Economic analysis of health effects from forest fires

R. Rittmaster, W.L. Adamowicz, B. Amiro, and R.T. Pelletier

Abstract: Epidemiological studies have shown that high levels of fine particulate matter (PM) are correlated with adverse human health effects. Approximately one-third of PM emissions in Canada originate from forest fires. However, air quality concerns are not typically included in resource allocation decisions in fire management. In this paper we examine the economic magnitude of these health concerns and compare them to other costs of forest fires using the 2001 fire in Chisholm, Alberta, as a case study. We construct an empirical air dispersion model to estimate the concentration of PM arising from the fire. Benefit transfer methods were used to determine the health impacts associated with elevated PM from the fire and to value these impacts. The economic impacts appear to be substantial, second only to timber losses. The approaches used in this case study can be extended to construct a map that identifies the values at risk from health effects. The use of monetary values of these impacts helps in comparison and aggregation of the values at risk.

Résumé : Des études épidémiologiques ont montré que des niveaux élevés de fines particules étaient corrélés avec des effets néfastes sur la santé humaine. Environ le tiers des particules émises au Canada proviennent des feux de forêt. Cependant, les conséquences sur la qualité de l'air ne sont généralement pas considérées dans les décisions au sujet de l'allocation des ressources en gestion du feu. Les auteurs examinent dans cet article l'amplitude économique de telles considérations pour la santé et nous les comparons aux autres coûts associés aux feux de forêt en utilisant comme étude de cas le feu de 2001 à Chisholm, en Alberta. Ils construisent un modèle empirique de dispersion aérienne pour estimer les concentrations de particules dégagées par le feu. Ils utilisent la méthode des transferts de bénéfices pour déterminer les incidences sanitaires associées aux taux élevés de particules provenant du feu et pour les évaluer monétairement. Les impacts économiques apparaissent substantiels, dépassés seulement par la valeur des pertes de bois. Les procédures développées dans cette étude de cas peuvent être étendues afin d'établir une carte permettant d'identifier les valeurs à risque à cause des effets sur la santé. L'utilisation de valeurs monétaires caractérisant ces impacts facilite la comparaison et l'agrégation des valeurs à risque.

[Traduit par la Rédaction]

Introduction

Several severe forest fire years and some highly publicized forest fire events near urban centers have prompted much public and scientific debate on forest fire management. Efforts to manage the risks of forest fires often focus on risks to human life, property, timber, wildlife habitat, and infrastructure (Sustainable Resource Development 2003). While human life is considered a high priority, most of the focus surrounds evacuations from areas close to the fire and direct risks to loss of life from the fire itself. There has been relatively little analysis of the health effects of smoke associated with wildfire. While it is recognized that smoke reduces air quality, there have been few efforts to calculate the magnitude of these effects relative to those of other effects arising from wildfire.

In this paper, we present a case study that assesses the economic impact of air quality changes arising from forest fires. We examine the human health effects of a change in air quality arising from the May 2001 fire in Chisholm, Alberta, and calculate the economic impacts of these health effects. The 7-day Chisholm fire burned 116 000 ha, causing average fine particulate matter ($PM_{2.5}$) 160 km away in Edmonton to reach high levels (see, e.g., Cheng et al. 1998). We use the "damage function approach" to develop our estimates. The damage function model has three steps: (1) estimate the change in pollution concentration, (2) estimate the human health effects arising from the change in concentration, and (3) estimate the economic value of health effects. In applying this approach we take the following steps: first, we construct a statistical dispersion model based on satellite images of forest fire plumes to identify the concentration of PM associated with the Chisholm fire. We also compare these values with actual values from monitoring stations on the dates of the fire. Next, we employ the health and economic valuation relationships described in a model of air

Received 10 March 2005. Accepted 28 November 2005. Published on the NRC Research Press Web site at http://cjfr.nrc.ca on 29 March 2006.

R. Rittmaster and W.L. Adamowicz.[1] Department of Rural Economy, University of Alberta, 515 General Services Building, Edmonton, AB T6G 2H1, Canada.
B. Amiro. Department of Soil Science, University of Manitoba, 364 Ellis Building, Winnipeg, MB R3T 2N2, Canada.
R.T. Pelletier. Department of Renewable Resources, University of Alberta, 751 General Services Building, Edmonton, AB T6G 2H1, Canada.

[1]Corresponding author (e-mail: vic.adamowicz@ualberta.ca).

 doi:10.1139/X05-293

quality valuation developed by Environment Canada and Health Canada (Air Quality Valuation Model (AQVM)) to quantify the health and economic effects (Chestnut et al. 1999). The approach we take is to model the fire as one would model an air pollution event. Finally, we compare the magnitude of the health effects relative to timber losses and infrastructure losses. Our approach does not answer the question of whether efforts on fire suppression are 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 management.

Air quality

Particulate matter (PM) is a recognized pollutant, and recent studies have shown that high levels of $PM_{2.5}$ (particles smaller than 2.5 μm) are correlated with substantial adverse health effects. The Canadian Council of Ministers of the Environment set Canada-wide guidelines in 2000 for PM <2.5 μm in aerodynamic diameter ($PM_{2.5}$) to 30 $\mu g/m^3$ over a 24 h period. While these guidelines are focused on industrial emissions, approximately one-third of $PM_{2.5}$ emissions in Canada come from forest fires (Environment Canada 1999). There is considerable recognition of the effects of smoke from prescribed burning on air quality, as conditions must be evaluated relative to the potential to violate air quality guidelines. However, there is relatively little discussion of the impact of air quality changes arising from wildfire and the potential for these effects to influence fire management and priority setting (e.g., Chisholm Fire Review Committee 2001). A notable exception is Fowler (2003), who summarizes the literature on air quality and human health with applications to forest fire smoke. In this paper, we examine $PM_{2.5}$ as the sole pollutant from forest fire, recognizing that other materials are also generated by fires and may create human health hazards.

Health effects

Recent epidemiological studies have shown that high levels of $PM_{2.5}$ are correlated with substantial adverse health effects, including respiratory problems, pneumonia, chronic obstructive pulmonary disease, heart disease, stroke, and premature mortality (US EPA 2004). PM contains many different chemicals that contribute to adverse health impacts. Substances released from biomass burning include combustion materials, sulfates, nitrates, mercury, and dust (Rabl 1999). The amount of PM released depends on the type of fuels burned and the intensity of the fire. While there has been some speculation that the effects of PM from forest fires may be different than the effects of PM from industrial sources, there does not appear to be substantial evidence to support this speculation (Adamowicz et al. 2004).

The health effects of PM have been assessed using two major classes of data. The first type examines daily time series data and correlates the PM levels with health effects (mortality, hospital admissions, etc.). The second approach tracks cohorts of individuals and assesses their response to changes in air quality (Adamowicz et al. 2004). These latter studies tend to produce evidence of more significant health impacts because of the potential for chronic exposure and because the heterogeneity of individuals is accounted for. A recent cohort study provided evidence of increased mortality risks and increased cancer risks as a result of higher PM levels (Pope et al. 2002).

Samet et al. (2000) estimated a 0.5% increase in daily mortality per 10 $\mu g/m^3$ increase in PM_{10} (PM smaller than 10 μm). It has been suggested that $PM_{2.5}$ is more harmful than PM_{10} because the diameter is smaller and more of it is inhaled and absorbed through the lungs. In terms of morbidity, Lippman et al. (2000) found positive effects of $PM_{2.5}$ on both chronic obstructive pulmonary disease and ischemic heart disease. Similarly, Samet et al. (2000) concluded that for every 10 $\mu g/m^3$ increase in PM_{10} there was a 2% increase in the risk of pneumonia and chronic obstructive pulmonary disease hospital admissions. Schwartz et al. (1994) also found positive correlations between PM_{10} and respiratory disease as well as a positive correlation between PM_{10} and heart disease in the elderly.

Economic effects

Concentration response models provide information on changes in health risks associated with changes in pollutant concentrations. Economic methods are used to translate these changes in risks into economic (monetary) values. These economic values are calculated in various ways. The value of health effects include the costs of illness as reflected in hospital costs and lost wages as well as the individual's willingness to pay to avoid the change in risk, over and above the costs of illness. The latter are often referred to as the "pain and suffering" effects associated with the illness. For an overview of the calculation of these measures and an outline of their use in policy analysis see Adamowicz et al. (2004).

A particularly important value is the value of mortality risk reductions. This value typically accounts for a substantial percentage of the economic effects of air quality changes. Mortality risk reduction valuation studies identify the amount an individual would pay for a small reduction in the risk of death. The willingness to pay values can be derived from market data on wage risk trade-offs or from structured survey data eliciting individuals' responses to trade-off questions (Freeman 2003). When the willingness to pay value is added over a population and then divided by the total number of lives saved, a value of a risk reduction or 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 are willing to pay $6 million per statistical life. It is important to note that the purpose of a VSL is to estimate what a population would pay for a "small" risk reduction spread over the group. It does not attempt to estimate the intrinsic value of a life. VSL estimates in the literature vary considerably (see Adamowicz et al. 2004.) There is a great deal of literature behind the choice of a measure of VSL. The estimate we employ within the AQVM is an estimate based on a blend of studies of various types (hedonic wage studies and contingent valuation studies). For its central measure of mortality risk reduction value, AQVM uses a VSL that is adjusted slightly downward to attempt to take into account the variation in VSL for groups most likely to be affected by air quality changes. This results in a reduction of the general population VSL of CAN$5.2 million to CAN$4.1 million (both values in 1996 dollars). This modification of the VSL to account for different values among subgroups in the population (e.g., the elderly) is somewhat controversial. When the US EPA attempted to apply such differential valuation, there was a significant outcry from various nongovernmental organizations, forcing the EPA to reconsider their efforts (see, e.g., http://www.epa.gov/aging/listening/2003/tampa_tuft. htm and http://www.epa.gov/aging/listening/2003/balt_ctw. htm). We retain the value used in AQVM for our analysis and note that the sensitivity analysis will include values much larger than the general population value. This also produces a slightly more conservative estimate. Some of the economic values of nonmortality health effects are based on cost of illness information or a combination of cost-of-illness and willingness-to-pay approaches (Ostro and Chestnut 1998). In these cases, direct medical costs are costs from hospital admissions and emergency room visits.

AQVM

AQVM is a database of health concentration response functions and economic valuation estimates for a range of pollutants and economic impacts. It contains demographic information and projections for Canada at a census division level. AQVM has been used as a model for the estimation of the benefits of air quality improvements at a national level (see Adamowicz et al. 2004). It also provides a set of probability weights for various levels of estimates of health effects and economic impacts, thereby facilitating a form of sensitivity analysis. AQVM allows the user to modify the set of information used in constructing estimates by choosing which concentration response functions to use, what weights of economic values or health values to include, and whether to include thresholds below which no impact will be felt.

Concentration response functions in AQVM 3.0 are based on previous epidemiological studies. The designers of AQVM selected studies that controlled for confounding variables such as weather, seasonality, other pollutants, and location. In the case of $PM_{2.5}$, these functions estimate health effects resulting from a one unit increase in $PM_{2.5}$. The calculations multiply a risk value by the number of people affected.

Health endpoints available for $PM_{2.5}$ in AQVM 3.0 include mortality, cardiac hospital admissions, respiratory hospital admissions, emergency room visits, asthma symptom days, restricted activity days, acute respiratory symptom events, 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. Probability weights for each function are provided based on the confidence of the estimate.

There are over 800 studies describing the correlations between health outcomes and PM (American Lung Association 2001). Developers of AQVM 3.0 assessed the values from studies that were evaluated based on the quality of the research. Criteria for data quality included continuous monitoring of PM levels, minimal location bias, control of seasonal and weather patterns, model specification, and studies that tested a Canadian population. Table 1 summarizes the concentration response functions used in 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 designers of AQVM adjusted the concentration response functions in the model.

AQVM 3.0 provides estimates of impacts based on census divisions and the population characteristics of the census divisions. Included in the model are demographic characteristics of the population. An older population will be more at risk for some illnesses than will a younger 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 are combined with the health concentration response information to generate estimates of the impacts of air quality changes on a variety of health categories. Economic valuation estimates in AQVM include measures of mortality risk reduction values (VSL) and morbidity reduction values. The mean of the “central” estimate used by AQVM 3.0 for a VSL is $4.1 million, and this value is used in our analysis. AQVM also provides an estimate of “high” and “low” VSL, which are $2.4 million and $8.2 million, respectively. A summary of the central value estimates, as well as high and low estimates for morbidity reductions, is provided in Table 2.

AQVM generates a form of sensitivity analysis using Monte Carlo methods. For each endpoint (illness) and each valuation estimate related to these outcomes, low, central, and high estimates are provided (see Tables 1 and 2 for examples). Each of these estimates is assigned a probability weight (Tables 1 and 2). These weights form a discrete probability distribution. In the sensitivity analysis, AQVM chooses a concentration response function (based on the weights in the probability distribution) and then combines it with a selected valuation amount (again based on the weights) and uses these over all endpoints to generate an estimate of value. This process is repeated 5000 times to create an empirical distribution of overall outcome, based on the discrete empirical distributions of the endpoint-specific concentration response and valuation amounts. The sensitivity analyses reported below are based on this approach and report the expected value as well as the 10% and 90% levels on this empirical distribution.

Methods

We apply AQVM to the short-term (daily) impacts arising from forest fires, assuming that concentration response functions estimated from daily health effects and pollution concentration data are applicable. Furthermore, concentration response models in AQVM are linear, thereby facilitating scaling over ranges of air quality and time. Clearly, this is an assumption that has been discussed in critiques of AQVM (e.g., Adamowicz et al. 2004), but standard practice for policy is to continue to employ these functions in the absence of improved information.

Table 1. Concentration response functions used in AQVM 3.0 for increases in $PM_{2.5}$ concentrations.

Health effect	Source	Concentration response function	
		Level	Estimate
Annual mortality risk per 1 µg/m^3 change in annual average $PM_{2.5}$ concentration.	Pope et al. 1995; Schwartz et al. 1996	Low	0.87×10^{-5} (22%)
		Central	2.14×10^{-5} (67%)
		High	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.	Burnett et al. 1995	Low	4.13×10^{-8} (25%)
		Central	1.21×10^{-8} (50%)
		High	1.42×10^{-8} (25%)
Cardiac hospital admissions daily risk per 1 µg/m^3 change in daily average $PM_{2.5}$ concentrations.	Burnett et al. 1995	Low	1.00×10^{-8} (25%)
		Central	8.27×10^{-8} (50%)
		High	12.4×10^{-8} (25%)
Net emergency room visits daily risk factors per 1 µg/m^3 change in daily averages $PM_{2.5}$ concentrations.	Stieb et al. 1995	Low	4.62×10^{-8} (25%)
		Central	8.27×10^{-8} (50%)
		High	12.4×10^{-8} (25%)
Asthma symptom days daily risk factors given a 1 µg/m^3 change in daily average $PM_{2.5}$ concentration. For population with asthma (6%).	Whittemore and Korn 1980; Ostro et al. 1991	Low	1.6×10^{-4} (33%)
		Central	2.64×10^{-4} (34%)
		High	3.65×10^{-4} (33%)
Restricted activity days daily risk factors given a 1 µg/m^3 change in daily average $PM_{2.5}$ concentration. Non-asthmatic population (94%).	Ostro 1987; Ostro and Rothschild 1989	Low	1.31×10^{-4} (25%)
		Central	2.50×10^{-4} (50%)
		High	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.	Krupnick et al. 1990	Low	1.25×10^{-4} (25%)
		Central	2.79×10^{-4} (50%)
		High	4.14×10^{-4} (25%)
Child acute bronchitis annual risk factors given a 1 µg/m^3 change in annual average $PM_{2.5}$ concentration.	Dockery et al. 1996	Low	0.62×10^{-3} (25%)
		Central	1.65×10^{-3} (50%)
		High	2.69×10^{-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), p. D-2.

Table 2. Economic valuation of morbidity estimates used in AQVM 3.0.

Health effect	1996 CAN$ (millions)			Source	Study type[a]
	Low	Central	High		
Respiratory hospital admission	3 300	6 600	9 800	Canadian Institute for Health Information 1994	COI
Cardiac hospital admission	4 200	8 400	12 600	Canadian Institute for Health Information 1994	COI
Emergency rom 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, 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 morbidity estimates	33%	34%	33%		

Note: Table extracted from Chestnut et al. (1999), pp. 5–31.
[a]COI, cost-of-illness study; WTP, illingness-to-pay study.

The following modifications and adjustments were made to the concentration response components of AQVM to focus on the effects of smoke from wildfire. All estimates of chronic illness related to $PM_{2.5}$ were excluded from the model. 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. Included in the calculation of the impacts of short-term increases in PM are the following health outcomes: premature mortality, childhood bronchitis, cardiovascular disease, asthma, and other acute respiratory symptoms.

For this study, only acute impacts were modeled, as the increases in $PM_{2.5}$ were of short duration. This means that for the mortality risk values, only the time series studies, and not cohort studies, were included in the measurement of effects. Concentration response functions are multiplied by the PM 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 µg/m^3 annual increase in $PM_{2.5}$ is 0.87×10^{-5}, and if the population exposed to the increase is 1 million, then the number of mortality cases each year is expected to increase by six. Annual concentration

Fig. 1. Average daily $PM_{2.5}$ levels (µg/m^3) in Edmonton and Red Deer, Alberta, 1 April 2001 – 30 May 2001.

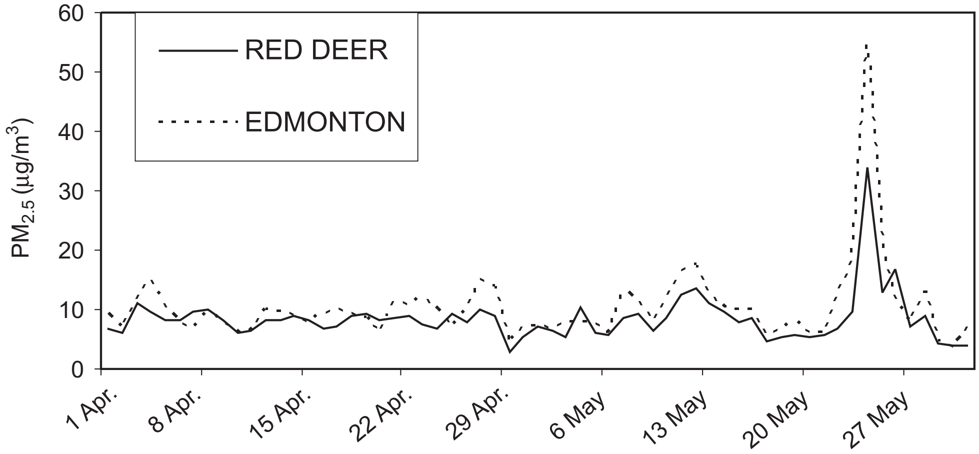
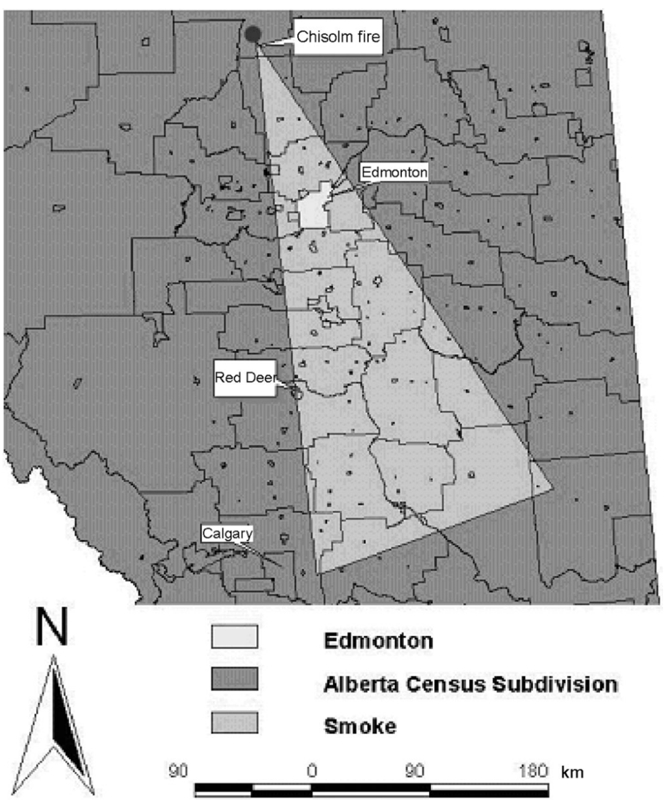


Fig. 2. Predicted smoke dispersion from the 24 May 2001 Chisholm fire, Alberta, Canada.



response functions were divided by 365 to calculate daily responses to changes in $PM_{2.5}$ levels, while daily concentration response functions were retained as specified in AQVM.

The Chisholm fire case study

The 2001 Chisholm fire in northern Alberta burned from 23 to 29 May 2001. In total, the fire burned 116 000 ha of forest land and burned structures in the town of Chisholm and surrounding settlements and infrastructure. The weather during the fire was characterized by a high temperature of 27 °C, low humidity, and winds gusting to 50 km/h (Chisolm Fire Entrapment Investigation 2001). These extreme weather conditions combined with a dry, flammable boreal forest contributed to the fire event. Values lost due to the fire include timber values, energy sector infrastructure, rail impacts, housing values, and impacts on wildlife (CFRC 2001). On May 28 2001, a state of emergency was declared for the Chisholm region.

Chisholm is 160 km north of Edmonton, a major metropolitan area of approximately 1 million people. The fire caused PM levels in Edmonton to rise from a daily average level of about 12 µg/m^3 to an hourly high of 261 µg/m^3. Significant increases in PM were also experienced in the city of Red Deer (125 km south of Edmonton) and surrounding areas. These smoke events were only experienced for a short period (approximately 2 days), as the wind changed direction and the smoke plume turned north. Nevertheless, the effects on air quality were dramatic (Figs. 1, 2).

To determine health impacts from the Chisholm fire we first assessed the contribution of the fire to PM concentrations. Health and economic impacts were then assessed using the relationships described in AQVM 3.0. These steps are explained in detail below.

PM concentration estimates

Changes in concentrations of PM arising from the fire were calculated in two ways. In the first approach the actual PM levels as reported at monitoring stations on the days of smoke impacts were used to provide a measure of concentration change. This approach, however, examines the PM levels on those days relative to an average PM level. It may be that other sources of particulates were coincidentally high on those days. The source of PM is not necessarily the Chisholm fire. Therefore, we also construct a statistical emissions transport function to estimate particulate concentrations in the regions surrounding the fire. This approach provides a concentration change solely attributable to the Chisholm fire.

Approach 1: monitoring stations

PM levels were obtained from the Red Deer and Edmonton central monitoring stations (obtained from Clean Air Strategic Alliance, CASA data archives: http://www.casadata.org). The hourly levels at these stations were used to calculate daily averages. The effect of the fire is illustrated in the graph of PM levels over time (Fig. 1) for April and May

Table 3. Mean values of health impacts related to increased $PM_{2.5}$ levels from the Chisholm Fire on 24 May 2001.

Health outcome	Mean 1996 $CAN	
	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 30 μg/m^3.
[b]Population exposed: 670 000. Exposure levels: Edmonton, 55 μg/m^3; Red Deer, 35 μg/m^3.

2001. The 7-day Chisholm fire caused average $PM_{2.5}$ levels 160 km away in Edmonton to reach 55 μg/m^3 on a daily basis and levels of 261 μg/m^3 on an hourly basis. These $PM_{2.5}$ levels were well above the Canadian standard of 30 μg/m^3 averaged over a 24 h period. To be conservative we used average daily PM levels for the 2 days that contained smoke effects for the Edmonton and Red Deer census divisions. The census division that included Calgary was also affected by the fire but not to the extent that Edmonton and Red Deer were. Because the effects on Calgary and other census divisions fall within what could have been near normal variation in PM levels, we excluded Calgary and all other census divisions from the analysis.

Approach 2: concentrations calculated from a smoke dispersion model

The second strategy for estimating particulate concentrations involved the development of a simplified smoke dispersion model. This was based on information gathered on the width of smoke plumes with distance from the fire through measurements of satellite imagery. This was complemented by weather information to develop an atmospheric dispersion estimate. The dispersion relationship was combined with estimates of the source term to determine atmospheric particulate concentration. The details of this method are presented in Appendix A.

Results

Results from monitoring station calculations

On the second day of the fire, the Edmonton central monitoring station reported a 24 h average of 55 μg/m^3, while in Red Deer the reported average was 35 μg/m^3. The health impacts from the increases were estimated using AQVM 3.0, with the results presented in Table 3. Aggregate impacts on health from a 1-day increase of PM totaled $12.1 million. The bulk of the estimate comes from premature mortality risks, with other contributions from restricted activity days and acute respiratory symptoms. A sensitivity analysis that varied the weights on the economic valuation measures and the weights on the concentration response relationships according to the procedure described previously provides a range for these impacts from $4.9 million (10th percentile) to $22.9 million (90th percentile) (see Rittmaster 2004 for details).

Results from the smoke dispersion model

The total population exposed to the smoke plume on 24 May 2001 was 1.1 million people. Edmonton is the largest city affected, containing approximately 1 million people in the metropolitan area. The predicted exposure level in Edmonton on 24 May 2001 is 35 μg/m^3 as a daily mean. Table 3 summarizes these impacts. As expected, the largest component of the total health impact was related to the increase in premature mortality risk. Morbidity values, for example, restricted activity days, are much lower in magnitude and form a smaller component of the total health value lost from the Chisholm fire. Sensitivity analysis that considers variation in the weights of the concentration response functions and the economic values applied as described previously suggests a range of total values of health impacts between $4 million (10th percentile) and $19.4 million (90th percentile).

A comparison of the health values with other fire impacts

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. Morbidity effects 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, we compare these impacts with other estimated costs of the Chisholm fire. Approximately 75 buildings were lost, including 21 homes. The economic value of these losses is difficult to assess. The value of lost infrastructure (bridges) was estimated to be $2 million. Loss of electrical power infrastructure was estimated at $1 million. Details on these monetary estimates are provided in the Chisholm Fire Review Committee's (2001) report. Detailed information on impacts can be found in the appendices of the review committee's report. Fire-fighting costs were reported to be $10 million over the 7-day fire, which is close to our estimates of the 1-day total value of the health impacts. Timber supply lost during the fire was estimated at $20 million, based on the reported loss of annual allowable cut of 50 000 m^3 (Chisholm Fire Review Committee 2001, section 4) and calculations assuming a 5% discount rate over 100 years (extending to approximately one rotation) and a net price of \$20/m^3. Note that in the public comments section of the Chisholm Fire Review Committee's (2001) report, some agencies reported annual allowable cut losses twice this size, but those values were not reported in the table in section 4 of the report. Additional impacts on other industries were also reported, but are smaller in magnitude. Health effects, while not as large as impacts on timber supply, are a significant component of the total impact of the fire. This will not be the case in all fires, but it is interesting to note that even in this case study, only 1 or 2 days of the fire generated smoke effects over the large city in the region. A very different result would have been obtained if the wind continued to blow smoke over the Edmonton region for the entire length of the fire or if the main population had been totally missed.

Discussion

It is required under the Government of Canada Regulatory Policy that all public expenditures be evaluated by a cost–benefit analysis. Future forest fire management decisions would benefit from detailed information regarding the values at risk from fire, including potential losses of timber values, recreation values, and the cost of forest fire smoke on human health. The importance of such knowledge is increasing with the likelihood that fire will increase in future climates (Flannigan et al. 2005). The calculations provided previously illustrate how such effects could be measured for the case of health impacts. The impacts are measured in monetary terms to provide a common metric for analysis of magnitudes of impacts.

Some of the limitations associated with this case study are related to AQVM 3.0 and the epidemiological and economic estimates contained within the model. As noted by Adamowicz et al. (2004), the largest areas of concern are the exposure–response relationships for mortality and the monetary values used for VSL. The VSL value used determines the bulk of the health valuation estimates. Research on VSLs and related measures continues; however, there is a general consensus on the approach used to calculate VSLs and their use in such studies.

There are also questions that arise regarding the epidemiological studies. While a large number of time series models have confirmed the effects of PM on health, a recent study by Koop and Tole (2004) raises questions about these estimates. There has been limited analysis of the presence of threshold effects. Current literature suggests there is little evidence of a minimum threshold level at which PM does not affect health. Recent research by Schwartz (2000) has suggested that the minimum dose may be around 2 μg/m^3. The shape of the concentration response function is also surrounded by uncertainty. It may be that the relationship is not linear, as assumed in AQVM 3.0. A maximum level at which adverse health effects taper or increase substantially has not been determined. An additional concern put forth by the Royal Society Panel (Adamowicz et al. 2004) is that the frequency and variation of PM levels may play an important role in determining the extent of the pollution-related health problems. The Royal Society Panel questions the extent to which peaks in concentration levels, similar to the ones experienced during the Chisholm fire, would be more harmful than just an increase in the annual average.

This case study examined two census divisions, Edmonton and Red Deer. It is possible that the smoke plume affected health in populations outside the study area. Additionally, the final estimate only evaluates the consequences of increased $PM_{2.5}$ levels while ignoring PM_{10} and other emissions. While $PM_{2.5}$ is more harmful, it is probable that PM_{10} contributes to changes in health outcomes as well. In addition, there is some evidence that mercury emissions from forest fires may also generate health effects (Friedli 2003).

There is significant variability in estimates of the impact depending on assumptions made about the health endpoints, the concentration response functions, and the valuation estimates. If, for example, we do not use 30 μg/m^3 as a low-end cutoff for evaluating health effects, the outcomes will be doubled (approximately). Similarly, the values for the 90th percentile from the sensitivity analysis are approximately double the estimates of central tendency, providing some indication of the impact of relaxing some of the conservative assumptions made. On the other hand, one must also consider that the lower tail of the distribution arising from the sensitivity analysis is approximately 40% of the mean values provided. Clearly, there is a need for research to narrow this range of uncertainty.

Reductions in air quality at different times of the day may have very different health effects. In our analysis we assume a concentration holds for a 24 h period. If instead a large concentration of $PM_{2.5}$ occurs for 1 h and normal levels prevail for the remaining hours in the day, our estimates will overstate the health impacts. Furthermore, it is possible that concentrations at night will generate lower health effects than those generated by increased concentrations during the day. These issues suggest that a more detailed analysis based on hourly concentrations may provide significantly more accurate estimates of impact.

While it would be instructive and desirable to examine actual health records for the dates of these smoke events, such an examination is challenging to conduct. There are a variety of factors generating mortality, hospital visits, and other health effects in addition to the smoke effects. Also, population sizes in northern Canadian cities and towns are relatively small. Therefore, there is considerable noise in the health estimates and the sample sizes are likely not large enough to identify mortality and morbidity effects from short-term fire events. Thus, we rely on the simulation approach where the concentration response models are based on larger populations and detailed analyses of health information. Nevertheless, continued research is necessary to ground truth these simulation results.

Conclusion

Based on the simulation modeling presented in this paper, the health costs of the Chisholm fire were economically significant. This is not to suggest that the health costs of all fires are as high relative to other impacts. The Chisholm fire significantly affected large urban areas, while most fires do not. In most cases, fire management agencies are doing their utmost to prevent, contain, and extinguish fires near human populations. Hence, health effects are being indirectly considered,and it is not clear that adding health as a further value at risk would be an important additional decision factor in the management of individual fires. The exception could be situations with multiple fires that are far away from populations, but have the potential to create high particulate concentrations during certain wind directions and dispersion conditions (e.g., Wotawa and Trainer 2000). However, the evaluation of health effects can provide insights into the returns on investment to fire management, especially when evaluated over alternative strategies for managements over large areas and time periods. The approach used in this case study is being extended to construct a map that identifies key areas in Alberta that will be at risk of experiencing high health costs resulting from forest fires. This map will overlay the various values at risk associated with fires and will be used to aid in fire management resource planning.

Acknowledgements

This research was funded by the Sustainable Forest Management Network. The authors would like to thank Paul De Civita of Health Canada and Dean Stinson O'Gorman of Environment Canada for their support of the project and for providing access to the Air Quality Valuation Model. Also thanks to Robert Field and two anonymous reviewers for helpful comments and suggestions.

References

Adamowicz, V., Dales, R., Hale, B.A., Hrudey, S.E., Krupnick, A., Lippman, M., McConnell, J., and Renzi, P. 2004. Report of an expert panel to review the socio-economic models and related components supporting the development of Canada-Wide Standards (CWS) for particulate matter (PM) and ozone to the Royal Society of Canada. J. Toxicol. Environ. Health Part B Crit. Rev. **7**(3): 147–266.

Alberta Sustainable Resource Development. 2001. Chisholm fire entrapment investigation. Available on line at http://fire.feric.ca/other/chisholmwildfireentrapmentinvestigation.pdf. [accessed April 2004].

American Lung Association. 2001. Selected key studies on particulate matter and health (1997–2001) [online]. Available from http://www.lungusa.org/atf/cf/{7A8D42C2-FCCA-4604-8ADE-7F5D5E762256}/PM_HEALTH97-01.PDF [accessed April 2004].

Cheng, L., Mcdonald, K.M., Angle, R.P., and Sandhu, H.S. 1998. Forest fire enhanced photochemical air pollution. A case study. Atmos. Environ. **32**: 673–681.

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

Chisholm Fire Entrapment Investigation. 2001. Chisholm Fire Entrapment Investigation [online]. Available from http://fire.feric.ca/other/CHISHOLMWILDFIREENTRAPMENTINVESTIGATION.pdf [accessed March 2006].

Chisholm Fire Review Committee. 2001. Chisholm fire review committee final report. Chisholm Fire Review Committee, Edmonton, Alta. http://www3.gov.ab.ca/srd/forests/chisholm/pdfs/Chisholm.pdf [accessed 22 December 2004].

Environment Canada. 2004. Air pollutant emissions. Environment Canada, Gatineau, Que. Available online at http://www.ec.gc.ca/pdb/ape/cape_home_e.cfm [accessed April 2004].

Flannigan, M.D., Logan, K.A., Amiro, B.D., Skinner, W.R., and Stocks, B.J. 2005. Future area burned in Canada. Clim. Change. **72**(1–2): 1–16.

Friedli, H.R., Radke, L.F., Lu, J.Y., Banic, C.M., Leaitch, W.R., and MacPherson, J.I. 2003. Mercury emissions from burning of biomass from temperate North American forests: laboratory and airborne measurements, Atmos. Environ. **37**(2): 253–267.

Forestry Canada Fire Danger Group. 1992. Development and structure of the Canadian Forest Fire Behavior Prediction System. For. Can. Inf. Rep. ST-X-3.

Fowler, C.T. 2003. Human health impacts of forest fires in the southern United States: a literature review. J. Ecol. Anthropol. **7**: 39–63.

Freeman, A.M. 2003. The measurement of environmental and resource values theory and methods. Resources for the Future, Washington, D.C.

King, M.D., Menzel, W.P., Kaufman, Y.J., Tanré, D., Gao, B.-C., Platnick, S., Ackerman, S.A., Remer, L.A., Pincus, R., and Hubanks, P.A. 2003. Cloud and aerosol properties, precipitable water, and profiles of temperature and water vapor from MODIS. IEEE Trans. Geosci. Remote Sens. **41**: 442–458.

Koop, G., and Tole, L. 2004. Measuring the health effects of air pollution: To what extent can we really say that people are dying from bad air? J. Environ. Econ. Manage. **47**: 30–5.

Lippmann, M., Ito, K., Nadas, A., and Burnett, R.T. 2000. Association of particulate matter components with daily mortality and morbidity in urban populations. Res. Rep. Health Eff. Inst. **95**: 5–72, 73–82.

Nash, T., Alberta Forest Protection Advisory Committee, and KPMG. 1999–2000. Alberta fire review 1998 final report. KPMG, Edmonton, Alta.

Ostro, B., and Chestnut, L. 1998. Assessing the health benefits of reducing particulate matter air pollution in the United States. Environ Res. **76**: 94–106.

Pope, C.A., III, Burnett, R.T., Thun, M.J., Calle, E.E., Krewski, D., Ito, K., and Thurston, G.D. 2002. Lung cancer, cardiopulmonary mortality, and long-term exposure to fine particulate air pollution. JAMA, **287**: 1132–41.

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, 27–29 July 1999. Available from http://www-cep.ensmp.fr/francais/themes/impact/pdf/Rabl1999c.pdf.

Rittmaster, R. 2004. Economic analysis of the human health effects from forest fires. M.Sc. thesis, Department of Rural Economy, University of Alberta, Edmonton, Alta.

Samet, J.M., Dominici, F., Zeger, S.L., Schwartz, J., and Dockery, D.W. 2000. The National Morbidity, Mortality, and Air Pollution Study. Part I: methods and methodological issues. Res. Rep. Health Eff. Inst. **94**: 5–14; 75–84.

Samet, J.M., Zeger, S.L., Dominici, F., Curriero, F., Coursac, I., Dockery, D.W., Schwartz, J., and Zanobetti, A. 2000. The National Morbidity, Mortality, and Air Pollution Study. Part II: morbidity and mortality from air pollution in the United States. Res. Rep. Health Eff. Inst. **94**: 5–70; 71–90.

Schwartz, J. 1994. Air pollution and hospital admissions for the elderly in, Detroit, Michigan. Am. J. Respir. Crit. Care Med. **150**: 648–655.

Schwartz, J. 2000. Assessing confounding, effect modification and thresholds in the association between ambient particles and daily death. Environ. Health Perspect. **108**(6): 563–568.

Sustainable Resource Development. 2003. Alberta's forests: wildfire in alberta's forests. Government of Alberta, Edmonton, Alta.

US EPA. 2004. Six common air pollutants: particulate matter [online]. Available from http://www.epa.gov/oar/urbanair/6poll.html [accessed 3 November 2004].

Wotawa, G., and Trainer, M. 2000. The influence of Canadian forest fires on pollutant concentrations in the United States. Science (Washington, D.C.), **288**: 324–328.

Appendix A. Dispersion calculations of particulate concentrations

Two components are needed to estimate the level of particulate matter at a given distance. The first is an emission release term, Q (g/min), which describes the quantity of $PM_{2.5}$ that is released from the fire.

$$[1] \qquad Q = RAE$$

where R is the rate of spread of the burned area (ha/min), A is the amount of fuel consumed (kg/ha), and E is an emission factor giving the amount of $PM_{2.5}$ released per unit of fuel burned (g/kg).

The second component is a dispersion term, D (min/m^3):

$$[2] \qquad D = \frac{1}{WUH}$$

where W is the plume width (m) generated by a statistical analysis predicting the width of the smoke plume as a function of distance and the atmospheric dispersion index (described below), U is the wind speed (m/min), and H is the mixing height (m). The mass of $PM_{2.5}$ per cubic metre of air, C (g/m^3), is then:

$$[3] \qquad C = QD$$

For the Chisholm case study, R and A were taken from a fire report (Alberta Sustainable Resource Development 2001), but these could alternatively have been estimated using the Canadian Fire Behaviour Prediction System (Forestry Canada Fire Danger Group 1992). This system required inputs of fire weather and fuel type. Two spread rate measurements were used from the second day of the fire (24 May 2001), with an average spread rate of 5.65 ha/min.

Smoke is an excellent tracer and offers the possibility of calibrating a model of dispersion in the lateral dimension to estimate the parameter W. This was done by analyzing a population of 35 smoke plumes from western Canada and the northwestern United States during the 2000–2003 fire seasons. A relationship was developed between W and distance from the source, which represents daytime dispersion for a range of conditions of wildfires. The plumes were observed using NASA's Rapid Response System, which provides access to Moderate Resolution Imaging Spectroradiometer (MODIS) satellite images (see King et al. 2003; http://modis.gsfc.nasa.gov/about/specifications.php). The MODIS images are 1 km resolution and provide visible smoke plumes as well as indicating thermal hotspots from the middle infrared and thermal infrared bands. While the hotspots do not indicate fire size, they are a good indicator of the general fire location. Weather data were available for 21 of these plumes. This empirical smoke-plume data set is limited to a finite range of conditions. Hence, we developed a regression between W and the atmospheric dispersion index, ADI, to be able to calculate W for individual fires. The ADI is a standard meteorological station output used to indicate ease of dispersion. Our regression relationship between W and ADI had an $R^2 = 0.77$ using a random effects model (Rittmaster 2004). We then applied this model to the Chisholm fire case study.

The vertical plume mixing parameter, H, was assumed to equal the atmospheric mixing height. This assumes that the plume is uniformly distributed vertically and that it does not penetrate the top inversion layer. For some intense fires, this may not be true, since vigorous convective mixing can be created by a fire such that the convective column penetrates the atmospheric boundary layer. For such cases, our model would overestimate the surface concentration because some of the plume is more widely spread at higher altitudes. The mixing layer height estimate came from standard meteorological measurements at the Slave Lake weather station, approximately 70 km from the fire. Additional data were obtained from weather stations at Whitecourt (130 km away) and Lac la Biche (120 km away).

The weather during the Chisholm fire was characterized by a high temperature of 27 °C, low humidity, and winds gusting to 50 km/h (Chisolm Fire Entrapment Investigation 2001). These extreme weather conditions combined with a dry, flammable forest contributed to the fire event. The average ADI value on 24 May 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/h. For this case study the daily averages over the three weather stations are used for ADI (27), mixing height (806 m), and wind speed. Figure 2 shows the predicted path of the smoke plume from the smoke dispersion model.

A geographic information system (ArcView) allowed for the dimensions of the smoke plume to be overlaid onto a map of census subdivisions (see Fig. 2). 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 A1 shows the predicted concentration levels at various distances from the fire.

As a practical concentration cutoff, we only assessed health impacts for concentrations greater than 30 μg/m^3. Therefore, the approach provides a conservative estimate from health impacts because individuals exposed to levels below 30 μg/m^3 are not included in the analysis.

Fig. A1. Particulate matter concentration predictions over distance from source during the Chisholm fire, at actual atmospheric and "worst-case" levels. The worst-case scenario represents a fire situation that has low levels of dispersion and high emission release rates. The data related to the worst-case scenario were obtained from confidence intervals of fire behavior variables. For comparison, the two dotted lines represent a confidence interval based on the random effects parameter that was used as part of the smoke dispersion model for the Chisholm fire. The parameters for the cases are as follows: Chisholm mixing height, 806 m; worst-case mixing height, 900 m; Chisholm wind speed, 15 km/h; worst-case wind speed, 25 km/h; Chisholm atmospheric dispersion index (ADI), 27; worst-case ADI, 25; Chisholm spread rate, 5.65 ha/min; worst-case spread rate, 2.6 ha/min; Chisholm fuel consumption, 2.35 kg/m^3; worst-case fuel consumption, 3.33 kg/m^3; fire emission factor, 13.5 g/kg.