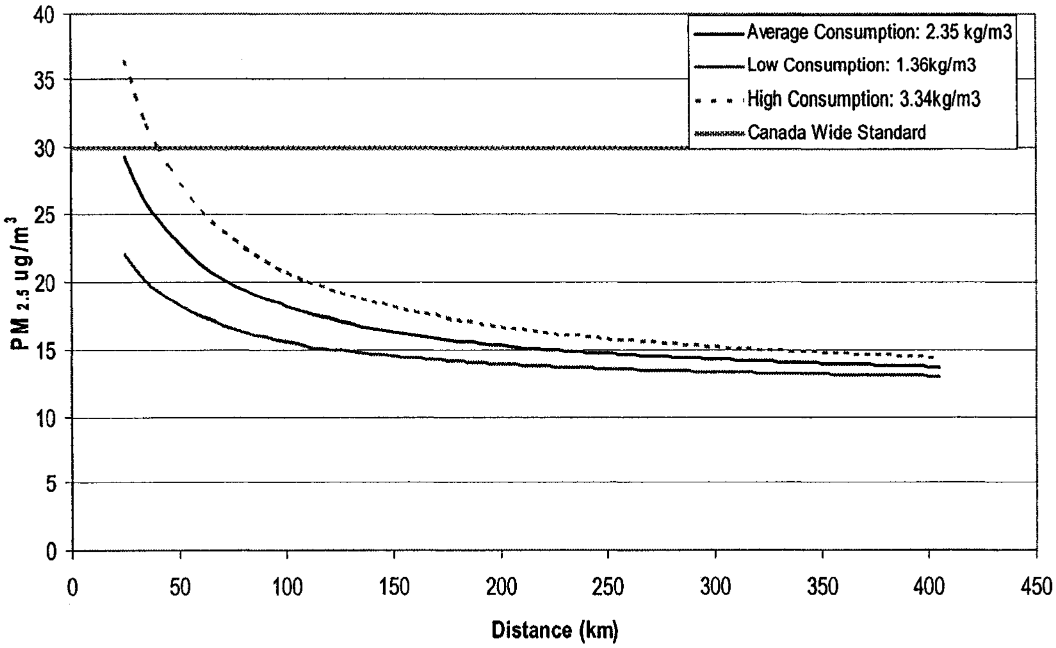


Figure 5.8. Sensitivity analysis of the influence of particulate matter emission variables (fire spread rate and fuel consumption) on concentration levels.

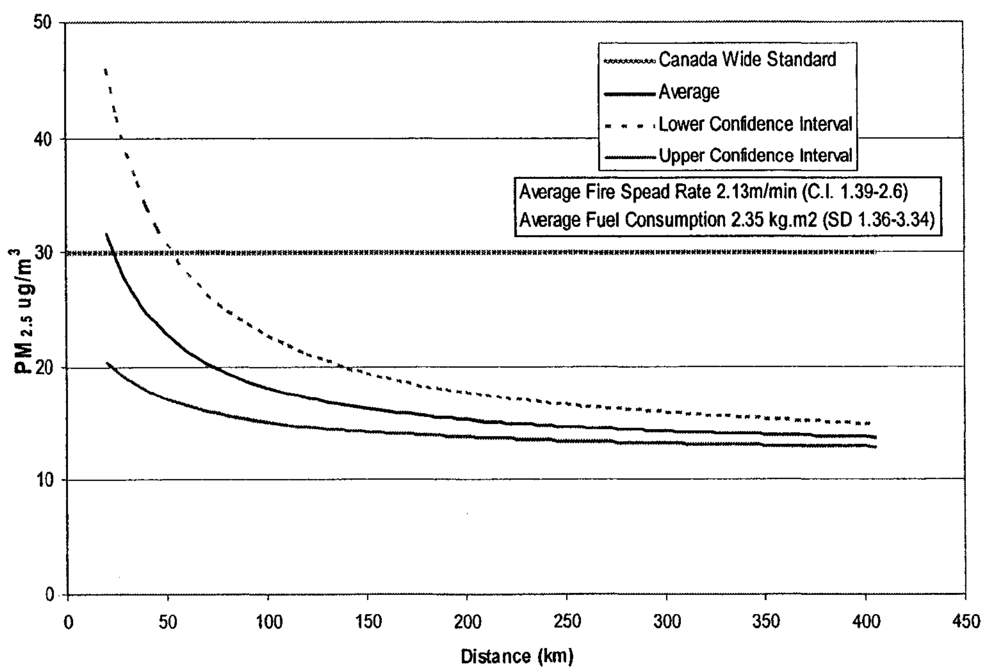
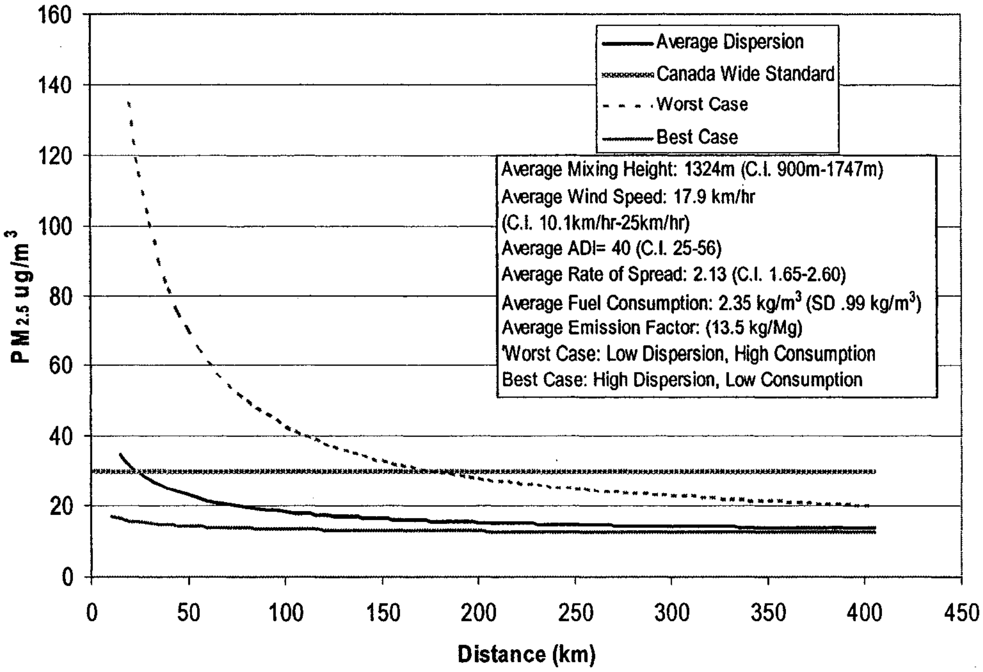


Figure 5.9. Comparisons of particulate matter concentrations for the best case and worst case scenarios



5.4 DISCUSSION

Many of the factors that determine smoke dispersion are not captured in the Smoke Dispersion Model. In terms of plume width, it is reasonable to assume that the variables that affect width vary throughout the day and over the distance of the plume. Mixing height for example ranges between values below 100m in early morning and evening to midday values above 1500m. A dynamic computer model, allowing for a continuous flow of variables over time and space, may improve estimation. Weather variables for wind speed, mixing height and ADI are limited because of the lack of weather stations. Ideally weather stations would be located at the location of the fire and at varying intervals away from the fire. An additional limitation of the model is that weather variables and the prediction of width are based on 20 fires in Alberta. The large range of each value and small sample size contribute to estimations with large standard deviations. Fire spread rate is an example where the standard deviation is as large as the estimate itself. While it is possible to have negative fire spread rates, especially during fire suppression efforts, the large standard deviation is more likely driven by a small sample size. Optimally more fires should be included in the analysis, however, satellite images used to predict plume width are difficult to obtain and analyze.

The variables estimating emissions are probably dependent on location-specific factors for fuel type and fire intensity, which are not included in the statistical model. However, their influence is accounted for in the random effects terms, but not explicitly. The 45 fires studied to estimate fire spread rate were from Alberta and can be assumed to contain an average of fuel types in Alberta. A more comprehensive model would allow for variations based on fuel type. Fuel consumption estimation is from the boreal forest of Alberta, where many fires occur. $PM_{2.5}$ emission factors are not Alberta specific, but are averaged over all fuel types. Overall, it is not recommended that the smoke dispersion model be used, outside of Alberta.

5.5 SUMMARY OF FINDINGS AND POTENTIAL POLICY APPLICATIONS

The sensitivity analysis described previously shows the changes in particulate matter levels downwind from a fire based on ranges of weather and fire specific variables. Predicting concentration levels is best presented as a possible range of values because of the uncertainty of each variable, but also because weather conditions change throughout the duration of a fire. The difference between the distance it takes for concentration levels to fall to the Canada Wide Standard for the best and worst case scenario is 165 kilometers. This large distance suggests that under certain conditions the concentrations of particulate matter may not be a large factor in fire management decisions. The best use of the model for forest mangers responsible for fire control will be to use scenario analysis to predict when particulate levels will reach dangerous levels. Generally this will be when there are high emissions rates and low dispersion characteristics combined with a wind direction blowing smoke towards large population centers. If concentration levels are expected to be high, managers may decide to increase suppression and control resources in an effort to contain particulate matter concentrations.

A notable finding from comparing the sensitivity analysis of all the weather and fire variables is that there is not one variable specifically that can be highlighted as being the largest contributor to particulate matter concentration levels. Rather, it is the combination of variables that contributes to dangerous levels. Future research should focus on the probabilities of having conditions that are similar to the worst case scenario of low dispersion and high emissions. Additionally, it would be helpful to focus research on the correlation between variables. When the dispersion index is low it may be that wind speed will also be low contributing to adverse conditions.

The policy challenges related to emissions from forest fires are not isolated to forest managers. While forest managers have some control over fire spread rate, public health educators can influence the behaviour of the population at risk from the fire's emissions. When weather conditions near a fire are predicted to have low dispersion characteristics public advisories can be sent to populations at risk. With the ultimate goal of lowering health impacts as cost-effectively as possible, it may be that targeting specific populations is more efficient than increasing fire suppression efforts. It would be beneficial for policy makers to know the fire fighting costs per unit reduction in particulate matter. This can then be compared to the health benefits or the costs of defensive measures taken by the individuals to avoid exposure.

By overlaying the Smoke Dispersion Model onto an actual population, the health effects and associated costs can be estimated. The next section uses a recent fire in Alberta as a case study to show the use of the dispersion model with the Air Quality Valuation Model (AQVM 3.0) to estimate the monetary health costs related to increases in particulate matter levels.

6.0 CHISHOLM CASE STUDY: ONE DAY ESTIMATE OF HEALTH IMPACTS

6.1 INTRODUCTION

This chapter presents a case study that assesses the economic impact of air quality changes arising from forest fires. The human health effects from a change in air quality, arising from the Chisholm Fire (May 2001), are estimated along with the economic values of these health impacts. The Smoke Dispersion Model, developed previously, is used to predict concentration levels downwind from the fire. These levels are then compared to actual $PM_{2.5}$ levels taken from monitoring stations. The population of each area exposed to the plume is determined from Canadian census data while the resulting health effects are valued using the AQVM 3.0 model. Our approach does not answer the question of whether efforts on fire suppression were beneficial in an economic sense. However, the information on air quality effects will be an important element in assessing the return on investment from fire suppression.

6.2 BACKGROUND

Chisholm is located 150 kilometers north of Edmonton, Alberta. While there are a few population centers between Chisholm and Edmonton, Edmonton was the largest city (population approximately 600,000 in 2001) affected by the increase in $PM_{2.5}$ levels. The Chisholm Fire burned from May 23^{ed} to May 29th, 2001. In total, the fire burned 116,000 hectares of forest land and burned structures in the town of Chisholm and surrounding settlements and infrastructure. On May 28th, 2001 a state of emergency was declared for the Chisholm region. Wind direction during the Chisholm fire blew the smoke away from the larger population centers, however on May 24th wind was blowing in a south-east direction directly towards the city of Edmonton. This case study models smoke dispersion from the Chisholm Fire and associated health effects on May 24th, 2001.

6.3 METHODS FOR ESTIMATION WITH THE SMOKE DISPERSION MODEL

6.3.1 Input Variables

The weather during the fire was characterized by a high temperature of 27°C, low humidity and winds gusting to 50km/h (CFEI, 2001). These extreme weather conditions combined with a dry, flammable forest contributed to the fire event (CFEI, 2001). The Slave Lake weather station, approximately 70 kilometers from the fire, is the closest station providing weather data. Other weather stations include Whitecourt (130 kilometers away) and Lac la Biche, which is 120 kilometers away. Data from stations near lakes may be distorted by lake effects, thus data averaged over the three weather stations is used to estimate weather conditions during the fire. ADI, wind speed and mixing height are used in the Smoke Dispersion Model as input variables predicting the volume of the smoke plume. Fire spread rate is calculated from wildfire data reports, while the fuel consumption and emission factors are obtained from current literature.

The average ADI value on May 24th, 2001 between the three weather stations at noon was 100, while the average daily ADI from the weather stations was 27. Values for ADI are lowest in the mornings and highest at noon. Similar daily patterns for mixing height were observed with the highest values occurring midday. The average daily wind speed from all three weather stations was 15 km/hr. For this case study the daily averages over the three weather stations are used for ADI (27), mixing height (806 m) and wind speed. Fire spread rate was obtained from an Alberta Sustainable Resource Development Wildfire data report. Two spread rate measurements were taken on the second day of the fire (May 24th, 2001) with an average between the two measurements totalling 5.65 hectares per minute. Figure 6.1 shows the predicted path of the smoke plume from the Smoke Dispersion Model. The dimensions of the plume are overlaid onto a map of Alberta census subdivisions that will be used to estimate the number of individuals exposed to the plume.

Figure 6.2 shows the predicted concentration level at increasing distances from the fire. Presented in the graph are estimates of concentration levels depending on the random effects parameter in the estimation of plume width (Equation 4.2). The parameter, which represents the plume's width at distance zero, ranges between 2.74 kilometers and 7.46 kilometers.

6.3.2 Estimation of Population Exposed to Increased $PM_{2.5}$ Levels

A geographical information system computer program (ArcView), allowed for the dimensions of the smoke plume to be overlaid onto a map of census subdivisions (see Figure 6.1). Therefore, the specific census subdivisions exposed to the smoke from the Chisholm fire were evaluated. Population and demographic information relating to the population of each census subdivision was used in AQVM 3.0 to account for the number of individuals affected and age-specific health factors. The Smoke Dispersion Model estimates $PM_{2.5}$ concentration levels contained in the plume. Figure 6.2 shows the predicted concentration levels at various distances from the fire. For comparison, the predictions from the worst case scenario described in Chapter Five are included.

In terms of plume dimensions, it is assumed that once the plume's $PM_{2.5}$ concentration level falls below the Canada Wide Standard of $30\mu g/m^3$, individuals are no longer exposed and consequently the end of the plume is defined. The assumption was made because it is unlikely that the smoke plume will maintain its original shape and direction at long distances from the source. Allowing concentrations to fall to the baseline level to estimate the plume's end creates an unrealistic plume size especially in the worst case type scenarios. Therefore, the approach provides a conservative estimate from health impacts as some individuals that are exposed to levels below $30\mu g/m^3$ are not included in the analysis. Typically, $PM_{2.5}$ concentration levels in Alberta are approximately $12\mu g/m^3$ and as mentioned in Chapter Two any increase in $PM_{2.5}$ levels is associated with adverse health effects. However, in the calculation of adverse health effects from individuals exposed to the plume, it is assumed that any increase in concentrations above baseline contributes to health problems (even if the level is below

30μg/m^3). In other words for populations exposed there is assumed to be no minimal threshold level at which individuals do not experience adverse health impacts.

Figure 6.1 Predicted smoke dispersion from the 2001Chisholm Fire (May 24th) Alberta, Canada.

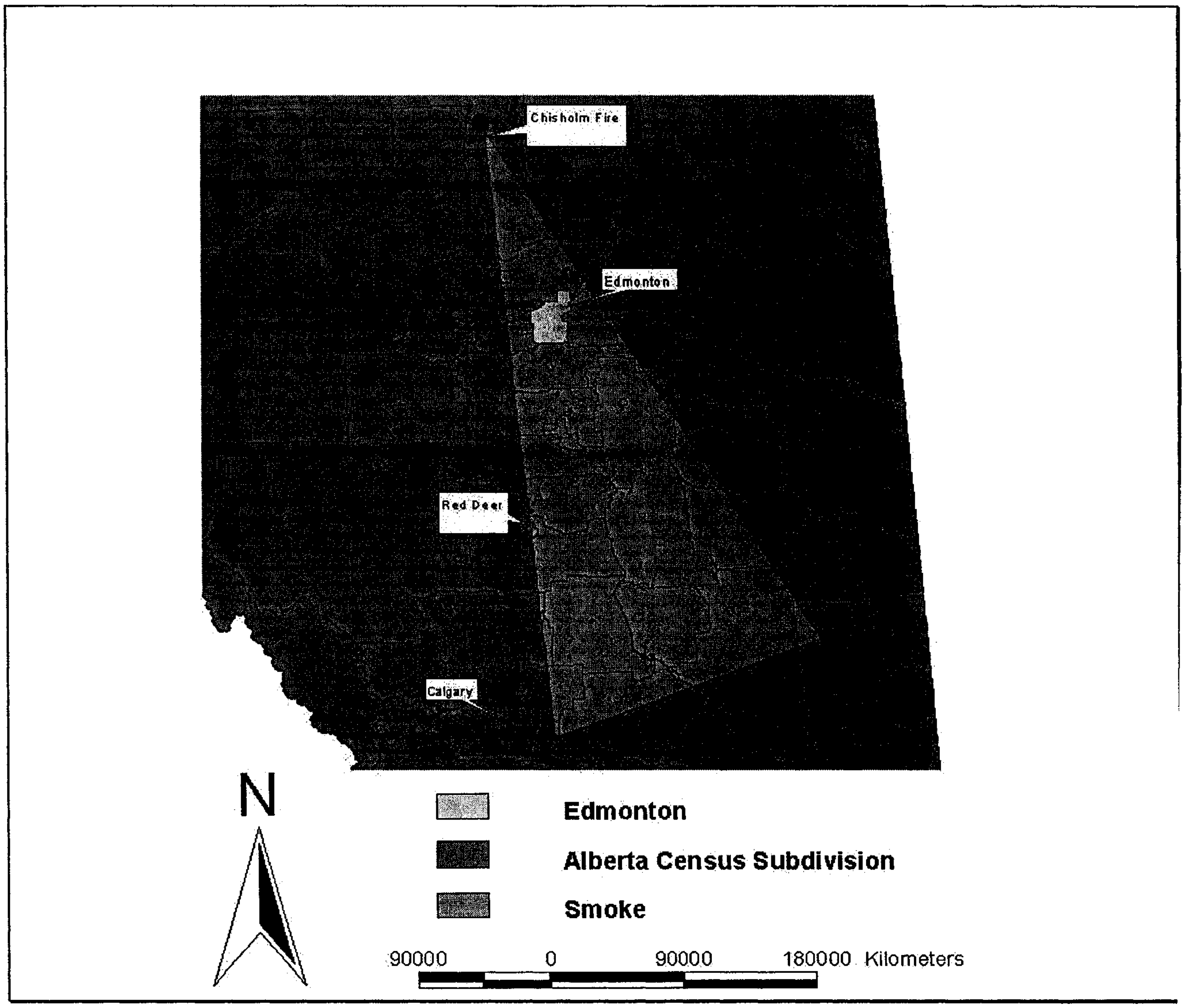
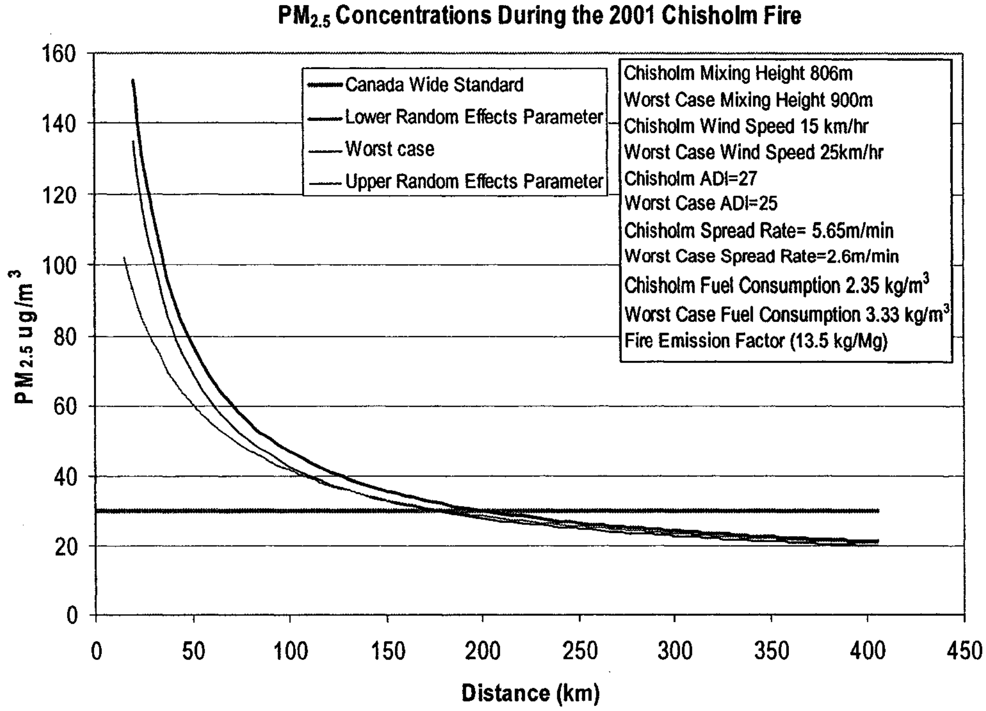


Figure 6.2. Concentration levels during the Chisholm Fire compared to concentration levels for the worst case scenario.



6.4.0 ECONOMIC IMPACTS OF THE CHISHOLM FIRE

6.4.1 Results from the Smoke Dispersion Model

The total population exposed to the smoke plume on May 24th 2001 was 1.1 million people. Edmonton is the largest city affected, containing approximately 600,000 individuals. The predicted exposure level in Edmonton on May 24th, 2001 is 35μg/m^3. Table 6.1 summarizes these impacts. A sensitivity analysis, developed from Table 3.2 and Table 3.4, is presented in Table 6.1. The values used in the sensitivity analysis are based on a range of estimates from concentration-response functions and a range of monetary values associated with each health outcome. Total values of health impacts ranged from $4 million to $19.4 million. As expected, the largest component of the total health impact was related to the increase in premature mortality risk. Indirect costs due to lost productivity and pain and suffering are the next largest impacts. Direct medical costs, which include lost wages, were a small component of the total health value lost from the Chisholm Fire.

Table 6.1. Sensitivity analysis of health related impacts from the Chisholm Fire due to increased $PM_{2.5}$ levels determined by the Smoke Dispersion Model on May 24th, 2001.

Health Outcome	Economic Impact			Percent of Total
	Upper Estimate[a]	Mean	Lower Estimate[a]	
Premature Mortality Risk	$18,642,146	$9,617,474	$3,789,410	94.93%
Respiratory Hospital Admissions	$4,666	$3,150	$1,571	0.03%
Cardiac Hospital Admissions	$5,057	$3,394	$1,686	0.03%
Emergency Room Visits	$1,899	$1,270	$640	0.01%
Restricted Activity Days	$467,687	$302,121	$150,030	3.04%
Asthma Symptom Days	$64,626	$28,230	$6,502	0.29%
Bronchitis Admissions	$12,711	$7,784	$2,930	0.08%
Acute Respiratory Symptom Day	$229,681	$148,488	$69,348	1.58%
Total	$19,428,471	$10,111,911	$4,022,115	

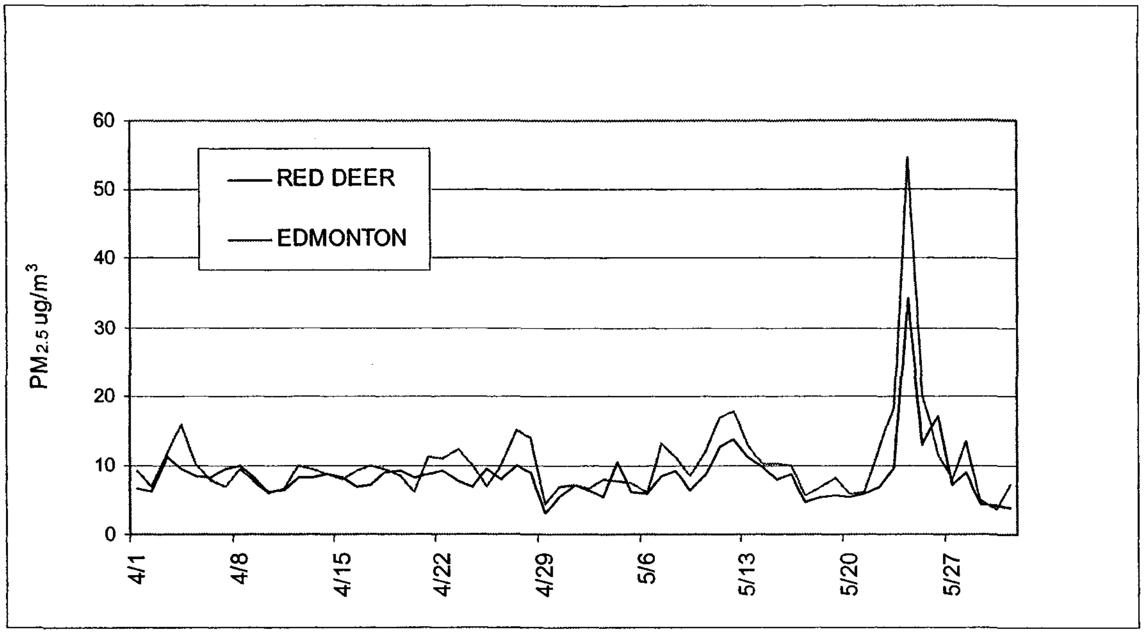
[a] The upper and lower estimates coincide with the 10th and 90th percentile from a distribution of results generated by the AQVM 3.0.

6.5.2 RESULTS FROM MONITORING STATION CALCULATIONS

Instead of using the Smoke Dispersion model to predict concentration levels, an alternative approach is to measure particulate matter levels from monitoring stations. These measurements were obtained from the Red Deer and Edmonton Central weather stations during the second day of the fire (May 24^{th}, 2001). The stations measure hourly particulate levels which were used to calculate a daily average. The effect of the fire is obvious from the peak in the graph of $PM_{2.5}$ levels over time (Figure 6.3). To be conservative the daily $PM_{2.5}$ averages are used, rather than hourly levels which are much higher.

On the second day of the fire the Edmonton central monitoring station reported a 24 hour average of $55\mu g/m^3$ while in Red Deer the reported average was $35\mu g/m^3$. The health impacts from the increases were estimated using the AQVM 3.0. with the results presented in Table 6.2. Aggregate impacts on health from a one day increase of particulate matter totalled \$12.1 million. The bulk of the estimate comes from premature mortality risks with other contributions from chronic bronchitis, restricted activity days and acute respiratory symptoms. A sensitivity analysis is also presented using the probabilities of the estimates provided within AQVM 3.0 and shown in Tables 3.2 and 3.4. These results range from \$4.9 million to \$22.9 million.

Figure 6.3. Average daily $PM_{2.5}$ concentrations during the 2001 Chisholm Fire.



*Data extracted from the Clean Air Strategic Alliance data download center. (www.casadata.org).

Table 6.2. Values of health impacts related to increased $PM_{2.5}$ emissions estimated from monitoring stations in Edmonton and Red Deer.

Health Outcome	Economic Impact		
	Upper Estimate	Mean	Lower Estimate
Total	$22,985,300	$12,137,043	$4,978,746
Premature Mortality Risk	$21,955,350	$11,326,764	$4,462,894
Respiratory Hospital Admissions	$5,494	$3,709	$1,850
Cardiac Hospital Admissions	$5,955	$3,997	$1,985
Emergency Room Visits	$2,235	$1,495	$753
Restricted Activity Days	$888,889	$574,213	$285,147
Asthma Symptom Days	$76,112	$33,246	$7,657
Bronchitis Admissions	$30,496	$18,676	$7,028
Acute Respiratory Symptom Day	$270,501	$174,878	$81,673

[a] The upper and lower estimates coincide with the 10^{th} and 90^{th} percentile from a distribution of results generated by the AQVM 3.0.

Table 6.3. Mean values of health impacts related to increased $PM_{2.5}$ levels from the Chisholm Fire on May 24th, 2001.

Health Outcome	Mean 1996$	
	Smoke Dispersion Model[a]	Monitoring Station[b]
Premature Mortality Risk	$9,617,474	$11,326,764
Respiratory Hospital Admissions	$3,150	$3,709
Cardiac Hospital Admissions	$3,394	$3,997
Emergency Room Visits	$1,270	$1,495
Restricted Activity Days	$302,121	$574,213
Asthma Symptom Days	$28,230	$33,246
Bronchitis Admissions	$7,784	$18,676
Acute Respiratory Symptom Day	$148,488	$174,878
Total	$10,111,911	$12,137,043

[a] Population exposed: 1.1 million. Minimum exposure level 30µg/m^3

[b] Population exposed 670,000. Exposure levels: Edmonton: 55µg/m^3, Red Deer: 35µg/m^3

Table 6.4. Non-health costs from the 2001 Chisholm Fire.

Category	Costs
Loss in Timber Supply	$19,800,000[a]
Fire Fighting Costs	$10,000,000[b]
Lost Homes and Buildings	$2,100,000[b]
Lost Bridges	$2,000,000[b]

a. Calculations assume a 5% discount rate over 100 years, price of $20 per cubic meter for an annual allowable cut loss of 50,000 cubic meters.

b. Chisholm Fire Review Committee, 2001.

The differences between the two sets of values in Table 6.3 are due to the number of individuals exposed and the differences in predicted $PM_{2.5}$ levels in the exposed communities. Monitoring stations were only available for Edmonton and Red Deer while the Smoke Dispersion model allowed for estimations in all the communities exposed to smoke from the fire. The calculation of health impacts, in both cases, is evaluated with the AQVM 3.0. The predicted concentration level in Edmonton, from the Dispersion Model is $35\mu g/m^3$ while the monitoring station in Edmonton measured levels at $55\mu g/m^3$. If only the levels in Edmonton are compared, the Smoke Dispersion Model under-predicts health impacts because estimates are based on a lower exposure level. However, the Smoke Dispersion Model covers communities that do not have monitoring stations. The inclusion of these communities increases the estimate of the number of individuals exposed which contributes to a greater overall estimation of health impacts.

6.6 DISCUSSION

The results from two models predicting health impacts show the uncertainties in the estimates. The range between the central values estimated between the two models is between $9 and $12 million. Approximately 95% of the impacts are related to the increases in mortality risk. Direct medical costs account for a small percentage of the adverse impacts from $PM_{2.5}$ increases. The overall health impacts are significant especially when compared to other costs of interest such as fire fighting costs.

To provide context to the health impacts, Table 6.4 shows other estimated costs of the Chisholm fire. The fire fighting costs, timber values and infrastructure costs are the total estimated costs from the fire while the health costs reported are for one day during the fire. Fire fighting costs were reported to be $10 million over the seven day fire which is close to the one day total value of the health impacts. Timber supply lost during the fire was estimated at $20 million while homes, building and bridges were a combined $4 million.

The health impacts presented are probably an underestimate of the true health effects related to forest fires because only the costs related to $PM_{2.5}$ have been evaluated. PM_{10} and other emissions may also contribute to adverse health effects. Additionally, individuals outside of the study areas may experience illness as well, but are excluded from the analysis.

Ideally, particulate matter levels would be measured by monitoring stations in all communities exposed to the increases. Acknowledging the problem of the limited number of monitoring stations, the Smoke Dispersion Model is a good approximation of particulate matter levels. However, the Smoke Dispersion Model relies on weather data and fire specific information as input which is also limited due to lack of weather stations.

Using the weather and fire specific variables from the Chisholm Fire, the Smoke Dispersion Model predicts particulate matter levels in Edmonton to reach $35\mu g/m^3$. Actual measurements, from a central Edmonton weather station, on May 24th reached a daily average of $55\mu g/m^3$. Therefore, the Smoke Dispersion Model under predicts the true concentration levels. If further research consistently shows that the dispersion model under predicts concentrations, then a correction factor could be developed to account for the conservative estimation.

One possible explanation for the difference between the predicted concentration level and the reported value is due to the baseline value used in the dispersion model. The baseline value was assumed to be $12\mu g/m^3$ which was the average from the previous four months. However, it is possible that the previous day's level of particulate matter was high contributing to higher reported levels from monitoring stations on the day of the analysis. The Smoke Dispersion Model does not take into account the previous day's particulate matter concentration levels. In the case of the Chisholm Fire the reported $PM_{2.5}$ level for May 23rd, 2001, the day before this analysis was $18\mu g/m^3$ which is six units above the baseline used in the dispersion model.

Another explanation for the difference in particulate matter concentrations between the predictions of the Smoke Dispersion Model and the monitoring station's reported value is the change in weather patterns as smoke travels. It is probable that weather conditions from the start of the fire to the end of the plume vary, thus affecting dispersion at different rates throughout the area of the plume. However, dispersion calculations are estimated at the origin of the fire and not at specific distance intervals. Continuous monitoring of weather variables downwind from the fire is not possible because the number of weather stations, especially in northern Alberta, is limited. Therefore, errors occur when estimating weather variables. In the case of Chisholm, three weather stations exist to the north, east and west of Chisholm ranging from 70 to 150 kilometers away from the fire. An additional factor to consider is that it is assumed that the wind blows in one direction and at a constant speed. Realistically, the plume's direction will probably change as wind direction changes. A dynamic model is needed to capture this effect. The extent of the error from weather variables is unknown and the impact, overestimation or underestimation of concentration levels, is also not known. However, it is likely that part of the reason for the differences between reported concentration levels and the dispersion model's estimates lies in the values used for the weather variables.

In terms of the AQVM 3.0, the largest areas of concern are the concentration-response relationships for mortality and the monetary value of increased mortality risk. The value of increased mortality risk used determines the bulk of the health valuation estimates and therefore errors will be magnified compared to the other health estimates. Research on how individuals value mortality risk continues. However, there is a general consensus on the approaches used and their use in such studies. Also, in terms of the health impacts, there is some discussion in the literature exploring the analysis of threshold levels and the shape of the concentration-response functions. Currently, because the concentration-response functions used are linear there is no statistical difference between one individual exposed to an increase of $50\mu g/m^3$ compared to two individuals exposed to an increase of $25\mu g/m^3$. In terms of the Smoke Dispersion Model, the assumption of uniform mixing is made which allows all individuals to be exposed to the same concentration levels. However, it is possible that individuals near the center of a plume may be exposed to higher concentration levels compared individuals near the parameter. Additionally, in defining the plume's dimensions, health estimates are only calculated for individuals exposed to a level of 30$\mu g/m^3$ of $PM_{2.5}$ or higher. It is probable that there are health impacts from lower levels of $PM_{2.5}$ concentrations.

To verify and validate this case study, it would be desirable to examine actual health records for the dates of these smoke events, such an examination is challenging to conduct. Too few hospitals exist to correlate small changes in $PM_{2.5}$ levels to changes in hospital admissions. The scale of this case study is at the census subdivision level, hospitals would need to be located in each subdivision and individuals only go to hospitals in their area of residence to use actual hospital data.

6.7 CONCLUSIONS FROM THE CHISHOLM FIRE CASE STUDY

This case study describes the health impacts related to the 2001 Chisholm Fire. Weather and fire specific variables during the actual fire event were used to predict $PM_{2.5}$ concentration levels downwind from the fire. Impacts on the populations exposed were then estimated using the AQVM 3.0. The analysis focuses on one day during the fire in which wind direction was blowing towards large population centers. The results suggest a decrease in health value related to increased $PM_{2.5}$ concentration levels between $9 and $12 million, with the largest component of the total due to the increased risk of premature mortality. The values are notable especially when compared to other losses from the fire that are often highlighted by media, such as fire fighting costs ($10 million over seven days of the Chisholm Fire). Uncertainties from this case study are related to the AQVM 3.0 and the epidemiological and economic estimates contained within the model. Other limitations arise from insufficient weather and monitoring station data. While the analysis is for one day during the Chisholm Fire, it would be interesting to total the value in health lost over a longer time period such as a fire season.

7.0 RESEARCH CONCLUSIONS

This research provides an estimate of the importance of human health values in forest management decisions. Compared to other values at risk, health values are a significant proportion. A simple smoke dispersion model is used to estimate $PM_{2.5}$ concentration levels at various distances from a fire. It is not recommended that the model be used to predict exact smoke dispersion patterns, however it is a good approximation given specific fire and weather conditions. Sensitivity analysis demonstrates the weather and fire factors that will contribute to dangerous levels of $PM_{2.5}$. Scenarios with high emission rates and low dispersion characteristics predict concentrations to be above the Canada Wide Standard for distances up to 165 kilometers.

The AQVM 3.0 model estimates the health impacts of the increase in $PM_{2.5}$ levels on census subdivisions in Alberta. An important driver in the total health impacts is the number of individuals exposed. In terms of human health, areas with high populations are more at risk compared to lower populations and suppression efforts should take this into account especially when managers are faced with limited resources. Locations with a high rate of asthmatics or areas with elderly and young individuals should be identified as areas needing increased fire suppression efforts and public warning systems.

The health impact values derived are a conservative estimate because they only focus on the effects from $PM_{2.5}$ and ignore damages from other forest fire emissions. Additionally, the values presented are a mid-range of concentration-response functions and health valuation estimates. It is possible that actual values are higher. While direct medical costs are often a focus of media attention, the largest component of health values at risk come from the decrease in value to individuals because of the increase in premature mortality risk. It is also important to note that the health estimates examined in the case study are from one day during an intense fire. These are not values of a full fire season.

7.1. Fire Management Policy

The methods and use of this research can be implemented into fire management decisions. Fires occurring in multiple areas of the province raise concern because of limited resources available to suppress fires. Currently, fires occurring near large population centers are already given a high priority in terms of suppression. However, this research highlights the problems from smoke traveling to high population centers which are not just in close vicinity to the fire. Forest fire managers must continuously evaluate risks associated with each fire and monitor these risks relative to other fires in the province. The magnitude of the health impacts will be dependent on wind direction and the probability of the smoke blowing into large population centers. As weather patterns change, the predicted health values lost from a fire will also change. Therefore, ranking the potential losses from a fire must be a continuous process and ultimately the decision to suppress a fire based on health impacts may be dependent on the ability to predict wind direction.

Not only is this research useful for initial attack and fire control management, managers can also use this information in a benefit-cost analysis of fire suppression activities. After a fire season health values saved can be compared to the costs of fighting fires and suppression policies can be adjusted accordingly. An evaluation of fire suppression efforts should take into account the reduction in the number of days individuals are exposed as well as concentration levels during the exposure. With linear concentration-response functions there is no statistical difference between reducing exposure levels in half or by reducing the number of days exposed by half. Therefore, suppression activities can either focus on a successful initial attack which reduces the number of days exposed, or efforts can focus on lowering the emissions released from a fire.

7.2 Future Research

A limitation of this research is that the Smoke Dispersion Model is probably not accurate enough to predict the exact direction and concentration level of the flow of particulate matter over the full time period of a fire. The model is not dynamic in that it models dispersion in one direction and assumes weather variables are constant throughout the distance of the plume. Current computer-assisted models also have limitations in terms of ease of use and accuracy over long distances. Optimally, a dispersion model would be developed that is accurate enough to give early warning to individuals regarding increases in particulate matter levels. The focus of such a model must be its ease of use especially in terms of the amount of data input needed. A model automatically linked to weather stations without the need for manual data input is ideal. Additionally fires often occur in the same area, therefore a model is needed that can account for the contribution of numerous fires to the level of air quality.

While the accuracy in the level of exposure on a specific population is somewhat uncertain, there are also uncertainties in the concentration-response functions that describe the health effects associated with a given increase in particulate matter levels. Future research should focus on the issue of a threshold level in terms of a level in which particulate matter does not affect human health. A gap in the literature also exits in identifying the affect of cumulative impacts. Linear concentration-response functions lead to effects being additive which may not be the case.

This research focuses on the acute health effects from one fire event, the cumulative affects over a population exposed to many fires over a number of years probably produces chronic effects. Additionally with respect to concentration-response functions, it may be that the slope of the relationship between particulate matter and health effects is not linear. It is possible that after a critical level health effects increase exponentially with increases in particulate matter levels. These relationships may be dependent on the current health status of the individual exposed. Increased knowledge in the area of concentration-response functions, especially with respect to forest fires, will aid policy makers in their attempt to reduce the overall health impacts from particulate matter increases.

Increased research in the field of health valuation is also needed. Concerns arise from the instruments used to determine a value of a statistical life. While a combination of instruments are used by Environment Canada, each method has its own set of biases and limitations. There is also an emerging issue with respect to the increasing values associated with the value of a statistical life. As individuals real income and health status increase it is possible that the values placed on a statistical life will also increase. However, little research has been completed in this area.

The valuation of non-fatal health effects also needs improvement. Cost-of-illness studies only include direct medical costs and an estimate of lost wages while ignoring the value of pain and suffering associated with the morbidity. Methods used in healthcare intervention analysis include a summary measure of the values associated with a change in quality of life. However the practice has been to focus on numeric values and not a monetary measure that can be used in cost-benefit analysis. Economists will play an important role in the future acceptance of the methods used in health valuation. While the instruments used for health valuation are generally accepted among economists, non-economists have difficulty accepting their use in policy contexts.

With increased knowledge in the areas of smoke dispersion, concentration-response functions and health valuation, the ultimate use of the research will be in examining the return on investment from fire suppression efforts. It is likely that suppressing fires will reduce heath values lost. However, there is some concern that increasing suppression efforts (possibly to reduce health impacts) will cause larger, more intense fires in the future. The consequence of an increase in fire frequency and larger fires may increase exposure levels to amounts that have not been examined by the current research. Assessing the economic response to these dynamic affects is important.

An additional use of the methods presented in this research is to provide a starting point for the development of a values at risk map that will help guide fire management in spatial resource allocation decisions. There are likely population centers in Alberta that are more at risk from the health impacts associated with fires compared to other areas. Incorporating these values with other values at risk from fire will help fire managers allocate resources so that the impacts can be minimized. With the trend of increasing fire occurrence and limited suppression budgets, managers will continuously be forced to justify their expenditures. An accurate measure of health impacts will be essential.

Reference List

Abbey, D. E., Burchette, R. J., Knutsen, S. F., & McDonnell, E. F. Lebowitz, M. D. Enright P. L. (1998). Long-term particulate and other air pollutants and lung function in nonsmokers. American Journal of Critical Care Medicine, 158, 289-298.

Alberta Economic Development (2004). Alberta's Thriving Forest Industry [Web Page]. URL http://www.alberta-canada.com/forest/index.cfm [2004].

American Lung Association (2001). Selected Key Studies on Particulate Matter and Health: 1997-2001. URL http://www.kintera.org/atf/cf/%7B7A8D42C2-FCCA-4604-8ADE-7F5D5E762256%7D/PMSTUDIES.PDF [2004].

Amiro, B. D., Todd, J. B., Wotton, K. A., Logan, M. D., Flanigan, B. J., Stocks, J. A., Mason, D. L., Martell, D. L., & Hirsch, K. G. (2001). Direct carbon emissions from Canadian forest fires, 1959-1999. Canadian Journal of Forest Research, 31, 512-525.

Battye, W., & Battye, R. (2002). Development of Emissions Inventory Methods for Wildland Fire: Final Report. (EPA No. 68-D-98-046).

Bessie W.C., & Johnson E.A. (1995). The relative importance of fuels and weather on fire behaviour in subalpine forests. Ecology, 76(3), 747-762.

Breyfogle S. and Ferguson, A.S. (1996). User assessment of smoke-dispersion models for wildland biomass burning. United States Department of Agriculture Technical Report. PNW-GTR-379.

Brown, D. F. and Dunn, W.E. Lazaro, M.A. Policastro, A.J. (1999). The FirePlume model: Tool for eventual application to prescribed burns and wildland fires. The Joint Fire Science Conference and Workshop.

Canadian Forest Service (Eds.) (2004). The State of Canada's Forests. Ottawa, ON: Forestry Canada.

CASA (Clean Air Strategic Alliance) (2004). Data Warehouse. [Web Page]. URL http://www.casadata.org/ [2004].

CCME (Canadian Council of Ministers of the Environment) (2000). Particulate Matter and Ground-Level Ozone-Standard/ Status [Web Page]. URL http://www.ccme.ca/assets/pdf/pmozone_standard_e.pdf [2004].

Cardinal, M. (2004). Sustainable Resources Business Plan 2004-2007. Alberta Sustainable Development.

Chestnut, L. G., Mills, D., Rowe, R. D., De Civita, P., Stieb, D., Environment Canada, Health Canada, & Stratus Consulting (1999). Air quality valuation model version 3.0 (AQVM 3.0) report 2: methodology : final report. Boulder, Colo: Stratus Consulting.

Chisholm Fire Review Committee. (2001). Chisholm Fire Review Committee Final Report . S.l: The Committee.

Cumming S.G. (2001). Forest type and wildfire in the Alberta boreal mixedwood: What do fires burn. Ecological Applications, 11(1), 97-110.

Daniels, M., Dominici, F., Samet, J., & Zeger, S. (2000). Estimating particulate matter-mortality dose-response curves and threshold levels: An analysis of daily time-series for the 20 largest US cities. American Journal of Epidemiology, 152(5), 397-406.

Deck, L. B., Post, E. S., Smith, E., Wiener, M., Cunningham, K., & Richmond, H. (2001). Estimates of the health risk reductions associated with attainment of alternative particulate matter standards in two U.S. cities. Risk Analysis, 21(5), 821-36.

Delfino, R.J. Murphy-Moulton, A.M., Burnett, R.T. Brook, J.R. and Becklake, M.R. (1997) Effects of Air Pollution on Emergency Room Visits for Respiratory Illness in Montreal, Quebec. American Journal of Respiratory and Critical Care Medicine,155, 568-576.

Dobson, S., Thompson, J., Alberta, & Alberta Environmental Protection (1996). Parks and protected areas their contribution to the Alberta economy : a discussion paper (Pub. No. : T/375: Publication (Alberta. Alberta Environmental Protection) No. no. T/375). Edmonton: Alberta Environmental Protection, Strategic and Regional Services Division.

Dominici, F., Daniels, S., Zeger, S. L., & Samet, J. M. (2002). Air pollution and mortality: Estimating regional and national dose-response relationships. American Statistical Association, 97(457), 100-111.

Finney, M.A., Andrews, P.L. (1999). FARSITE-A program for fire growth simulation. Fire Management Notes. 59(2), 13-15.

Freeman, M.A. (2003). The Measurement of Environmental and Resource Values. Resource for the Future, Washington D.C.

Gauderman, W. J., Gilliland, G. F., Vora, H., Avol, E., Stram, D., McConnell, R., Thomas, D., Lurmann, F., Margolis, H. G., Rappaport, E. B., Berhane, K., & Peters, J. M. (2002). Association between air pollution and lung function growth in southern California children: results from a second cohort. American Journal of Respiratory and Critical Care Medicine, 166 (1), 76-84.

Goldberg, M. S., Bailar, J. C. 3rd, Burnett, R. T., Brook, J. R., Tamblyn, R., Bonvalot, Y., Ernst, P., Flegel, K. M., Singh, R. K., & Valois, M. F. (2000). Identifying subgroups of the general population that may be susceptible to short-term increases in particulate air pollution: a time-series study in Montreal, Quebec. Research Report Health Effects Institute, (97), 7-113; discussion 115-120.

Government of Alberta (2004). Climate and Geography [Web Page]. URL http://www2.gov.ab.ca/home/Index.cfm?Page=8 [2004].

Green, L. C., Crouch, E. A., Ames, M. R., & Lash, T. L. (2002). What's wrong with the National Ambient Air Quality Standard (NAAQS) for fine particulate matter? Regulatory Toxicology and Pharmacology, 35(3), 327-337.

Greene, W. H. (2000). Econometric Analysis (4th ed.). Upper Saddle River, N.J: Prentice Hall.

Heuss, J. (1999). The Health Effects of Particulate Matter: An Overview. The Annapolis Center for Science Based Public Policy. White Paper. Available on line http://www.annctr.org/Reports/particulatematter.pdf [2004].

Johnson, E. A., Miyanishi, K. & Bridge, S. R. J. (2001). Wildfire regime in the boreal forest and the idea of suppression and fuel buildup. Conservation Biology **15** (6), 1554-1557

Koop, G., & Tole L. (2004). Measuring the health effects of air pollution: to what extent can we really say that people are dying from bad air? Journal of Environmental Economics and Management, 47, 30-54.

Leksell, I., & Rabl, A. (2001). Air pollution and mortality: quantification and valuation of years of life lost. Risk Analysis, 21(5), 843-57.

Levy, D., Lumley, T., Sheppard, L., Kaufman, J., & Checkoway, H. (2001). Referent selection in case-crossover analyses of acute health effects of air pollution. Epidemiology, 12(2), 186-192.

Levy, D., Sheppard, L., Checkoway, H., Kaufman, J., Lumley, T., Koenig, J., & Siscovick, D. (2001). A case-crossover analysis of particulate matter air pollution and out- of-hospital primary cardiac arrest. Epidemiology, 12(2), 193-199.

Levy, J. I. (2003). Issues and uncertainties in estimating the health benefits of air pollution control. Journal of Toxicology and Environmental Health Part A, 16, 1865-1872.

Li, C., Flannigan, M. D., & Corns I.G.W. (2000). Influence of potential climate change on forest landscape dynamics of west-central Alberta. Canadian Journal of

Forest Research, 30, 1905-1912.

Lippmann, M., Ito, K., Nadas, A., & Burnett, R. T. (2000). Association of particulate matter components with daily mortality and morbidity in urban populations. Research Report Health Effects Institute, (95), 5-72, discussion 73-82.

Mckenzie, L.M. Hao, M.W. Richards, G..N. Ward, D.E. (1995). Measurement and modeling of air toxins from smoldering combustion of biomass. Environmental Science and Technology. 29, 2047-2054.

Nash, T. (1999). Alberta Forest Protection Advisory Committee, & KPMG (Firm). (1999-2000). Alberta Fire Review '98 Final Report. Edmonton: KPMG.

NRCan, Natural Resources Canada (2004). URL http://www.NRCan.gc.ca/cfs-scf/national/what-quoi/po_e.html [2004].

Ostro, B., & Chestnut, L. (1998). Assessing the health benefits of reducing particulate matter air pollution in the United States. Environmental Research, 76(2), 94-106.

Pandey, M. D., & Nathwani, J. S. (2003). Canada Wide Standard for particulate matter and ozone: cost-benefit analysis using a Life Quality Index. Risk Analysis, 23(1), 55-67.

Peters A, Dockery D.W., & Muller J.E. (2001). Increased particulate air pollution and the triggering of myocardial infarction., **103,** 2810-2815.

Pope, A. C., Burnett, R. T., Thun, M. J., Calle, E. E., Krewski, D., Ito, K., & Thurston, G. (2002). Lung cancer, cardiopulmonary and long-term exposure to fine particulate air pollution. Journal of American Medical Association. 287(9), 1132-1141.

Pope, C. A. 3rd, Bates, D. V., & Raizenne, M. E. (1995). Health effects of particulate air pollution: time for reassessment? Environmental Health Perspective, 103(5), 472-480.

Rabl, A. (1999). Uncertainty about the components of PM10 and their toxicities and its effects on the estimation of health damage. Presented at Air Pollution 99, Palo Alto .

Reinhardt, E. D., Keane, R. E., Brown, J. K., & Intermountain Research Station (Ogden, Utah) (1997). First Order Fire Effects Model FOFEM 4.0, User's Guide (General technical report: General technical report INT No. 344). Ogden, UT: United States Dept. of Agriculture, Forest Service, Intermountain Research Station.

Royal Society of Canada (2001). Report of an Expert Panel to Review the Socio-Economic Models and Related Components Supporting the Development of Canada-Wide Standards for Particulate Matter and Ozone. Ottawa: Royal Society of Canada.

Samet, J. M. , Dominici, F., Zeger, S. L., Schwartz, J., & Dockery, D. W. (2000). The national morbidity, mortality, and air pollution study. Part I: Methods and methodological issues. Research Report Health Effects Institute, (94 Pt 1), 5-14;

Samet, J. M., Zeger, S. L., Dominici, F., Curriero, F., Coursac, I., Dockery, D. W., Schwartz, J., & Zanobetti, A. (2000). The National morbidity, mortality, and air pollution study. Part II: morbidity and mortality from air pollution in the United States. Research Report Health Effects Institute, 94(Pt 2), 5-70

Sampson, N. R., Atkinson, D. R., & Lewis, J. W. (2000). Mapping Wildfire Hazards and Risks. Journal of Sustainable Forestry, 11, 119-157.

Schneider R.R. (2002). Alternative Futures: Alberta's Boreal Forests at the Crossroads. Edmonton, AB: Federation of Alberta Naturalists & Alberta Centre for Boreal Research.

Schwartz, J. (2000). Assessing confounding, effect modification, and thresholds in the association between ambient particles and daily deaths. Environmental Health Perspectives, 108(6), 563-568.

Schwartz, J. (1996). Air Pollution and Hospital Admissions for Respiratory disease. Epidemiology, **7(1).**

Schwartz, J. Laden, F. Zanobetti, A. (2002). The concentration–response relation between PM2.5 and daily deaths. Environmental Health Perspectives, 110(10), 1025-1029.

Schwartz, J. (2000). Assessing confounding, effect modification and thresholds in the association between ambient particles and daily death. Environmental Health Perspectives, 108(6), 563-568.

Statistics Canada. Canadian Statistics [Web Page]. URL http://www.statcan.ca/english/Pgdb/popula.htm#pop [2004].

Steenland, K., & Savitz, D. A. (1997). Topics in Environmental Epidemiology. New York: Oxford University Press.

Stieb, D. M., De Civita, P., Johnson, F. R., Manary, M. P., Anis, A. H., Beveridge, R. C., & Judek, S. (2002). Economic evaluation of the benefits of reducing acute cardiorespiratory morbidity associated with air pollution. Environmental

Health, 1(1), 7.

Sullivan, J., Ishikawa, N., Sheppard, L., Siscovick, D., Checkoway, H., & Kaufman, J. (2003). Exposure to ambient fine particulate matter and primary cardiac arrest among persons with and without clinically recognized heart disease. American Journal of Epidemiology, 157(6), 501-9.

SRD, (Sustainable Resource Development). 2004a. Forest Act [Web Page]. URL http://www3.gov.ab.ca/srd/forests/fmd/directives/fa.html. [2004]

SRD, (Sustainable Resource Development) 2004b. Historical Wildfire Database [Web Page]. URL http://www3.gov.ab.ca/srd/wildfires/fpd/wi_hdhwd.cfm [2004, April].

SRD, (Sustainable Resource Development). 2004c. FireSmart Manual. [Web Page]. URL http://www3.gov.ab.ca/srd/wildfires/fpd/pdf/firesmart03.pdf [2004]

United States, Environmental Protection Agency, & Office of Air Quality Planning and Standards. (1995). Compilation of air pollutant emission factors (5th ed.). Research Triangle Park, NC: U.S. Environmental Protection Agency, Office of Air and Radiation, Office of Air Quality Planning and Standards.

USDA (United States Forest Service).2004a. Fire Management Tools Online. [Web Page] URL http://www.fire.org/. [2004].

USDA (United States Forest Service).2004b. Fire Management Tools Online. [Web Page] URL http://www.fs.fed.us/bluesky/. [2004].

Ward, D. E. 1999. Smoke from wildland fires. Health Guidelines for Vegetation Fire Events, Lima, Peru, 6-9 October 1998. WHO background papers.

APPENDIX

Table A1: List of fire images used to estimate plume width[a]

Fire #	Date	Time	Location	Plume Length (Km)	Wind Direction at Noon[b]
F1	06/25/2002	N/A	Northern Alberta	58	W
F2	06/10/2002	N/A	Northern Alberta	75	NE
F3	06/10/2002	N/A	Northern Alberta	22	SW
F4	06/10/2002	N/A	Northern Alberta	15	SW
F5	06/10/2002	N/A	Northern Alberta	26	SW
F6	07/10/2002	N/A	Northern Alberta	82	SW
F7	06/15/2002	AM	Northern Alberta	43	NE
F8	06/15/2002	AM	Northern Alberta	56	NE
F9	06/15/2002	AM	Northern Saskatchewan	24	NE
F10	06/15/2002	AM	Northern Alberta	28	NE
F11	06/15/2002	N/A	Northern Saskatchewan	40	N
F12	06/15/2002	N/A	Northern Saskatchewan	85	N
F13	08/01/2003	PM	Rocky Mountain Region (USA)	9	SW
F14	08/01/2003	PM	Rocky Mountain Region (Canada)	18	SW
F15	08/01/2003	PM	Rocky Mountain Region (USA)	35	SW
F16	08/01/2003	PM	Rocky Mountain Region (Canada)	15	SW
F17	07/28/2003	N/A	Rocky Mountain Region (Canada)	28	NW
F18	05/30/2002	N/A	Northern Saskatchewan	38	NW
F19	05/30/2002	N/A	Northern Saskatchewan	28	NW
F20	07/10/2002	N/A	Wood Buffalo National Park	32	W
F21	07/10/2002	N/A	Wood Buffalo National Park	62	W

a: Fire images obtained from Alberta Sustainable Development

b: Wind direction obtained from the Canadian Forest Service

Appendix 2: Distance and width data from 21 forest fires in western Canada and north western United States

F1		F2		F3		F4		F5		F6	
Distance from fire (Km)	**Actual Width (km)**	**Distance from fire (Km)**	**Actual Width (km)**	**Distance from fire (Km)**	**Actual Width (km)**	**Distance from fire (Km)**	**Actual Width (km)**	**Distance from fire (Km)**	**Actual Width (km)**	**Distance from fire (Km)**	**Actual Width (km)**
0	2.878	0	19.54875	0	15.0375	0	6.766875	0	10.36	0	13.068
7.195	7.195	7.51875	30.82688	7.51875	18.045	7.51875	16.54125	1.85	11.47	8.1675	19.602
14.39	12.2315	15.0375	37.59375	15.0375	21.0525	15.0375	27.0675	3.7	12.025	16.335	31.0365
21.585	15.829	22.55625	40.60125	22.55625	22.55625			5.55	12.395	24.5025	35.937
28.78	24.463	30.075	49.62375					7.4	12.95	32.67	36.75375
35.975	29.4995	37.59375	54.135					9.25	14.8	40.8375	40.8375
43.17	30.9385	45.1125	42.105					11.1	15.54	49.005	44.1045
50.365	33.097	52.63125	18.045					12.95	16.65	57.1725	52.272
57.56	38.853	60.15	21.0525					14.8	15.355	65.34	55.539
		67.66875	22.55625					16.65	15.91	73.5075	52.272
								18.5	16.65	81.675	55.539
								20.35	17.39		
								22.2	18.315		
								24.05	16.28		
								25.9	17.02		

F7		F8		F9		F10		F11		F12	
Distance from fire (Km)	Actual Width (km)	Distance from fire (Km)	Actual Width (km)	Distance from fire (Km)	Actual Width (km)	Distance from fire (Km)	Actual Width (km)	Distance from fire (Km)	Actual Width (km)	Distance from fire (Km)	Actual Width (km)
0	2.481	0	3.10125	0	6.82275	0	4.272	0	4.528	0	7.924
6.2025	8.06325	6.2025	4.962	6.2025	8.6835	7.12	5.696	5.66	9.056	5.66	10.754
12.405	8.06325	12.405	6.2025	12.405	12.405	14.24	9.968	11.32	8.49	11.32	10.754
18.6075	9.30375	18.6075	4.962	18.6075	12.405	21.36	12.816	16.98	13.584	16.98	13.584
24.81	9.924	24.81	8.06325	24.81	14.886	28.48	14.952	22.64	10.188	22.64	12.452
31.0125	8.6835	31.0125	10.54425					28.3	11.32	28.3	14.716
37.215	11.1645	37.215	6.2025					33.96	11.32	33.96	15.282
43.4175	4.962	43.4175	10.54425					39.62	9.056	39.62	16.98
		49.62	23.5695							45.28	15.848
		55.8225	12.405							50.94	11.886
										56.6	9.056
										62.26	14.716
										67.92	23.772
										73.58	29.432
										79.24	30.564
										84.9	31.696

F13		F14		F15		F16		F17		F18	
Distance from fire (Km)	**Actual Width (km)**	**Distance from fire (Km)**	**Actual Width (km)**	**Distance from fire (Km)**	**Actual Width (km)**	**Distance from fire (Km)**	**Actual Width (km)**	**Distance from fire (Km)**	**Actual Width (km)**	**Distance from fire (Km)**	**Actual Width (km)**
0	5.4936	0	19.2276	0	15.543	0	19.782	0	18.4	0	5.88
2.289	9.156	2.289	20.601	7.065	15.543	7.065	24.021	2.3	18.4	2.94	5.88
5.48266	9.6138	4.578	23.3478	14.13	22.608	14.13	28.26	4.6	19.32	5.88	5.88
5.74795	10.5294	6.867	22.89	21.195	25.434	21.195	35.325	6.9	23	8.82	7.056
5.92481	8.2404	9.156	23.8056	28.26	21.195			9.2	24.38	11.76	7.644
6.1901	11.2161	11.445	27.0102	35.325	19.782			11.5	26.22	14.7	6.468
7.0744	11.9028	13.734	29.757					13.8	28.52	17.64	7.938
7.42812	10.9872	16.023	32.046					16.1	29.9	20.58	11.172
7.9587	14.1918	18.312						18.4	30.36	23.52	12.348
7.33969	15.5652							20.7	32.2	26.46	12.348
7.60498	18.7698							23	30.82	29.4	12.936
7.9587	17.8542							25.3	28.98	32.34	12.348
8.31242	17.8542									35.28	13.524
8.75457	16.023									38.22	13.818

F19		F20		F21	
Distance from fire (Km)	Actual Width (km)	Distance from fire (Km)	Actual Width (km)	Distance from fire (Km)	Actual Width (km)
0	4.6	0	1.132	0	5.66
2.3	5.29	5.66	2.264	5.66	6.792
4.6	8.74	11.32	2.264	11.32	12.452
6.9	9.2	16.98	5.094	16.98	11.32
9.2	9.2	22.64	5.66	22.64	12.452
11.5	14.26			28.3	13.584
13.8	14.72			33.96	16.98
16.1	14.72			39.62	19.244
18.4	19.78			45.28	22.64
20.7	19.78			50.94	23.772
23	18.86			56.6	24.904
25.3	18.86			62.26	27.168
27.6	18.4				