Building and Environment 89 (2015) 336–344

Contents lists available at ScienceDirect

Building and Environment

journal homepage: www.elsevier.com/locate/buildenv

Impact of residential building regulations on reducing indoor exposures to outdoor $PM_{2.5}$ in Toronto

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ARTICLE INFO

Article history:
Received 23 October 2014
Received in revised form
24 February 2015
Accepted 8 March 2015
Available online 14 March 2015

Keywords:
Cost-benefit analysis
Exposure
Public health
Outdoor $PM_{2.5}$
Residential building regulations

ABSTRACT

We conducted a cost benefit analysis of residential building regulations on reducing the exposure to outdoor $PM_{2.5}$ in Toronto. By combining a combined mass balanced model, a time-weighted activity exposure model, epidemiological based concentration-response, and monetary valuation method, various morbidity and mortality outcomes were estimated for different residential building scenarios. It was found that retrofitting residential buildings to comply with minimum building code regulations can save US$2.3 billion/year in health care. Citywide adoption of R2000 standard from current housing corresponds to US$3.8 billion/year. Use of mechanical HVAC systems, improved filtration (with recirculation) and tighter building envelope (lower infiltration rate) were noted as the key factors that influence $PM_{2.5}$ exposure reductions and health impacts. Estimated costs of retrofitting existing homes to adopt these regulations were about 2.3–2.9 times of the health savings. Citywide use of filters with better efficiency is anticipated to lead to annual health savings significantly exceeding the capital and running costs.

1. Introduction

Recent epidemiological studies have reported adverse human health effects associated with exposure to ambient particulate matter (PM) [1]. Ambient concentrations of PM have been associated with a wide range of health impacts including asthma, heart attacks, and hospital admissions. An important PM-related health effect is premature mortality; in particular, increases in concentrations of PM with aerodynamic diameters of 2.5 μm or less ($PM_{2.5}$) have been closely associated with increases in cardiopulmonary and lung cancer mortalities in exposed populations [2]. In Toronto, there were 226 premature deaths, 555 respiratory and 812 cardiac hospital admissions associated with from outdoor PM exposure [3]. Since then, many policies have been implemented to mitigate human exposures of ambient $PM_{2.5}$ to reduce the citywide health impact. These policies focused on outdoor related strategies such as reducing emission strengths of stacks and using "cleaner" alternative sources. However, a person's total $PM_{2.5}$ exposure (given as a sum of ambient and indoor exposures) are more influenced by indoor environments, where most people spend 90% of their time. An alternative strategy can involve building intervention policy that reduces exposures of $PM_{2.5}$ in indoor environment.

In Canada, there are various codes, guidelines and regulations that have the potential to influence a person's indoor exposure to outdoor $PM_{2.5}$ in residential buildings. The National Building Code (NBC) addresses the design and construction of new buildings and the substantial renovation of existing buildings [4]. The NBC is only enforced following adoption or adaption by the provincial/territorial authorities having jurisdiction. In the province of Ontario, building construction and equipment is regulated by NBC and the Ontario Building Code (OBC) [4,5]. These codes establish the limiting design factors such as minimum ventilation rates per person, minimum building envelope insulation values and guidance on use of filters for safety and fire protection purposes. Residential buildings adopting the building codes typically install Heat Recovery Ventilators (HRVs). Future revisions of NBC include possible reduction of $PM_{2.5}$ using air cleaning devices in the HVAC system if the outdoor air pollution levels are above ambient threshold levels.

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http://dx.doi.org/10.1016/j.buildenv.2015.03.010
0360-1323/

In addition to the above mentioned building codes, the R-2000 standard is a voluntary standard meant to exceed building code requirements, regulating and promoting high energy efficiency and improved air quality initiatives by offering incentives on retrofit and new construction. Typical R-2000 houses have high-efficiency heating and ventilation systems (e.g. installation of HRV and exhaust fans certified by the Home Ventilating Institute), additional insulation, and an airtight building envelope. To comply with the R-2000 standard, the calculation of the ventilation energy is based on the assumption that a house is ventilated at a monthly average rate of 0.30 normal air changes per hour, with a minimum of 90 m^3/h and a maximum of 360 m^3/h, of combined natural and mechanical ventilation. R-2000 houses normally have selected features related to indoor air quality (IAQ) which include items such as low-emission building materials, indoor moisture control, sub-slab depressurization system and air filtration using high performance filters [6].

The parameters affecting the indoor exposure of outdoor $PM_{2.5}$ include the design and use of HVAC systems to provide adequate ventilation, airtight building envelope to reduce infiltration, utility of efficient filtration for increased removal of $PM_{2.5}$, and the airflow rates that pass through the filtration devices. While the provision of residential ventilation rates in building regulations is needed to control the levels of other priority indoor contaminants, care should be taken to ensure that the ventilation air contains reduced $PM_{2.5}$ levels. With higher ventilation rates, more outdoor $PM_{2.5}$ can be brought indoors [7]. Air filtration can be employed to reduce $PM_{2.5}$ levels in the residences [7,8]. Furthermore, the location of filters within the HVAC system can also impact the removal rates of $PM_{2.5}$. A high efficiency filter placed in lower flow rate fresh air ducts may have the same or lower PM removal rate as a lower efficiency filter placed in higher airflow rate return air ducts [7]. Outdoor $PM_{2.5}$ can also be transported indoors via leaks in the building envelope, indoor–outdoor pressure difference, the amount of airflow through openings, and air exchange rate (AER). Particle penetration is higher in leaky buildings suggesting improved airtightness associated with higher efficient building envelope systems can reduce the indoor exposure to outdoor $PM_{2.5}$ [7]. In summary, different regulations parameters may increase or decrease outdoor $PM_{2.5}$ exposures in residences. It is unclear what the combined effects of implementing the building regulation are on indoor exposure to outdoor $PM_{2.5}$ and its health impact if implemented at the population level.

To address this knowledge gap, we conducted a modeling study to investigate the effects of implementing the NBC residential codes and R-2000 residential standard on the indoor exposures of $PM_{2.5}$ of outdoor origin and on the health impact at the population level. Due to the complexity of the investigation, we limited the scope of study to the province of Ontario, Canada and in particular to the city of Toronto which represents much of the metropolitan population in Ontario and a significant portion of all Canada.

2. Materials and methods

2.1. Model scenarios

We adopted the framework derived by the study of Zuraimi [9] to determine the public health impact of the associated $PM_{2.5}$ and the time-weighted exposures in different buildings with mechanical ventilation and filtration systems in a population. A typical single detached house design is used as a model subject upon which exposure and public health impact estimates under:

1) a baseline situation with no regulatory compliance;
2) an intervention situation with compliance of building codes in Ontario; and
3) an intervention situation adopting the R-2000 standard was applied and evaluated.

The baseline was a building with the average size of 370 m^2 including the basement space [10]. Under this baseline situation (Model 1), existing houses were modeled with forced air furnace heating for the winter months, and natural cooling and ventilation for the summer months [11]. Model 1 buildings were considered built in the ′60s and ′70s. Building envelope was assumed to have high air leakage and low envelope insulations including single-glazed windows [11,12]. Under the intervention with compliance of the building codes (Model 2), average new houses were modeled with forced air heating and cooling with medium efficiency furnace and standard furnace filter (MERV8 rating) [4]. HRV was assumed to be installed in simplified connection to the furnace return duct. Building envelope was assumed with minimum air tightness to code requirements and minimum building code envelope insulation and windows (i.e. R20 walls, R40 ceilings, double glazed windows, R12 basement walls as per Ontario building code). As a subset of Model 2, we simulated an intervention (Model 2a) where the NBC include possible reduction of $PM_{2.5}$ using air cleaning devices in the HVAC system. Model 2a differs from Model 2 by replacing standard furnace filters with MERV15 high efficient filters. For intervention situation adopting the R-2000 standard (Model 3), R-2000 houses were modeled with high efficiency HVAC equipment (95% furnace), forced air system for both heating and cooling and filter with MERV13 rating as one of the selected pick list [6]. Building envelope was assumed with relatively higher air tightness and higher envelope insulation (R30 walls, R50 ceilings, R22 basement walls) including double-glazed windows with argon fill and e-coating.

2.2. Building parameters and models

The residential indoor concentrations C_{IA}, can be determined from the outdoor $PM_{2.5}$ concentrations using a mass balanced model (Eq. (1)). The assumptions used in this model are based on a single well-mixed compartment and steady-state conditions.

$$C_{IA} = \frac{S/V + C_{OA}\cdot[Q_{OA}\cdot(1-\varepsilon_S)/V + Q_I\cdot P/V]}{Q_R\cdot\varepsilon_S/V + (Q_{OA}+Q_E)/V + \beta} \quad (1)$$

where, Q_R is the recirculation airflow rate through the supply air filters; Q_{OA} is the outside airflow rate through the supply air filters; Q_I is the rate of infiltration of unfiltered air; Q_E is the rate of exfiltration; ε_S is the $PM_{2.5}$ removal efficiency of the supply air filters; β is the rate of $PM_{2.5}$ depositional loss which is the product of depositional velocity and surface area to volume ratio; S is the indoor $PM_{2.5}$ of outdoor origin emission rates; V is the effective volume of the indoor environment; and P is the penetration factor for $PM_{2.5}$ entering via air infiltration. C_{OA} is the outdoor $PM_{2.5}$ concentration. Although it is well known that other indoor sources exist and can contribute to indoor $PM_{2.5}$ levels, the following analysis is focused on outdoor $PM_{2.5}$. Therefore, the S term is reduced to zero.

The air flow rates in Eq. (1) are determined based on the sizes of the building models and HVAC systems. The approach to sizing the residential input models was based on local mechanical engineering practice and application of building codes and standards. First, outdoor temperatures were determined using the weather data for summer (defined as June, July, August: 31 °C DB and 23 °C WB) and winter (defined as December, January, February: −20 °C) seasons in Toronto. The heating and cooling loads were estimated as 88 W/m^2

and 44 W/m^2 for summer and winter, respectively. The heating and cooling loads of new buildings following the codes were estimated using factors 63 W/m^2 and 28 W/m^2, respectively. The heating and cooling loads of R-2000 homes were estimated using the factors 60 W/m^2 and 22 W/m^2, respectively. From the resulting heating and cooling loads, the supply and return air flow-rates were determined for the new building code based homes and new R-2000 homes. Supply cooling flow rates were estimated using a factor of 680 m^3/h per 1000 kg of refrigeration as minimum [5]. Outside airflow rate, assumed through natural ventilation via open windows, for existing homes during summer season was calculated using Eq. (2) (ASHRAE, 2013):

$$Q_{OA} = C_e A_o U \tag{2}$$

where C_e is the effectiveness window opening, A_o is the total area of window opening, and U is the average wind speed. We used average summer wind speed of 3.0 m/s obtained from Environment Canada, total area of inlet openings of 1 m^2, and an effectiveness opening of 0.35. Outdoor airflow rates for building code and R-2000 homes were estimated using a room count approach - total capacity equal to the sum of the individual room ventilation requirements amounting to a total of 305 m^3/h [4]. Infiltration rates during winter and summer seasons were calculated using Eq. (3) [13]:

$$Q_I = \frac{A_e}{1000}\sqrt{C_s \Delta T + C_w U^2} \tag{3}$$

where A_e is the effective air leakage area, C_s is the stack coefficient, C_w is the wind coefficient, and ΔT is the average indoor–outdoor temperature difference. We used average wind speeds and outdoor temperatures in Toronto for summer and winter seasons obtained from Environment Canada, effective leakage areas of Ontario houses obtained from the study by Hamlin and Gusdorf [14], C_s value for a one storey house, and C_w value for a shelter class 4.

For all models in this study, we used the mean $PM_{2.5}$ penetration factor of 0.73 as measured for 37 residences by Williams et al. [15]. We followed the approach by Macintosh et al. [16] to use a β value of 0.5 h^{-1} derived from an experimental study by Thatcher et al., [17]. For $PM_{2.5}$ removal efficiencies of various MERV filters, we relied on the values provided by McDonald [8]. USEPA ambient particle data was used to determine how different MERV rated filters [18] would perform in terms of outdoor $PM_{2.5}$ removal efficiencies. Input parameters for the model residential buildings under existing homes, complying with Building Code regulations and adopting R-2000 standard homes are summarized in Table 1.

2.3. Exposure calculation

Outdoor $PM_{2.5}$ concentration data used to predict indoor concentrations in various models for Toronto were obtained from the National Air Pollution Surveillance (NAPS) network [19]. The Toronto monitoring site is located in the downtown area heavily influenced by local transportation emissions. Annual data under summer and winter seasons for the year 2003–2008 were used.

The micro-environmental model (Eq. (4)) was used to define time-weighted average $PM_{2.5}$ concentrations encountered by individual adults as they pass through different micro-environments [20].

$$C_{TW} = \frac{\sum t_{ij} \cdot C_{ij}}{t_{total}} \tag{4}$$

where, C_{TW} is the time weighted exposure for period i in micro-environment j, t_{ij} are the hours expended in micro-environment j and period i, C_{ij} is the $PM_{2.5}$ concentration in micro-environment j for period i, and t_{total} is the total time. Here, we assume that individuals leave the home routinely and go to another fixed location (i.e. workplace) every day for an appreciable amount of time. To consider other indoor microenvironments, the mean indoor $PM_{2.5}$ concentration of 2.38 μg/m^3 measured in 24 office buildings [21] was used. Time activity patterns for winter and summer seasons within the main microenvironments (residences, office buildings) considered in this study were obtained from Leech et al. [22].

2.4. Health impacts and valuation

The next step is to determine the health impacts associated with estimated reductions in annual time-weighted $PM_{2.5}$ exposures for the population of Toronto. Of particular interest is premature mortality as being recognized by many researchers [23]. In addition, to help communicate the range of health endpoints, C-R functions were also applied for chronic bronchitis, respiratory and cardiovascular hospital admissions and restricted activity days (RAD). Although, other morbidity outcomes have been associated with $PM_{2.5}$ exposures, these were not addressed because of unavailable quantitative data for baseline incidences for Toronto, C-R functions) or both. The standard health impact function [23,24] used is as follows:

$$\Delta y = MP\left(e^{\alpha \Delta x} - 1\right) \tag{5}$$

where M is the baseline incidence; P is the population size; α is the coefficient of C-R functions (i.e. log relative risk obtained from the relevant studies); Δx is the estimated change in the annual time-weighted $PM_{2.5}$ concentration, and Δy is the attributable number of cases of the health effect associated with the change in the $PM_{2.5}$, Δx. The annual time weighted concentrations were obtained by averaging the time weighted concentrations under winter and summer seasons. We referred to the Canadian cohort study [25], which reported a hazard ratio of 1.15 (95% CI: 1.13, 1.16) from non-accidental mortality for each 10 μg/m^3 increase in concentrations of $PM_{2.5}$. The equivalent value of α is 0.0139 per 1 μg/m^3 change in $PM_{2.5}$ exposure. Morbidity-related C-R functions are approved by the Health Canada for evaluating health impact using the AQBAT software [24]. The Ontario mortality and Provincial health planning

Table 1
Indoor air quality (IAQ) model parameters for modeled building scenarios.

	Existing homes		Building code regulations		R2000 standard	
	Winter	Summer	Winter	Summer	Winter	Summer
Recirculation rate, h^{-1}	0.0	0.0	2.8	1.7	2.6	1.3
Outside air rate, h^{-1}	0.0	3.7	0.3	0.3	0.3	0.3
Infiltration rate, h^{-1}	0.3	0.1	0.2	0.1	0.1	0.0
Filter efficiency	0.0	0.0	0.4	0.4	0.8	0.8
Deposition rate, h^{-1}	0.5	0.5	0.5	0.5	0.5	0.5
Penetration factor	0.7	0.7	0.7	0.7	0.7	0.7

databases from the Ontario Ministry of Health were used as the baseline incidence rates [3]. The population and population breakdown that were considered at risk for each scenario was taken from Canadian census information for 2001 [3,26].

The monetary valuations of health outcomes were obtained from Health Canada endorsed endpoint valuation (EPV) used in the AQBAT software. In AQBAT, two endpoint valuations relate to either mortality or morbidity outcomes: for mortality, the value of a statistical life (VSL) is used, which is a measure of people's willingness to accept different levels of risk, and for morbidity, the combined value of lost wages, cost of treatment, averting expenditures and pain and suffering related to morbidity outcomes. The C-R coefficients and their health valuation used in the analysis are given in Table 2.

2.5. Economic implications

The models were then compared under 5 case scenarios (Table 3). Cases 1 and 2 evaluate the impacts of citywide adoption of residential building code regulations from existing homes. Case 3 is computed to evaluate the impact of citywide adoption of R-2000 standard from existing homes. Case 4 evaluates the impact of improved filtration while conforming to residential building code. The last scenario (Case 5) evaluates the impact of building code homes adopting R-2000 standard. We estimated the potential economic benefits from reduced indoor exposures to outdoor $PM_{2.5}$ by multiplying economic costs by the proportion of the health effect estimated to be preventable through improvements under different case scenarios.

We next determined the capital and operating energy costs under the case scenarios above. The estimation of capital costs for material and labor associated with attic, basement, window, wall cladding, gas furnace, air-conditioning (AC) unit and heat recovery ventilators (HRV) retrofits of Toronto houses were obtained following the methodology introduced by Lio and associates for the Ontario Ministry and Municipal Affairs and Housing [31]. Cost to adopt R-2000 standard for an Ontario building code house was obtained from Gray et al. [32]. The costs for filters were obtained from the USEPA study [33] after the conversion from US to Canadian dollars using an exchange rate of 1.13 for the year 2008. Operational costs for the various models were determined using building energy simulation via the HOT2000 software [34]. The software is a Canadian standard for evaluating the energy performance of houses and multi-unit residential buildings and has been put through rigorous International Energy Agency BESTEST for testing and diagnosing its energy simulation capabilities [34].

We then analyzed the energy consumption and energy required by the space heating and cooling, water heating, and lighting and appliances of the home on an annual basis. All base loads from occupancy and appliances were standard defaults within the software and corresponded to Canadian averages. The buildings were simulated in a Toronto climate and all had the same orientation and layout. The total costs in Toronto under different case scenarios were calculated by multiplying the capital and operating costs for a single house with the total number of houses in Toronto. Here, we used the total number of population of Toronto that was used to derive the baseline incidence rates [3] with an assumed average number of persons in household of 3.3 [26]. All costs were converted to 2012 Canada dollar using an annual inflation rate of 3%. The benefit to cost ratios for cases 1 to 5 were calculated by dividing the health savings with the capital and operational costs.

2.6. Sensitivity analysis

A sensitivity analysis was conducted to quantify the sensitivity of the calculated net benefits of improved building regulations to the inputs and assumptions used in the models. It was assumed that the effect of each parameter is independent and linear, and they are additive. The alternative values of each input parameter were entered into the model while holding all other variables constant. The corresponding results are reported in terms of the percentage difference in benefit-cost ratios (BCR) compared with the baseline estimation. We examined the sensitivity of our results to twelve parameters including total window area, the effectiveness of opening, penetration factor, effective air leakage area, building volume, $PM_{2.5}$ filter efficiency, outdoor temperature, $PM_{2.5}$ concentration, wind speed, mortality effect estimates for $PM_{2.5}$ exposure, health outcome valuations for mortality, and chronic bronchitis. For normally distributed numerical parameters (e.g. outdoor $PM_{2.5}$ concentration and temperature), "high" and "low" values were calculated as the means $\pm$ 2.33 standard deviations, the range that encompasses 99% of the values assuming normally distributed data [35]. For other variables such as window opening, we proposed values to reflect changing the aperture to ±50% of the total window area. We used the lower range of diagonal wind and upper and of perpendicular wind [13] to obtain the impact of opening. For the penetration factor, we used the maximum and the minimum penetration factors of 1 and 0.11 [15] for homes under

Table 2
Concentration-response (C-R) coefficients and health valuation of various health endpoints.

Health endpoint	Concentration-response coefficients, α (standard error)	Location	References	Health valuation	References
Non-accidental mortality	0.0139 (0.0009)	Canada	[25];	$6,500,000	[24]
Chronic bronchitis	0.00132 (0.0068)	US	[24,27]	$266,000	[24]
Hospital admission – cardiovascular	0.0711 (0.0170)	Canada	[24,29]	$5200	[24,28]
Hospital admission – respiratory	0.0754 (0.0132)	Canada	[24,29]	$4200	[24,28]
Restricted activity days	0.00481 (0.00101)	US	[24,30]	$48	[24]

Table 3
Case comparisons for different residential regulations, standards, air tightness and filtration scenarios.

Case scenarios	Comparison	Description
Case 1: building code regulation effect	Model 1–model 2	Comparing the existing homes model with building code regulations model
Case 2: future building code regulation effect	Model 1–model 2a	Comparing the existing homes model with future building code regulations model
Case 3: R2000 standard effect	Model 1–model 3	Comparing the existing homes model with R2000 standard model
Case 4: enhanced filtration effect	Model 2–model 2a	Comparing the building code regulations model with future regulations incorporating enhanced filtration
Case 5: enhanced filtration and air tightness effect	Model 2–model 3	Comparing the building code regulations model with R2000 standard model

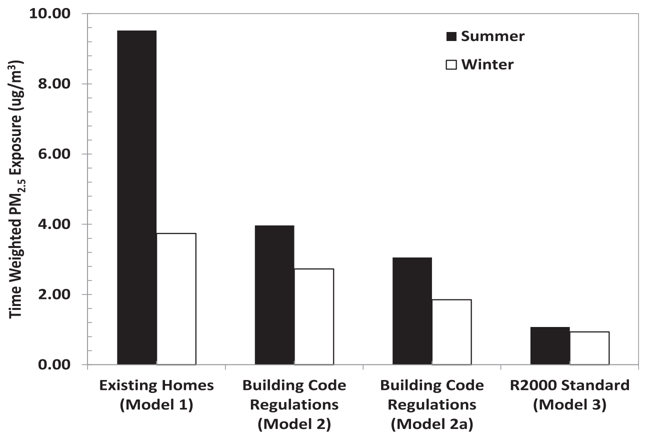

Fig. 1. Time weighted $PM_{2.5}$ exposures under modeled residential building types in Toronto for summer and winter seasons.

Models 1 and 3, respectively. The effective air leakage areas were obtained from the study by Hamlin and Gusdorf [14]; we considered data from Atlantic homes for “high” since they are the ‘tightest’ homes among all provinces, and British Columbia homes for “low” as they are the ‘leakiest’. For mortality effect estimates, we adopted the lower and upper range (0.7% and 1.6%) provided by expert solicitation [36]. Low and high EPV estimates provided by Health Canada [24] were used to evaluate impacts of mortality and chronic bronchitis economic valuation.

3. Results

3.1. Exposure modeling

The seasonal variability of time weighted $PM_{2.5}$ exposures calculated for the modeled building scenarios is shown in Fig. 1. It is observed that the highest $PM_{2.5}$ population exposure is when the model is simulated for existing homes with average time weighted exposures at 9.5 and 3.7 μgm^{-3} for summer and winter seasons, respectively. The difference between summer and winter exposures (5.8 μgm^{-3}) in existing homes is due to the combined impact of lower fraction of time spent in residences (0.58 vs 0.69) and higher outdoor air exchange rates attributed to natural ventilation (open windows) during summer. Under building code regulations, exposures were reduced to 4.0 and 2.7 μgm^{-3} (mean percentage reduction of 58 and 36%) for summer and winter respectively. This reduction was greater in summer since existing homes have natural cooling through open windows instead of forced air systems which limited the protective nature of the building envelope. The reduction of time weighted $PM_{2.5}$ concentration exposure in new homes constructed to minimum building code requirements compared to existing homes in the winter is attributed to tighter building envelope (lower infiltration rate) and filtration in forced air systems through recirculated air (Table 1). An additional benefit is the provision of higher ventilation rate (0.3 h^{-1}) under building code based homes. Lower exposure during winter for building code regulations is attributed to the lower fraction of time spent indoor residences and better $PM_{2.5}$ removal through higher recirculation rates going through the filters.

Comparison between Models 2 and Model 2a shows that replacing standard furnace filters with MERV15 filters while still complying to building code regulations could further reduce the $PM_{2.5}$ exposures by 19 and 28% for summer and winter seasons, respectively. The higher $PM_{2.5}$ removal in winter is due to the higher recirculation rates through the filters and more time spent indoors. Adopting R-2000 standard for residences result in the lowest $PM_{2.5}$ exposures at about 1.0 and 0.9 $\mu g\ m^{-3}$ for summer and winter, respectively. This took place despite the decrease in air flow rates of about 100 cfm due to the reduced loads in R-2000 homes for high efficiency heating (1759 cfm (2990 cmh) vs 1852 cfm (3150 cmh). These reductions constitute mean percentage values of 90%–75% when compared to the time weighted $PM_{2.5}$ exposures in current homes; 75%–67% when compared to time weighted $PM_{2.5}$ exposures in homes adopting building code regulations. Comparing Model 2a with Model 3 shows further $PM_{2.5}$ exposure reductions, and they are attributed to the tighter building envelope (lower infiltration rate and penetration factor) despite using MERV13 rated filters only.

3.2. Health impacts and cost

As presented in Table 4, the calculated non-accidental mortality cases are the highest for existing homes. For homes conforming to building code requirements, there is about 53% of decrease in the annual risk of mortality compared to existing homes. Conversion from MERV8 to MERV15 filters while complying with minimum building code regulations further reduced the mortality cases by additional 13%. For R-2000 standard homes, annual mortality is about 13% of existing homes. For all morbidity outcomes, the attributed cases are the highest in existing homes, followed by minimum building regulation homes, and R-2000 standard homes. The total economic health costs associated with $PM_{2.5}$ exposures were approximately $4.4 billion/year for existing homes; largely driven by the associated mortalities as AQBAT uses a central value of statistical life (VSL) of $6.5 million. The lowest total mortality and morbidity cost can be seen for R-2000 standard homes ($0.6 billion/year).

3.3. Estimated potential economic benefits

Retrofitting existing homes to meet minimum building code regulations would result in a saving of $2.1 billion/year. Enhanced

Table 4
Attributable number of cases and costs for various health endpoints associated with $PM_{2.5}$ exposures under different residential models.

Health endpoint	Attributable number of cases (ANC)[a]				Health economic cost[a]			
	Existing homes (model 1)	Building code regulations (model 2)	Building code regulations (model 2a)	R2000 standard (model 3)	Existing homes (model 1)	Building code regulations (model 2)	Building code regulations (model 2a)	R2000 standard (model 3)
Non-accidental mortality	630	310	230	90	$4,074,000,000	$2,011,000,000	$1,464,000,000	$594,000,000
Chronic bronchitis	60	30	20	10	$15,000,000	$8,000,000	$6,000,000	$2,000,000
Hospital admission – cardiovascular	5330	2380	1690	660	$28,000,000	$12,000,000	$9,000,000	$3,000,000
Hospital admission – respiratory	3540	1570	1110	430	$15,000,000	$7,000,000	$5,000,000	$2,000,000
Restricted activity days	1930	970	710	290	$90,000	$50,000	$30,000	$10,000

[a] ANC and health economic cost rounded to whole numbers.

Table 5
Estimated economic savings under different residential regulations, standards, air tightness and filtration scenarios.

	Estimated economic savings ($/yr)				
	Case 1	Case 2	Case 3	Case 4	Case 5
Non-accidental mortality	$2,063,000,000	$2,609,000,000	$3,480,000,000	$547,000,000	$1,418,000,000
Chronic bronchitis	$8,000,000	$10,000,000	$13,000,000	$2,000,000	$5,000,000
Hospital admission – cardiovascular	$15,000,000	$19,000,000	$24,000,000	$4,000,000	$9,000,000
Hospital admission – respiratory	$8,000,000	$10,000,000	$13,000,000	$2,000,000	$5,000,000
Restricted activity days, RAD	$50,000	$60,000	$80,000	$10,000	$30,000
Total savings in millions	$2093	$2648	$3794	$ 554	$1436

filtration with building code regulations increases this saving further by $0.5 billion more per year. Retrofitting existing homes to meet R2000 standard would result in a total saving of approximately $3.8 billion/year. Increasing filtration efficiency in building code based homes is estimated to provide an additional saving of $0.5 billion/yr. Adopting R-2000 standard for building code based homes would result in an estimated saving of $1.4 billion/yr. All these calculations shows that retrofitting existing homes to adopt improved building regulations can result in substantial public health benefits.

The capital and operational costs under different residential regulations, standards, air tightness, and filtration scenarios are presented in Table 6. These values are comparable to those reported by other researchers [12,32]. Comparing Tables 5 and 6 it can be observed that the estimated costs to retrofit existing homes to adopt improved building regulations are about 2.3–2.9 times of the estimated savings. The estimated benefit to cost ratios (BCR) for Cases 1 to 5 are 0.4, 0.4, 0.4, 56.2 and 0.4, respectively. The lower than unity BCR values for cases 1, 2, 3 and 5 suggest that investment to retrofit existing homes to adopt improved building regulations in Toronto on the basis of outdoor $PM_{2.5}$ exposure reduction alone will be difficult to justify. However, using filters with a better efficiency (Case 4) is anticipated to lead to annual savings dramatically exceeding the capital and running costs.

3.4. Sensitivity analysis

Table 7 shows the results reported as the percentage difference in benefit to cost ratios (BCRs) for all health outcomes compared with the results derived from Tables 5 and 6. The references in BCRs for Cases 1 to 5 are 0.4, 0.4, 0.4, 56.2 and 0.4, respectively.

It is noted that high variations of BCRs are derived from changing mortality effect estimate and mortality valuations. Recent expert judgment assessments of the relationship between long-term exposure to $PM_{2.5}$ and mortality increase were estimated to be within the range of 0.7%–1.6% mortality per $\mu g/m^3$ of annual average $PM_{2.5}$ concentration [36]. However, average values of BCRs based on the high range of 1.6% in mortality per $PM_{2.5}$ $\mu g/m^3$ are still below unity for Cases 1, 2, 3 and 5. The mortality values were in the range of $3.5- $9.5 million [24]. Again, the average BCR values calculated using the high range of $9.5 million per mortality case are below unity for Cases 1, 2, 3 and 5. Natural ventilation related parameters such as total window area, the effectiveness of opening and wind speed led to moderate differences in BCRs. The variation in outdoor $PM_{2.5}$ concentrations results in an average difference of about 25% in BCR values. Other parameters did not impact much the calculated BCR values.

Table 6
Estimated capital and operating costs under different residential regulations, standards, air tightness and filtration scenarios.

	Case 1	Case 2	Case 3	Case 4[d]	Case 5
Operating cost ($/yr)[a,b,c]	−$3800	−$3800	−$4000	$50	−$200
Capital cost ($/yr)[c]	$34,500	$34,500	$44,800	$0	$17,500
Net cost ($/yr)[c]	$30,700	$30,700	$40,800	$50	$17,300
Total cost in millions ($/yr)	$6053	$6053	$8045	$ 10	$3411

[a] Heating and cooling costs from HOT2000 software.
[b] Negative costs represent savings.
[c] Operating and capital costs rounded to hundred.
[d] Impact of improved filtration while conforming to residential building code (i.e. replacing standard furnace filters with MERV15 high efficient filters).

4. Discussion

Despite the public health benefits of adopting improved building regulations or standard, the relatively low benefit to cost ratios (0.4–0.5) demonstrate that it is difficult to justify retrofitting all existing homes on the basis of reducing $PM_{2.5}$ exposure alone. More research is needed to clearly show that there are other compelling benefits from adopting improved building regulations or standards, which can be received by different stakeholders throughout the residential building life cycle. For example, considering that ozone is associated with mortality and morbidity outcomes, we can conduct a similar cost benefit analysis of residential building regulations in reducing outdoor ozone exposures indoors. Another example is the cost benefit analysis of the impact of reduced exposure to volatile organic compounds (VOCs) resulting from the use of "green" building materials advocated by the R-2000 standard.

We have identified the use of improved filtration as an important building related parameter influencing $PM_{2.5}$ exposure and health impacts through the adoption of building regulations. However, in calculation of the benefits brought by filtration we should consider actual practice because the adoption of building regulations does not necessarily mean proper implementation. Home owners need to be trained or educated on the right choice of filtration devices and proper maintenance practices to ensure optimum filtration of outdoor $PM_{2.5}$. Filters should be regularly cleaned or replaced as per manufacturer recommendations. Clogged filters may lead to lowered air flow rates, provide sites for microorganisms to grow and cause captured dust particles to react with incoming ozone heterogeneously to produce harmful formaldehyde and ultrafine particle [37]. Furthermore, in our modeling work, we deliberately chose the use of MERV13 filters instead of electronic or catalytic air cleaners as recommended in the R-2000 standard [6]. Filtration technologies should not generate ozone or other harmful pollutants [38]. In addition, it is to be noted that a given MERV rating for a filter is its performance from a standardized test and not an indication of how it will perform in a particular application. In this study, we have attempted to minimize this variation by relying on how rated filters would perform when challenged with outdoor $PM_{2.5}$.

Although we reported a great benefit to cost ratio for the city-wide use of high efficiency filters, there is a need to understand the real impact on building energy consumption. We have simulated the energy use using the HOT2000 software and did not include the

Table 7
Percentage changes in benefit cost ratios when important parameters were simulated with "high" or "low" values in model simulations.

Parameter	Average difference (%) in benefit-cost ratios (BCR)[e]									
	BCR (case 1)		BCR (case 2)		BCR(case 3)		BCR (case 4)		BCR (case 5)	
	"Low"	"High"	"Low"	"High"	"Low"	"High"	"Low"	"High"	"Low"	"High"
Total window area	−10.4	4.1	−8.6	2.9	−6.6	2.6	0.0	0.0	−0.2	−0.2
Effectiveness of opening	1.2	7.0	0.6	5.2	0.3	2.6	0.0	0.0	−0.2	−0.2
Penetration factor[a]	24.3	9.9	16.6	9.7	11.7	11.7	−14.1	6.5	−5.0	14.0
Effective air leakage area	−1.7	−1.7	−1.7	0.6	−2.0	2.6	−2.4	5.9	−2.6	6.9
House volume	−1.7	−4.6	12.0	−10.0	7.1	−8.8	67.2	−28.8	23.5	−9.7
$PM_{2.5}$ filter efficiency[b]	−1.7	1.2	−1.7	0.6	−2.0	0.3	0.5	−0.6	−0.2	−0.2
Outdoor $PM_{2.5}$ concentration[c]	−21.9	21.4	−22.3	23.4	−22.5	23.1	−26.5	27.1	−26.4	25.9
Outdoor temperature[c]	1.2	−7.5	0.6	−6.3	0.3	−4.3	3.6	−0.7	2.1	−0.2
Outdoor wind speed[d]	−10.4	4.1	−6.3	2.9	−6.6	2.6	−0.7	0.8	−0.2	−0.2
Mortality effect estimates	−50.9	15.6	−49.7	14.3	−49.9	16.2	−50.2	15.0	−50.1	14.0
Mortality valuation	−45.1	44.5	−45.1	46.3	−45.3	45.9	−45.5	45.5	−45.4	44.9
Chronic bronchitis valuation	1.2	1.2	0.6	0.6	0.3	0.3	0.0	0.3	−0.2	−0.2

[a] "Low" and "high" values for penetration factors are minimum and maximum penetration factors from the study [15].
[b] ±5% average value used for filter efficiencies.
[c] ±2.33 standard deviation of average summer and winter values.
[d] 25th and 75th percentile of summer and winter values (Environment Canada).
[e] Reference BCRs for cases 1 to 5 are 0.4, 0.4, 0.4, 56.2 and 0.4 respectively.

impact of higher pressure drop associated with denser filter media. Higher-efficiency filters generally have higher pressure drops and are believed to increase energy consumption in residential systems. Recent field measurements of 17 residential and light-commercial forced air cooling systems conducted in Austin, Texas revealed a weak link between higher-efficiency filters and energy use [39]. Considering differences in climate, future research is needed to evaluate whether the findings will be similar for Toronto. Any potential impact of increased pressure drop on net benefits of high efficient filters is anticipated to be insignificant considering the very high BCR values (Case 4). This result is comparable to earlier studies indicating that overall benefits of using particle air filtration in buildings (from decreased occupant morbidity and mortality) are several times larger than associated running costs [9,40].

Indoor sources of $PM_{2.5}$ such as wood-burning, smoking, cooking, respiratory droplets and cleaning which can significantly influence IAQ [7,15,48,49] were not considered in the analysis. We did not expect that indoor sources could substantially impact on our results as our study was focused on the indoor exposure to outdoor particles which has established epidemiological relationships with health outcomes described in Table 2. For these epidemiological studies, where indoor and outdoor sources are independent, the exposure variation owing to indoor-source exposures does not lead to the bias in C-R coefficients. We also did not take into account ultrafine particles (UFPs) which are also markers of anthropogenic emissions and are associated with health outcomes in the cost-benefit analysis. This is due to the lack of monitoring information from the NAPS network and unavailability of C-R functions for the population of Canada.

Admittedly, it is subject to debate that applying steady-state model in the analysis may lead to potential biases. For example, ambient $PM_{2.5}$ and infiltration rates vary diurnally and outdoor-to-indoor transport can be higher in residences with opened windows (Model 1). All of these could have impacts on the calculated results. However, we view that this effect to be not significant considering we evaluated the impact of long term $PM_{2.5}$ exposure as opposed to acute exposure. Similar to the effects of indoor sources on $PM_{2.5}$, C-R coefficients used in Table 2 are derived from long term exposure assessments, where diurnal variation of exposures and infiltration rates as well as building parameters do not bias the epidemiological results. For studies relying on acute outcomes, dynamic models to explore impacts of these diurnal variations are certainly needed as the exposure time scales and lag-time for studying epidemiological relationships between exposures and health outcomes are very short. For example, Dominici and colleagues [41] reported associations with hospitalizations for multiple diseases, using single day average $PM_{2.5}$ while the study by Crouse et al. [25] used a yearly average $PM_{2.5}$ to study long term mortality. In addition, we have used C-R functions and monetary valuations of health outcomes recommended by Health Canada and have adopted their assumptions in using values given in Table 2 [24,28]. The variations of BCR ratios arising from alternative values were assessed using our sensitivity analysis.

4.1. Limitations

There are several limitations in this study. Firstly, we did not include attached or multi-family houses that could increase the representation of houses in Toronto. This is due to the unavailability of some input parameters such as capital cost associated with building retrofits and indoor air quality data (e.g. infiltration rates). In order to extend this framework to other housing types, changes in the current model can be implemented by modifying the parameters presented in the materials and methods section as more data is made available in the future.

Secondly, with the exception of existing homes under summer season scenario, modeled mechanical systems in houses were assumed to run all the time to meet minimum building regulations or R-2000 standard. We acknowledge that there could be different system run-time practices that may change the energy costs and $PM_{2.5}$ exposure calculations. However, we have not included the variation in system run-times in our studies because we have not found data regarding home owner practices. Furthermore, the impact of occupant behavior on the cost-benefit analysis is beyond the scope of the paper.

Thirdly, we provided only central estimates of the results with general assessments of the magnitude of various uncertainties using sensitivity analysis. Indeed, quantitative uncertainty analysis could be improved considering whole reports have been dedicated to quantify them [23,42–44]. However, we believe the sensitivity analysis presented herein is deemed sufficient to examine the robustness of conclusions arising from critical input assumptions. Input parameters for which model results are especially sensitive can be included in future research to refine uncertainty.

Finally, although our sensitivity analysis revealed an average of 25% variation for the average BCR values by using ±2.33 standard deviation of the mean concentration values, the concentrations measured at monitoring stations neglect spatial variation of $PM_{2.5}$. Unfortunately, we are not aware of outdoor particle spatial distribution data in Toronto, which is required to support a more refined model. We had assumed that measurement disparities between distant areas from monitoring stations should be minor for regional pollutants like $PM_{2.5}$ [42,43]. To further assess the accuracy of the model results, we compared the predicted time weighted concentrations to the measurements reported in studies that quantified personal $PM_{2.5}$ levels. Van Ryswyk et al. [45] reported a mean personal exposure of 6.4 and 8.3 μg/m^3 in Windsor, Ontario homes during winter and summer seasons. These personal exposure values included contributions for other indoor sources. If we assume negligible indoor source contribution occurring during sleep time [46], modeled personal exposure during winter and summer seasons were 3.8 and 5.2 μg/m^3 respectively. In another study [47], researchers reported indoor winter and summer concentrations of 7.8 and 10.4 μg/m^3 respectively. Estimates of the ambient fraction of indoor $PM_{2.5}$ exposure of the participants were 59% ambient in winter and 65% ambient in summer. The resulting $PM_{2.5}$ exposure levels were 4.6 and 6.8 μg/m^3, respectively. Comparing the results of these two studies in Windsor with our modeled time-weighted concentrations shows that the values are comparable for winter but higher for summer. A possible reason for this difference is the very low air exchange rates in Windsor in summer (0.18/h). If our modeling had been conducted with this air exchange rate, the modeled time-weighted exposure would have been close to a level similar to that observed in the experimental studies.

5. Conclusion

This study concerned the investigation of retrofitting existing Toronto homes which consisted of hydronic heating for the winter and natural cooling/ventilation for the summer to both minimum national building code and R-2000 high efficiency homes, respectively. All scenarios in these cases investigated retrofit into forced air HVAC systems and tighter building envelope in Canadian weather. The results demonstrate that improved residential building regulations could reduce indoor exposures to outdoor $PM_{2.5}$ and associated health outcomes. We noted the use of mechanical HVAC system, improved filtration (with recirculation) and tighter building envelopes (lower infiltration rate) are the key building related parameters that influence $PM_{2.5}$ exposure and its impacts on population health. Estimated health benefits through adoption of building regulations are lower compared to operating and retrofit costs with benefits to cost ratio values about 0.4. However, citywide use of filters with higher efficiency is anticipated to lead to great annual health savings exceeding the capital and running costs. Through sensitivity analysis, the key parameters influencing the BCR ratios in this study are the use of effect estimates and valuation associated with premature mortality. Although modeling and data limitations exist, the results of this study suggest that modifying building parameter are expected to result in changes in indoor $PM_{2.5}$ exposure and health benefits/costs for the city of Toronto.

Acknowledgment

This research is supported by the University of Waterloo and the Natural Sciences and Engineering Research Council of Canada (NSERC) under Grant# RGPIN311770-2010. The materials and views expressed in the article are solely the authors and do not represent official endorsement.

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