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Wildfire smoke: What is the relationship between outdoor and indoor air pollutants?

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Key Messages

Climate change is driving more frequent and severe episodes of wildfire smoke, raising concerns about indoor infiltration of outdoor air pollutants during these events. This review focused on the available evidence describing indoor versus outdoor concentrations of non-particulate matter (PM) air pollutants during episodes of smoke from wildfires and similar combustion events. The evidence on non-PM pollutants is supported by additional evidence on PM pollutants, including factors that influence indoor infiltration and strategies to reduce indoor exposure to wildfire smoke.

Overall implications

- Buildings offer meaningful but incomplete protection during smoke episodes.
- There are few studies examining non-PM pollutant infiltration and effective interventions.
- Evidence shows that combining multiple strategies (closing windows, PACs, HVAC use) provides the best protection for indoor $PM_{2.5}$, ions, and trace metals.

Indoor air pollutants during smoke episodes

- Indoor PAHs were increased by up to 10-fold when comparing non-smoke and smoke episodes.
- Indoor concentrations of ions (e.g., nitrate or ammonium) and trace metals (e.g., iron or potassium) doubled during smoke episodes.
- Indoor CO_2 and NO_2 were generally not elevated during smoke episodes but were associated with indoor combustion sources.
- CO was elevated in one study during a smoke episode, with concentrations below health-based exposure limits.
- Median indoor $PM_{2.5}$ levels were consistently lower than outdoor levels (31.5 vs. 37.4 µg/m³) during smoke episodes, and higher than indoor background levels (31.5 vs. 4.7 µg/m³).
- Infiltration of smoke $PM_{2.5}$ was substantial, with a median indoor/outdoor (I/O) ratio of 0.75 (range 0.31 to 1.3) and a median infiltration factor (F_{inf}) of 0.45 (range 0.2 to 0.7).

Influence of building characteristics

- No studies were identified that examined building envelope tightness and non-PM pollutants.
- Tighter building envelopes reduced $PM_{2.5}$ infiltration by 68% vs. 31% in leakier envelopes.
- Post-1975 buildings had lower $PM_{2.5}$ infiltration (F_{inf} =0.27) than older buildings (F_{inf} = 0.41).

The effect of keeping windows closed without air filtration

- When comparing natural ventilation (i.e., windows open) with restricted ventilation (i.e., windows closed with air conditioning running), the I/O ratios were reduced for ions (1.00 to 0.71, respectively) and trace metals (0.79 to 0.42, respectively).
- Closing windows reduced $PM_{2.5}$ I/O ratios from 0.76 to 0.62.

Additional interventions

- PACs reduced median $PM_{2.5}$ by 71% (29.2 vs. 8.5 µg/m³) in closed environments, lowering concentrations by 20–60 µg/m³. Median effectiveness dropped to 37% if windows were open.
- Compared with keeping windows open, running a portable air cleaner (PAC) with windows closed reduced indoor ions by 74% and trace metals by 86%.
- High-efficiency HVAC systems and fan filter units with MERV 13 filters reduced indoor PM concentrations by 40–95%, depending on building type, PM size, and outdoor smoke levels.
- Evidence on exhaust fans was limited, and their effects on $PM_{2.5}$ infiltration remain unclear.

Introduction

Climate change is increasing the frequency, severity, and duration of wildfires worldwide. The smoke plumes created by these events can have substantial impacts on the air quality in nearby communities as well as those much further away, presenting a growing public health concern in Canada and elsewhere. Wildfires release particulate matter (PM) and other non-PM pollutants that are linked to serious respiratory and cardiovascular health effects. While attention is often focused on outdoor air quality, most people spend most of their time indoors, raising questions about the level of exposure to smoke inside buildings. Exposure to smoke indoors depends on the outdoor pollution levels and on individual building construction, maintenance, and use. Currently, in the absence of an extreme heat event, Health Canada recommends staying inside during wildfire smoke events, and protecting indoor air by closing windows and doors and using portable air cleaners (PACs) or air conditioning devices on recirculate mode.[1] Understanding how outdoor smoke enters indoor spaces is critical for developing strategies to reduce exposure and protect health.

The aim of this review is to synthesize available evidence on how outdoor smoke affects indoor air quality, with a focus on non-PM pollutants, factors that affect infiltration, and strategies for protecting indoor air quality during smoke events. The review examines findings across a range of smoke sources, including wildfires, residential wood burning, and other biomass burning. It then summarizes the available evidence on common patterns and differences in air pollutant infiltration, mitigation measures, and building characteristics.

Definitions

- **Air filtration and air cleaning:** Passing air through any technology designed to reduce pollutant concentrations, including filters, electrostatic precipitators, sorbent materials (i.e., activated charcoal) and additive technologies (e.g., ionizers, plasma, ozone generators).
- **Building envelope:** The outer elements of a building that separate the outdoor and indoor environments, including walls, windows, doors, roofs, and floors in contact with the ground.
- **Smoke episode**: Episodes of air pollution caused by combustion events such as wildland fires, coal mine fires, interface fires, agricultural fires, prescribed burns, industrial fires, landfill fires, tire fires, multi-day structural or urban fires, and residential wood combustion. This does not include air pollution generated from fossil fuel combustion (e.g., traffic-related air pollution or industrial processes).
- **Infiltration:** Direct entry of outdoor air into indoor conditioned spaces through openings in walls (including open windows and doors), ceilings, and floors.

Methodology

Literature search

A systematic literature search was conducted using variants and Boolean operator combinations of key search terms such as “wildfire,” “smoke,” “combustion,” “prescribed burn,” “biomass burning,” “infiltration,” “inside,” “concentrations,” and “indoor air quality” (a full list of search terms is available upon request). The following databases were searched: MEDLINE, Embase, CINAHL, and Environment Complete. Bibliographies and citations of key articles were used to retrieve additional literature via forward and backward chaining, along with supplemental searches as necessary. English documents published over the previous 20 years (2004–2025) were considered.

Table 1 provides a summary of the inclusion and exclusion criteria for studies. The titles and abstracts of results were screened by one reviewer. The full texts of results included in title and abstract screening were retrieved and screened by a single reviewer. Continuous artificial intelligence (AI) reprioritization was used to sort results during both levels of screening (DistillerSR), but all references were screened manually. Exclusions included sources published ahead of print, before peer review, and mathematical modelling studies that exclusively used estimated data.

The original objective of this review was to examine indoor infiltration of non-PM pollutants during smoke episodes because there is very limited evidence on this topic compared with indoor infiltration of PM. As such, the search strategy was designed to identify studies that measured both indoor and outdoor concentrations of non-PM pollutants during smoke episodes. However, most studies that measured non-PM pollutants also measured PM, and these results were also considered in the review. Forward and backward chaining of key studies identified additional results providing evidence on pollutant infiltration indoors and the influence of interventions (air cleaners, window closing) or building characteristics on indoor concentrations. Many of the studies identified through forward and backward chaining measured only indoor and outdoor PM, and those with relevant information were included; however, a systematic search for PM-specific studies was not conducted for this review because it was beyond the scope of the primary research question.

The quality of included evidence was evaluated using critical appraisal tools, as indicated by the study design below. Completed quality assessments for each included study are available on request.

Study Design	Critical Appraisal Tool
Randomized Controlled Trial	JBI Checklist for Randomized Controlled Trials
Quasi-experimental	JBI Checklist for Quasi-Experimental Studies
Analytical Cross Sectional	JBI Checklist for Analytical Cross Sectional Studies

Table 1. Inclusion and exclusion criteria

	Inclusion	Exclusion
Population	General population exposed in residential environments (both indoor and outdoor), clean air spaces, public/institutional buildings (e.g., schools, daycares, malls, libraries, community centres, healthcare centres, long-term care centres, etc.), vehicles	Population exposed in industrial workplaces
Context	During *combustion*-derived air pollution episodes that may be caused by fire (wildland, coal mine fires, peat fires, interface fires, landscape fires, agricultural fires, prescribed burns, industrial fires, landfill fires, tire fires, any multi-day structural fires (e.g., 9/11), and residential wood combustion (i.e., wood stoves), including studies that use source attribution methods)	During non-combustion derived episodes, air pollution events exclusively due to engine exhaust, traffic pollution, coal combustion, etc. (not excluding coal mine fires)
Exposure	Staying indoors, closing windows and doors	
Comparator	Indoor vs. outdoor; no comparator, during combustion event vs. not during combustion event	Gas burning appliances indoors
Outcomes	Concentration of pollutants	
Study designs	Exposure studies (including randomized and non-randomized), before-and-after studies, case studies, laboratory studies, grey literature (incl. technical reports from standard setting bodies, NRC/NIST, etc.)	Modelling studies
Publication date	Published in last 20 years	

When reported in the included studies, data relevant to the research question, including study design, setting, location, population characteristics, interventions and outcomes, were extracted by one reviewer. The results were synthesized narratively due to the variation in methodology and outcomes for the included studies. The study protocol was registered in PROSPERO (CRD42024594865).

Background

The association between non-occupational wildfire smoke exposure and adverse health effects such as increased risk of cardiovascular and respiratory disease has been systematically reviewed and is well understood. However, most wildfire smoke studies have focused on PM mass concentrations and PM composition, not co-occurring pollutants.[2,3] Moreover, differential risks of cardiovascular disease, depending on wildfire fuel type, have been documented,[4] indicating the importance of understanding which types of pollutants can infiltrate indoors during a smoke episode and at what concentrations.

Combustion-derived air pollutants (i.e., smoke)

Smoke from combustion is a complex mixture of particles and gases, including large quantities of fine particulate matter or $PM_{2.5}$. Smoke also contains gases such as carbon monoxide (CO) and nitrogen oxides (NO_x), metals such as aluminum, iron, and manganese, and organic pollutants such as dioxins, furans, volatile organic compounds (VOCs), and polycyclic aromatic hydrocarbons (PAHs). The concentrations of these pollutants vary depending on the fuels combusted and the atmospheric conditions during the combustion event.

Wildfire smoke can differ from smoke derived from other types of combustion. For example, increased fractional concentrations of magnesium, phosphorus, and potassium, and increased concentrations of CO and nitric oxide (NO) gases have been measured in wildfire smoke. In comparison, traffic-related air pollution is typically characterized by increased concentrations of heavy metals such as nickel, iron, zinc, and manganese, as well as CO, sulfur oxides (SO_x), NO_x, and ozone (O_3).[5] Wildfire smoke often has low NO_x levels but contains increased levels of volatile organic compounds (VOCs) such as benzene, acrolein, and formaldehyde, and PAHs such as acenaphthylene, fluorene, and phenanthrene.[6,7] Furthermore, the median PM particle diameter in wildfire smoke plumes can be smaller than typical urban ambient air pollution. [8]

The mixture of air pollutants in wildfire smoke can change significantly as it ages in the atmosphere or when anthropogenic fuels are burning. For instance, VOCs undergo photochemical reactions leading to the formation of ozone and secondary organic aerosols that can condense on existing particles or form new particles depending on conditions and chemical speciation.[9] PAHs also dynamically undergo phase partitioning between PM-bound and the vapour or gas phase during atmospheric aging, with recent studies indicating a decrease in PM-bound PAHs in aged smoke either by evaporation into the gas phase or by PAH decomposition.[9,10] The dynamic interaction between the particles and gases in smoke has implications for infiltration into the indoor environment, especially with filtration devices such as PACs or heating, ventilation, and air conditioning (HVAC) systems and air conditioning units predominantly focusing on filtering smoke particles in the $PM_{2.5}$ fraction. It is poorly understood how particles of different sizes and non-PM pollutants infiltrate indoors during smoke episodes, and how different interventions influence indoor pollutant concentrations.

Infiltration of air pollutants into buildings

Although modern buildings are designed to limit the infiltration of outdoor air or exfiltration of indoor air for heating and cooling efficiency, even the most advanced buildings are not airtight. Infiltration of air pollution is typically measured by two values. The first is the ratio between indoor and outdoor pollutant concentrations (I/O ratio), with values above 1 indicating air pollutants are higher indoors than outdoors. Indoor sources of pollutants can include cooking, burning candles, or using wood-burning appliances. Another measure of air infiltration is the infiltration factor (F_{inf}), which is similar to the I/O ratio, but removes indoor pollutant spikes that do not correspond to an outdoor spike. F_{inf} is a better indicator of the influence of outdoor pollution on indoor air quality.

Outdoor air can enter a building through openings in the envelope, such as around poorly sealed doors and windows, through holes drilled for lighting, fixtures, or communications cables, or through passive air vents, chimneys, or fireplaces. Buildings that are at negative pressure (e.g., interior pressure is lower than exterior air pressure), can cause air to be drawn inside. Depending on the building type, both filtered and unfiltered outdoor air can also be deliberately drawn into a building via air intakes for HVAC systems and air conditioning units. Pollutants can be introduced to indoor air unintentionally when seals around these units have broken down or when air filters are overloaded or installed improperly.[11] Moreover, some HVAC systems cannot accommodate high-efficiency filters and may introduce outdoor air pollutants indoors directly. For example, minimum efficiency reporting value (MERV) 13 filters block at least 50% of particles in the 0.3–1.0 µm size range as air passes through the filter.[12] Lower rated filters are not as effective at filtering smaller particles, which predominate in wildfire smoke. Where feasible, upgrading HVAC systems to use higher MERV rated filters, and maintaining positive air pressure indoors, can prevent or slow the entry of air pollutants into a building.[13]

Building occupants can also influence outdoor air infiltration. During clean air periods, indoor pollutants are removed either passively or actively. Passive air exchange occurs through open windows and doors, or leakage through other openings in the building envelope. Active or mechanical air exchange occurs through HVAC systems and exhaust fans in kitchens, bathrooms, or other locations. Active ventilation, (i.e., the use of externally vented exhaust fans or appliances) can cause negative pressure to develop within a building. When negative pressure builds indoors, the infiltration of outdoor air can increase due to the increased pressure differential between the indoor and outdoor environments.[14]

Additional building features can reduce the infiltration of outdoor air pollutants. These include using things like door sweeps, weather stripping, or functional dampers. The full range of considerations for preparing a structure for smoke episodes is outside the scope of this review. For more information on preparing buildings, see the following resources:

- Guideline 44-2024 - Protecting building occupants from smoke during wildfire and prescribed burn events (ASHRAE, 2024)
- A public health companion for ASHRAE Guideline 44: Protecting building occupants from smoke during wildfire and prescribed burn events (NCCEH, 2025)

Research questions

This review synthesizes current evidence on the impacts of outdoor smoke on indoor air quality, with particular attention to non-PM pollutants, infiltration factors, and measures to protect indoor air during smoke events. Specifically, this review sought to answer the following questions:

1. What is the relationship between outdoor and indoor air pollutant concentrations during smoke episodes?
2. What is the impact of closing windows and doors on indoor air pollutant concentrations during smoke episodes?
3. What is the impact of using portable air cleaners (PACs) on indoor air pollutant concentrations during smoke episodes?
4. How do building features impact the relationship between indoor and outdoor air pollutant concentrations during smoke episodes?

Results

Overview of studies

This review identified a total of 41 studies of moderate to high quality, including 23 quasi-experimental studies, 12 analytical cross-sectional studies, four randomized controlled trials, and two cohort studies (Appendix A). These studies covered a range of smoke episodes caused by combustion events. Sixteen were specific to wildfire smoke episodes or seasons, seven examined smoke from residential wood burning, two examined both wildfire and residential wood smoke, five examined episodes of haze (biomass, wildfire and urban air pollution), two covered biomass burning (indoor biomass cooking, and agricultural biomass burning), and one study was conducted during a landfill tire fire. The majority of studies were carried out in North America, with the remainder of the studies performed in Australia, Singapore, Brazil, and Spain.

The 41 included studies examined a range of pollutants:

- PAHs (n = 3; 7%)
- Trace metals, ions, and minerals (n = 3; 7%)
- Black carbon (BC) (n = 3; 7%)
- Inorganic gases (n = 4; 10%)
- Levoglucosan, as a marker of biomass burning (n = 4; 10%)
- PM number concentrations (n = 6; 15%)
- $PM_{2.5}$ mass concentrations (n = 31; 76%)

The sampling period for measurements ranged from 1.5 hours to 9 years, contributing to variability in reported concentrations. The settings for pollutant measurements in these studies also ranged from residential detached homes (n = 21; 51%) to apartment buildings (n = 5; 12%), and 12 studies (29%) examined air pollutants in public buildings such as schools, university buildings, a public library, assisted living or skilled nursing facilities, churches, fitness centres, childcare centres, museums, community centres, a fire station, and a homeless shelter. When reported, most buildings in these studies were built within the past 50 years.

Many of the included studies examined the effect of an intervention on reducing levels of indoor air pollutants during smoke episodes. When considering window use, nine studies reported windows to be closed, and 11 studies reported windows open at least some of the time, or window status was not reported. In total, 14 studies examined the use of PACs, nine examined the role of HVAC systems in reducing indoor air pollutant concentrations, and two examined both.

What is the relationship between outdoor and indoor air pollutant concentrations during smoke episodes?

Several different patterns of indoor-outdoor air pollutant relationships emerged across the studies in this review (Appendix B). Observations are reported below by pollutant type.

Non-PM pollutants

PAHs

Three studies examined PAH concentrations in the context of a wildfire smoke episode, agricultural burning, and a landfill tire fire. In Ghetu et al., comparing wildfire and non-wildfire periods revealed that indoor vapour-phase low molecular weight (LMW) and high molecular weight (HMW) PAH concentrations increased by three and six times, respectively, during wildfire periods.[15] Furthermore, indoor HMW PAH concentrations were generally higher than outdoor concentrations, except when the average air quality index for particulate matter exceeded 115 (unhealthy for sensitive groups).[16]

Cristale et al. examined the sugarcane burning season in Brazil, reporting that indoor PAH concentrations increased from 2.35 to 22.9 ng/m^3 when comparing against the non-burning season in one residential home.[17] The most abundant PAH compounds measured indoors during the burning period were phenanthrene, fluoranthene, pyrene, benzo(e)pyrene, and benzo(g,h,i)perylene. However, building materials and techniques relevant to the local climate in Brazil may result in different indoor concentrations when comparing with North American structures.

Another study by Artíñano et al. on landfill tire fire pollutants reported indoor PAH levels in deposited dust samples did not increase between fire and non-fire periods.[18] However, outdoor concentrations were elevated during the fire period relative to a reference sampling site 6 km away, with the top two most abundant PAHs measured being fluoranthene and pyrene at 27 and 64 ng/m^2, respectively. At the reference site, outdoor concentrations of 6.5 and 24 ng/m^2 were reported for fluoranthene and pyrene, respectively. Further research is needed to clarify which PAHs are more prevalent during wildfires, and which are most able to infiltrate into the indoor environment. This would clarify the potential types of health risks, because some PAHs are carcinogens.[15]

Trace metals, ions, and minerals

Three studies reported on concentrations of trace metals, ions, and minerals indoors and outdoors during smoke episodes—two about haze and one about a landfill fire. Metals and other elements that are found at low concentrations attached to particles include iron, potassium, and zinc. Ions include species such as nitrate, oxalate, or ammonium. During one haze episode in Singapore, Tran et al. reported that total ions and water-soluble trace metals in $PM_{2.5}$ increased from 8.77 to 14.98 µg/m^3 and 0.75 to 1.53 µg/m^3

indoors, respectively, when compared with non-hazy periods in a naturally vented (windows open) apartment.[19] Associated I/O ratios of 1.00 and 0.79 were reported for ions and trace metals, respectively. By mass, the most abundant indoor ions during hazy periods were sulfate and ammonium.

With respect to health risks, inhalational exposure to sulfate ions at the levels reported by Tran et al. (7482.15 ± 85.92 ng/m^3) during the 2019 haze episode represent a fraction of the daily ingestion exposure from drinking water estimated by Health Canada.[20] Similarly for ammonium, the increased levels in indoor air (2875.5 ± 7.1 ng/m^3) are a fraction of the concentrations estimated by Health Canada in typical outdoor urban air (25,000 ng/m^3).[21] Trace metals, potassium, tin, and zinc were the most abundant by mass indoors during haze episodes. In another study examining a haze episode in 2015, Sharma et al. performed carcinogenic and non-carcinogenic health risk estimates based on indoor measured levels of trace metals. They reported elevated lifetime cancer risk values of 5.67×10^{-6} for naturally vented conditions and 1.10×10^{-6} for windows closed with a PAC running.[22] Both reported values exceed the limit of 1×10^{-6} or one person in a population of one million developing cancer in their lifetime, representing a potential cancer risk. Non-carcinogenic health risk estimates were not elevated in the study.

In addition to haze studies, analysis of samples taken during a landfill tire fire in Artíñano et al. found indoor concentrations of total minerals in deposited dust samples were lower than outdoor concentrations (3075 vs. 5639 µg/m^2), with calcium oxide reported as the most abundant (1375 µg/m^2) inorganic species measured indoors.[18] Note that the concentration of calcium oxide would be further decreased if airborne and, for comparison, the current exposure limit for Ontario workplaces is set at 2 mg/m^3 over an eight-hour shift.[23]

Inorganic gases

A total of four studies examined several common inorganic gases that can be produced by combustion processes, including carbon monoxide (CO), carbon dioxide (CO_2), and nitrogen dioxide (NO_2). Three of the studies were focused on wildfire smoke, and one on indoor biomass burning appliances. The available evidence indicates that smoke episodes outdoors have little impact on indoor levels of these gases, which can have significant indoor sources. In Kaduwela et al., no differences were reported for CO_2 when comparing wildfire and non-wildfire periods in a high school, and there were no increases in local ambient outdoor concentrations CO_2.[24] In Lee et al., CO_2 was increased on wildfire smoke days (809.2 ppm) compared with non-wildfire days (689 ppm) in licensed childcare care facilities in British Columbia; however, this was likely due to decreased natural ventilation (i.e., closing windows) rather than wildfire smoke impacts.[25]

Shrestha et al. reported that during wildfire periods, detached houses without significant indoor sources such as gas stoves had similar concentrations of NO_2 indoors and outdoors, and were not impacted by wildfire plumes.[26] Concentrations of indoor CO were elevated compared with outdoor concentrations in this study (0.69 vs. 0.20 ppm). However, 26 of the 28 homes contained gas water heaters with standing pilot lights, and the study concluded these were the major source of CO, even after filtering the data for indoor source spikes. This study also reported up to a three-fold increase in indoor CO levels during wildfire plume periods, but concentrations remained well below the 1-hour 25 ppm Health Canada limit for residential exposure. Lastly, this study indicated that proximity to major roadways influenced indoor and outdoor NO_2 concentrations. Similarly, in Weaver et al., indoor CO levels were predominantly impacted in homes actively cooking with a biomass stove (0.002 vs. 11.2 ppm), not neighbouring homes (0.001 vs. 0.2 ppm) when comparing against baseline periods.[27]

Levoglucosan

Levoglucosan is a chemically stable organic compound that is released in large quantities during the combustion of wood or biomass and can be used a marker of wood or wildfire smoke. Four studies in this review examined levoglucosan concentrations in the context of residential wood combustion; however, due to methodological differences in detection and sampling, absolute values greatly varied (range 0.034, 613 ng/m^3) between studies. When considering relative concentrations, both Wheeler et al. and Allen et al. reported lower concentrations of levoglucosan indoors compared with outdoors, and an I/O ratio of 0.36 was reported by Wheeler et al.[28,29]

PM

The review included 20 studies that measured $PM_{2.5}$ concentrations during smoke episodes or seasons. The median (range) outdoor $PM_{2.5}$ mass concentration was 37.4 (3.9–157.0) µg/m^3 (Appendix B). The sampling periods for both indoor and outdoor measurements ranged from 3 hours to 334 days, and averaging periods consisted of 1 hour (n = 7), 24 hours (n = 9), 48 hours (n = 1), 7 days (n = 2), and 11 days (n = 1). For context, the Canadian Ambient Air Quality Standards for $PM_{2.5}$ 24-hour average is currently 27 µg/m^3 and will be reduced to 23 µg/m^3 in 2030.[30] For the 13 studies specific to wildfire smoke episodes or seasons, the median outdoor value was 32.9 (5.0–127.0) µg/m^3. Although only four of these wildfire-specific studies reported average outdoor $PM_{2.5}$ concentrations during non-wildfire periods, the values were notably lower with a median of 4.6 (2.0–9.1) µg/m^3.

Indoor $PM_{2.5}$ concentrations were lower than outdoor values across all 20 studies. The indoor $PM_{2.5}$ median (range) value was 31.5 (5.2–98.0) µg/m^3. These values represent non-intervention or baseline conditions during a smoke episode. For the 13 studies specific to wildfire smoke episodes or seasons the median was 27.8 (5.2–92.0) µg/m^3. Once again, the four studies with values during non-wildfire periods reported a much lower median concentration of 4.7 (1.3–7.5) µg/m^3.

Indoor/outdoor (I/O) ratios were calculated for 12 of the 20 studies, with a median (range) of 0.74 (0.31–1.3) reported during smoke episodes or seasons. For the seven wildfire-specific studies, the median was 0.67 (0.31–0.93). High I/O ratios may not always be due to increased infiltration of outdoor air pollutants because this value can be affected by indoor sources of such as cooking, vacuuming, smoking, or lighting candles or incense. Overall, however, the findings from these studies indicate that indoor air was influenced by outdoor air during smoke episodes.

To more accurately estimate the level of outdoor $PM_{2.5}$ infiltrating indoors, seven studies used censoring algorithms to remove data where indoor air $PM_{2.5}$ peaks did not correspond to outdoor $PM_{2.5}$ peaks. The resulting ratio, or infiltration factor (F_{inf}) was calculated across nine studies for $PM_{2.5}$ during smoke episodes or seasons, with a median (range) of 0.45 (0.20–0.70). Seven of these studies were specific to wildfire smoke episodes or seasons with a median F_{inf} value of 0.58 (0.27–0.70). When considering heavy wildfire smoke events, where outdoor $PM_{2.5}$ levels exceeded the CAAQS $PM_{2.5}$ 24-hour average value of 27 $\mu g/m^3$, F_{inf} values decreased moderately to 0.44 (0.27–0.59) across five studies. This may be due to the increased outdoor component in the F_{inf} calculation, and people implementing interventions (i.e., closing windows or using PACS) as the outdoor air quality worsens. For comparison, three of these five studies calculated F_{inf} values for non-wildfire periods and reported mean values of 0.39, 0.29, and 0.45.[31-33]

In addition to $PM_{2.5}$ mass concentrations, four of the 20 studies examined PM number concentrations for particles ranging in size from 0.3 to 10 µm. Artinano et al. measured PM_1 indoors and outdoors in a public school near a landfill tire fire and found that particle counts were an order of magnitude lower indoors compared with outdoors (3.9×10^4 vs. 3.8×10^5 #/cm^3).[18] Three of the four studies examined PM number concentrations during wildfire smoke events or seasons and reported lower particle counts indoors compared with outdoors. For example, Dev et al. reported $PM_{0.3\text{-}10}$ number concentration I/O values of 0.37, 0.14, and 0.44 for a public building and two houses, respectively, and Shrestha et al. reported a mean $PM_{0.5\text{-}2.5}$ F_{inf} of 0.4 for 28 residential detached homes.[26,34]

What is the impact of closing windows and doors on indoor air pollution concentrations during smoke episodes?

Closing windows and doors during combustion-derived air pollution episodes is often recommended as a strategy to reduce smoke exposure indoors. This review identified 11 studies that indicated whether windows and doors were closed during the study period (Table 2). Overall, indoor concentrations of air pollutants were consistently lower when windows were closed during smoke events, as indicated by lower I/O and F_{inf} values described in the following paragraphs. Furthermore, indoor concentrations were substantially reduced when windows were closed and PACs were used. (See next section for more details).

In six studies when the windows were reported to be closed, the median I/O ratio for $PM_{2.5}$ was 0.62. In seven studies when windows were open at least some of the time or window status was not reported, the median I/O ratio was 0.76. Along with reduced I/O ratios, there was a median 32% reduction in $PM_{2.5}$ concentrations indoors compared with outdoors when windows were closed in nine reporting studies.

Comparably, the median reduction in $PM_{2.5}$ concentrations was 23% for the 11 studies with windows open at least some of the time or window status was not reported. The median outdoor concentrations were similar for both sets of studies, at 42.17 vs. 45.4 µg/m^3, respectively. Variability in data collection and reporting for other air pollutants precluded a similar overview analysis.

Several studies included windows open or closed in their study design and statistical modelling. During a heavy smoke haze, Tran et al. examined the difference between having windows open in an high-rise apartment compared with having the windows closed with an air conditioner unit equipped with MERV 7 filters running.[19] The study found that all I/O ratios were reduced when windows were closed and the air conditioner was running compared with natural ventilation, though no statistical significance was reported. I/O ratios for $PM_{2.5}$ were reduced from 0.98 to 0.72, black carbon from 0.97 to 0.60, ions from 1.00 to 0.71, and trace metals from 0.79 to 0.42. For $PM_{2.5}$ specifically, this equated to a 45% decrease in indoor concentrations from 48.93 vs. 26.80 µg/m^3. In a similar design, Sharma et al. measured the difference between having windows open and closed in a different high-rise apartment during a haze episode, and reported indoor $PM_{2.5}$ concentrations were reduced from 98 to 71 µg/m^3 and I/O ratios were reduced from 0.76 to 0.62.[22] Once again, the statistical significance of these reductions was not reported.

In the case of wildfire specific studies, Shrestha et al. reported that $PM_{0.5\text{-}2.5}$ number concentration and black carbon F_{inf} values were significantly increased by 45% and 67%, respectively, when windows were open in residential detached homes for more than 12 hours of the sampling period.[26] This increase in F_{inf} also corresponded to a 33% increase in indoor $PM_{0.5\text{-}2.5}$ number concentration and 237% increase in black carbon. Similarly, Walker et al. reported that leaving windows/doors open increased indoor concentrations of $PM_{2.5}$ on wildfire days by 11.4% per every 10% increase in days with windows/doors open (95% CI: 6.4, 16.7) across 20 residential detached homes in Montana.[31] Although four studies tracked the amount of time windows were open or closed during the study period,[15,35-37] none used this information as a covariate in the analysis of indoor air pollutant concentrations.

Finally, there was little information on the effect of leaving exterior doors open or closed during smoke episodes. Holder et al. examined door opening across 13 buildings, reporting open doors led to increased infiltration rates and/or indoor $PM_{2.5}$ concentrations at seven locations, decreased values at four locations, and had no impact at two locations.[38] The mixed results were somewhat explained by 11/13 locations maintaining positive pressure indoors, reducing infiltration of outdoor air. Maintaining positive pressure inside buildings to reduce infiltration of outdoor air during smoke episodes is recommended in ASHRAE Guideline 44.[39]

Table 2. Building characteristics and the effect of having windows/doors closed on indoor air pollution concentrations during smoke episodes. All abbreviations are defined in a footnote below the table.

Reference	Setting	Building characteristics	Window status	Effect (I/O ratio or F_{inf})
Studies where windows closed for study period				
Willis et al. 2023[40]	Wildfire; Butte, Montana, USA	- Area (sq. ft.): Homeless shelter: 992; senior assisted living: 3680; school building: 14,880	Windows closed	- $PM_{2.5}$ I/O reduced from 0.75–0.87 to 0.49–0.72 during wildfire smoke events comparing rooms with or without a PAC
Montrose et al. 2022[33]	Wildfire; Idaho, USA	- Skilled nursing facilities (n=4, 15,000 to 30,000 sq. ft.) - Building age (approximate years): 10, 45, 50, and 50	Windows closed	- $PM_{2.5}$ I/O ratio increased from 0.27 to 0.56 comparing non-wildfire to wildfire days - $PM_{2.5}$ F_{inf} increased from 0.29 to 0.59 comparing non-wildfire to wildfire days
Dev et al. 2021[34]	Wildfire; Fairbanks, AK, USA	- 1 university building and 2 residential detached houses - Construction year: University building: 1964, renovated 2002	Windows closed	- $PM_{2.5}$ I/O ratios were 0.37, 0.14, and 0.44 for the university building, House A (no ventilation), and House B (HVAC) respectively
May et al. 2021[41]	Wildfire; Seattle, Washington, USA	- 2 rooms in a residential detached home (Room A 200 m^3, and B 50 m^3)	Windows closed	- $PM_{2.5}$ I/O ratio reduced from 0.31–0.50 to 0.003–0.22 for PAC on vs. off during a wildfire
Wheeler et al. 2021[42]	Wildfire; Port Macquarie, NSW, Australia	- Port Macquarie Library, Media Room 22 m^2	Windows closed	- $PM_{2.5}$ F_{inf} reduced from 0.32 to 0.17 with PAC use during wildfire periods
Xiang et al. 2021[43]	Wildfire; Seattle, Washington, USA	- 7 residential detached homes (n=5 PAC, n=2no filtration) - Construction year: ≤ 1940=1, 1941 to 1960=2, 1961 to 1980=0, 1981 to 2000=2, ≥ 2001=2	Windows closed	- $PM_{2.5}$ F_{inf} reduced from 0.56 to 0.19 comparing PAC on vs. off
Stauffer et al. 2020[44]	Wildfire; Butte, Montana, USA	- 2 Montana Technological University offices (12.2 m^2)	Windows closed	- $PM_{2.5}$ I/O ratio was 0.60 for PM2.5 when the PAC was turned off

Reference	Setting	Building characteristics	Window status	Effect (I/O ratio or F_{inf})
Henderson et al. 2005[44]	Wildfire; Colorado, USA	- 4 residential home pairs (1 with PACs, 1 without per wildfire event, n=4) - Area (m^3), windows (#), age (years) **Polhemus burn:** House 1 (PAC): 407, 17, 5; House 2: 815, 27, 7 **Snaking wildfire:** House 1 (PAC): 453, 7, 3; House 2: 424, 13, 30 **Schnoover wildfire:** House 1 (PAC): 1415, 37, 6; House 2: 1130, 28, 5 **Hayman wildfire:** House 1 (PAC): 510, 10, 39; House 2: 396, 21, 39	Windows closed	- $PM_{2.5}$ F_{inf} reduced from 0.58–0.93 to 0.06–0.34 comparing PACs vs. no PAC
Tham et al. 2021[45]	Haze; Singapore	- School building, two classrooms (313 m^3), same location one floor apart	Windows closed	- PM I/O ratios reduced from 0.47–0.50 to 0.05–0.07 comparing FFU to control - PM F_{inf} reduced from 0.2–0.3 to 0.06–0.11
Cao et al. 2016[46]	Haze; Singapore	- 5 neighboring classrooms (60 m^2), Nanyang Technological University	Rooms have no windows	- PM I/O ratios for all sizes reduced from 0.2–0.62 to 0.06–0.24 comparing MERV 13 or 14 filters with control
Chen et al. 2016[47]	Haze; Singapore	- Staff office (300 m^2), Nanyang Technological University	Windows closed	- $PM_{0.3}$ particle concentration I/O ratio reduced from 0.64 to 0.0.59 comparing AC on vs. off
		Studies with window conditions or analysis		
Holder et al. 2025[38]	Wildfire; Missoula, Montana, USA	- Church (n = 2), university (n = 2), office (n = 6), hotel (n = 1), fire station (n = 1), museum (n = 2), fitness centre (n = 7), childcare centre (n = 4), community centre (n = 3) - Area range: 6600 to 180,000 sq. ft. - Year range: 1892–2020	Window type	- $PM_{2.5}$ I/O ratios were significantly increased in buildings with inoperable or sliding windows compared with hinged windows

Reference	Setting	Building characteristics	Window status	Effect (I/O ratio or F_{inf})
Ghetu et al. 2022[15]	Wildfire; Washington, Oregon, California, and Idaho, USA	- Residential detached homes (n=15) - Age range: 1972–2015	Windows open average (days): Before wildfires: 10 (range 0–44); During wildfires: 2 (range 0–6)	- Vapor-phase LMW and HMW PAH were three and six times higher indoors during wildfires (not significant)
He et al. 2022[37]	Wildfire; Seattle, Washington, USA	- Residential detached homes (n = 4; 1,500 to 3,500 000 sq. ft.), apartment (n = 1, 800 000 sq. ft.), office (n = 2, 135 to 144 sq. ft.)	Window opening (#) Never: 2 homes, 2 offices; Sometimes: 2 homes; Always: 1 apartment	- $PM_{2.5}$ I/O ratio reduced from 0.82 to 0.43 comparing homes with filtration technologies vs. homes without
Reisen et al. 2019[48]	Wildfire and RWC; Yarra Valley and Gippsland, Victoria, Australia	- Residential detached homes: prescribed burns (n = 4), and RWC (n = 3) - Building age range (years): 8 to 98	Windows open: 4 homes Windows closed: 3 homes	- $PM_{2.5}$ I/O ranged from 0.13 to 2.9 - $PM_{2.5}$ F_{inf} was 0.58 and 0.3 for prescribed burns and RWC impacted houses, respectively
Shrestha et al. 2019[26]	Wildfire; Denver, Colorado, USA	- Residential detached homes (n = 28) - Building type (homes): Built green: 5; EER: 13; Non-EER 10	Windows open hours (homes): 0=3, 1–6=4, 7–12=5, >12=16	- $PN_{0.5–2.5}$ and BC F_{inf} were 0.4 and 0.52 during smoke plume periods (PM density ≤ 27 µg/m^3) - $PN_{0.5–2.5}$ and BC F_{inf} were significantly higher by 45% (0.46 vs. 0.84) and 67% (0.28 vs. 0.84), respectively, when windows were open for ≥12 hours
Barn et al. 2008[36]	Wildfire and RWC; British Columbia, Canada	- RWS: 21 residential detached homes - Wildfire smoke: 17 residential detached homes - Area (square feet): 640–4330 - Windows range (#): 4–21 - Building age (years): 5–60	Windows open: Never: RWS 12, Wildfire 0 Sometimes: RWS 2, Wildfire 9 Always: RWS 2, Wildfire 4	- $PM_{2.5}$ F_{inf} reduced from 0.61 to 0.19 (wildfire) and 0.28 to 0.10 (RWC) comparing PAC on vs. off - Increasing window number associated with increased infiltration ($p < 0.0001$)

Reference	Setting	Building characteristics	Window status	Effect (I/O ratio or F_{inf})
Tran et al. 2021[19]	Haze; Singapore	- Bedroom in a residential apartment (10.5 m^2), 13th of 25 stories, natural ventilation	Condition: NV = Windows kept open, AC = Windows closed AC on, PAC = Windows closed PAC on	- I/O ratios reduced for $PM_{2.5}$ (0.98 to 0.72 to 0.25), BC (0.97 to 0.60 to 0.15), Ions (1.00 to 0.71 to 0.21), and WSTE (0.79 to 0.42 to 0.07) for NV, AC, and PAC conditions, respectively
Sharma et al. 2017[22]	Haze; Singapore	- Living room in a residential apartment, 20th floor, natural ventilation	Windows open and closed conditions	- $PM_{2.5}$ I/O ratio reduced from 0.76 to 0.62 to 0.32 comparing windows open to windows closed to PAC on with windows closed
Kajbafzadeh et al. 2015[35]	RWC; Vancouver, British Columbia, Canada	- Residential detached homes (n=20)	Time open windows mean ± SD (%): 15.2 ± 31.3	- $PM_{2.5}$ I/O ratio reduced from 1.3 to 0.87 comparing PAC use with placebo
Artinano et al. 2017[18]	Landfill fire; Seseña, Toledo, Spain	- School building	Windows closed, windows open for 30 minutes (test condition)	- BC maximum with windows open vs. closed (1.2 vs. 1.8 μg/m^3)

*Abbreviations: **AC:** air-conditioning, **ACMV**: air-conditioning and mechanical ventilation, **AQI:** air quality index, **BC:** black carbon, **FFU:** fan filter unit, **HMW**: high molecular weight, **LMW:** low molecular weight, **NR:** not reported, **NV:** naturally ventilated, **OP:** oxidative potential, **$PM_{x\text{-}y}$**: particulate matter x-y μm in diameter, **PN:** particle number, **RWC:** residential wood combustion, **WSTE:** water soluble trace elements.

What is the impact of using portable air cleaners (PACs) on indoor air pollutant concentrations during smoke episodes?

PACs are a commonly recommended intervention for reducing concentrations of combustion-derived air pollutants indoors. The NCCEH recently completed a systematic review of the effectiveness of air filtration and air cleaning during smoke episodes, and reported an average reduction of 56% (range 5.3–99%) for indoor $PM_{2.5}$ mass concentrations across 17 studies.[49]

To examine the effect of using PACs with windows and doors closed in homes or other buildings, 14 studies that included measurements of both indoor and outdoor air pollutants were included in this review. The PACs evaluated used several different technologies such as HEPA filters (two studies), HEPA filters combined with activated charcoal filters (seven studies), MERV 13 filters (four studies), and an electrostatic precipitator combined with an activated charcoal filter (one study) (Table 3). Of the seven studies that reported having closed windows during study periods, indoor $PM_{2.5}$ mass concentrations were reduced from a median (range) of 29.2 (5.2–98) µg/m^3 in the control or baseline smoke condition to 8.51 (0.4-30.7) µg/m^3 in PAC condition, or a 71% (18–99%) reduction. For comparison, an overall median indoor $PM_{2.5}$ concentration of 32.8 µg/m^3 was reported across 17 studies in this review, regardless of window status. When considering the impact of PACs on median I/O ratios for $PM_{2.5}$ mass concentrations, a decrease from a median of 0.74 (0.31–0.93) in the baseline or PAC-off condition to 0.25 (0.003–0.72) in the PAC-on condition was reported across five studies. Four studies examining PAC use indicated that windows were left open for some or all of the study period or did not report on status. In these four studies, PACs were found to be less effective, with indoor $PM_{2.5}$ mass concentrations reduced by 37% (25–59%).

Two studies examined haze air pollution episodes with windows open, windows closed, and windows closed with PAC-on conditions. In Tran et al., indoor $PM_{2.5}$ mass concentrations in a single apartment were reduced from 48.9 to 26.8, when the windows were closed with an AC unit turned on and reduced again to 15.1 µg/m^3 when the PAC was turned on. The I/O ratios were reduced from 0.98 to 0.72 to 0.25 under the same conditions, respectively.[19] Similar results were reported by Sharma et al., with indoor $PM_{2.5}$ concentrations of 71, 98, 23 µg/m^3 and I/O ratios 0.76, 0.62, 0.32 reported for windows open, windows closed, and windows closed and PAC on conditions, respectively.[22] The higher indoor $PM_{2.5}$ concentrations during the windows closed condition are explained by the outdoor $PM_{2.5}$ concentrations being higher during the windows closed condition (157 ± 107 µg/m^3) than during the windows open condition (94 ± 34 µg/m^3).

For non-PM pollutants, only one study examining indoor air pollution concentrations during a haze episode reported on the impact of using a PAC. Tran et al. reported that running a PAC with HEPA and activated charcoal filter in an apartment with the windows closed reduced total ions, and trace metals by 74% and 86% (11.1 and 1.33 µg/m^3), respectively, when compared with the windows-open condition.[19]

Closing windows and turning on the AC reduced total ions and trace metals by 35% and 57% (5.3 and 0.9 μg/m^3), respectively. The I/O ratios were also reduced for ions (1.00 to 0.71 to 0.21), and trace metals (0.79 to 0.42 to 0.07) comparing windows open, AC with windows closed, and PAC-on with windows closed and AC off, respectively. Aside from Tran et al., three residential wood combustion studies examined the effect of PAC use on concentrations of the non-PM biomass marker levoglucosan, finding a reduction in concentrations ranging from 32% to 74%.[28,29,35]

Table 3. Portable air cleaners (PACs) and indoor air quality during smoke episodes. All abbreviations are defined in a footnote below the table.

Reference	Episode	Intervention, PAC details, and window condition	Outcome
Prathibha et al. 2024[50]	Wildfire and RWC	Baseline vs. DIY PAC (20" × 20" MERV 13) vs. Commercial PAC, HEPA and activated charcoal, PACs run on high for ≥ 8 hours for ≥ 33% of study days; windows closed	**DIY PAC** - Reduced $PM_{2.5}$ F_{inf} from 0.7 to 0.6 - Reduced indoor $PM_{2.5}$ 7%–10% **Commercial PAC** - Reduced $PM_{2.5}$ F_{inf} from 0.7 to 0.6 - Reduced indoor $PM_{2.5}$ 5%–20%
Wheeler et al. 2024[51]	Wildfire (prescribed burns)	HEPA cleaner model AUS-1250AZPU (Winix), HEPA and activated charcoal filters, installed in bedroom or main living room vs. PAC off; window status NR	- PACs reduced $PM_{2.5}$ 30%–75%
Willis et al. 2023[40]	Wildfire	No PAC vs. Pet 300 with H13 True HEPA 5 stage filtration filter (unbeaten), set to high, installed in one of each pair of rooms in each public building; windows closed	- PACs reduced $PM_{2.5}$ I/O ratios from 0.75–0.87 to 0.49–0.72 - PACs reduced $PM_{2.5}$ by 16%–35% (6.5–13.9 μg/m^3)
May et al. 2021[41]	Wildfire	PAC off vs. 50.8 cm × 50.8 cm MERV 13 air filter, attached to a standard home box fan with tape, installed in room A and B; windows closed	- PACs reduced $PM_{2.5}$ I/O from 0.31–0.50 to 0.003–0.22 - PACs reduced indoor $PM_{2.5}$ by 56%–99% (36–39.6 μg/m^3)
Wheeler et al. 2021[42]	Wildfire	PAC off vs. Cli-Mate AP20 (Aquaport Corporation Pty Ltd.), grade H12 HEPA filter and activated charcoal filters, set to medium, installed in the media room; windows closed	- PACs reduced $PM_{2.5}$ F_{inf} from 0.32 to 0.17 - PACs reduced $PM_{2.5}$ by 72% (median 14.3 μg/m^3)
Xiang et al. 2021[43]	Wildfire	PAC off vs. Air Purifier 2000i (Philips) (Auto setting, HEPA and activated charcoal filters, set to	- PACs reduced $PM_{2.5}$ F_{inf} from 0.56 to 0.19 - PACs reduced indoor $PM_{2.5}$ by 70% (33 μg/m^3)

Reference	Episode	Intervention, PAC details, and window condition	Outcome
		auto, installed in living room; windows closed	
Stauffer et al. 2020[44]	Wildfire	Office with PAC vs. no PAC, crossover design; Filtrete Ultra Clean Air Purifier FAP02-RS (3M), MERV 13, set to high; windows closed	- $PM_{2.5}$ I/O ratio was 0.60 for when PAC off - PACs reduced $PM_{2.5}$ by 73% (8.14 µg/m^3) during daytime and 92% (6.05 µg/m^3) during nighttime
Barn et al. 2008[36]	Wildfire and RWC	PAC off vs. Honeywell filter 18150 (Honeywell), HEPA and activated charcoal filters, set to high, installed in the primary bedroom; windows open: Never: RWS 12, Wildfire 0; Sometimes: RWS 2, Wildfire 9; Always: RWS 2, Wildfire 4	- PACs reduced $PM_{2.5}$ F_{inf} from 0.61 to 0.19 (wildfire) and 0.28 to 0.10 (RWC) - PACs reduced $PM_{2.5}$ by 65% (3.3 µg/m^3, wildfire) and 55% (3.2 µg/m^3, RWC)
Henderson et al. 2005[52]	Wildfire	No PAC vs. Friedrich C90 electrostatic precipitating (ESP) cleaners (Friedrich Air Conditioning Company, activated charcoal filter; windows closed	- PACs reduced $PM_{2.5}$ I/O ratios from 0.58–0.93 to 0.06–0.34 - PACs reduced $PM_{2.5}$ by 63%–88% (4.91–29.68 µg/m^3); PACs reduced all indoor $PM_{2.5}$ levels to < 3 µg/m^3
Tran et al. 2021[19]	Haze	Open windows (NV) vs. Panasonic Air Conditioner (AC) CS-S12TKZW (Panasonic), MERV7 polypropelene one-touch filter, set to medium speed vs. City M Air Purifier (Camfil), HEPA H13 and molecular activated carbon filters, set to medium speed, installed in bedroom; windows closed	- $PM_{2.5}$ I/O ratios reduced from (0.98 to 0.72 to 0.25), BC (0.97 to 0.60 to 0.15), Ions (1.00 to 0.71 to 0.21), and WSTE (0.79 to 0.42 to 0.07) comparing windows open, AC, and PAC use - PAC reduced $PM_{2.5}$, BC, ions, and WSTE by 69%, 80%, 74%, and 86% (33.81, 3.50, 11.07, 1.32 µg/m^3) compared with windows open
Sharma et al. 2017[22]	Haze	Open windows vs. closed windows vs. F-PXH55A (Panasonic), HEPA filter; windows closed	- $PM_{2.5}$ I/O ratio reduced from 0.76 to 0.62 to 0.32 comparing windows open to windows closed to PAC on with windows closed - PAC reduced $PM_{2.5}$ by 48% (75 µg/m^3) compared with windows closed *(Outdoor $PM_{2.5}$ varied greatly between conditions)*
Kajbafzadeh et al. 2015[35]	RWC	Placebo (i.e., no internal filter) crossover design vs. Honeywell filter (model 50300 in main activity room, and 18150 in bedroom, HEPA and activated charcoal filters, set to	- PACs reduced $PM_{2.5}$ I/O ratio from 1.3 to 0.87 - PACs reduced indoor $PM_{2.5}$ by 48% (3.1 µg/m^3) and levoglucosan by 60% (17.5 ng/m^3)

Reference	Episode	Intervention, PAC details, and window condition	Outcome
		high; Windows: Time open mean ± SD (%): 15.2 ± 31.3	
Wheeler et al. 2014[28]	RWC	Placebo (i.e., sham internal filter used), crossover design, vs. Filtrete Ultra Clean Air Purifier FAP02-RS (3M, set to high, installed in wood burning appliance room; window status NR	- PACs reduced PM I/O ratios from 1.42 to 0.42, $PM_{2.5}$ from 1.2 to 0.63, and Levoglucosan from 0.36 to 0.17 - PAC reduced PM F_{inf} from 0.56 to 0.26 - PAC use reduced PM, $PM_{2.5}$, and Levoglucosan levels by 63%, 50%, and 32% (median 1.7, 1.95 µg/m^3, and 0.016 ng/m^3)
Allen et al. 2011[29]	RWC	Placebo (i.e., no internal filter used) crossover design vs. Honeywell HEPA filter (model 50300 in main activity room, and 18150 in bedroom, HEPA and activated charcoal filters, set to high; window status NR	- PACs reduced $PM_{2.5}$ F_{inf} from 0.34 to 0.2 - PAC reduced $PM_{2.5}$ by 59% (6.6 µg/m^3) and levoglucosan by 74% (94 ng/m^3)

Abbreviations: **AC:** air-conditioning, **BC:** black carbon, **FFU:** fan filter unit, **NR:** not reported, **NV:** naturally ventilated, **PM_{x-y}:** particulate matter x-y µm in diameter; **PN:** particle number, **RWC:** residential wood combustion, **WSTE:** water soluble trace elements.

How do building features impact the relationship between indoor and outdoor air pollutant concentrations during smoke episodes?

The amount of smoke pollution infiltrating into a building can be influenced by several factors related to the construction and mechanical systems present in the building. For instance, some buildings may have more cracks or sealant failures around doors and windows that let more air pass through the building envelope, or HVAC systems may draw outdoor pollutants indoors without adequate filtration. This effect was examined in Wheeler et al. when smoke $PM_{2.5}$ infiltration in houses with lower and higher passive air exchange rates was compared. The first house had a 10.31 air changes per hour at 50 pascals pressure (ACH50), suggesting a tighter building envelope, while the second had an ACH50 of 20.40.[51] The house with the lower ACH50 provided greater passive protection from infiltration of $PM_{2.5}$ from smoke (68% vs. 31%). This review does not provide a comprehensive examination of the influence of all building features on infiltration of outdoor air pollutants, but provides context based on the factors discussed in the studies reviewed.

Number of windows

More windows can provide more potential gaps for outdoor air pollution to enter a structure. Walker et al. reported that residential detached homes (n = 14) and apartments (n = 2) with more than 10 windows had reduced $PM_{2.5}$ F_{inf} compared with homes that had fewer windows (0.29 vs. 0.39) (Table 3). However, the authors did not discuss why having more windows was associated with reduced F_{inf} or whether the finding could be due to confounding variables, such as home age.[31] The same study also found that residential detached homes built after 1975 had lower average $PM_{2.5}$ F_{inf} values (0.27) compared with homes built before 1975 (0.41).

Energy efficiency

One study on wildfire smoke in Colorado by Shrestha et al. assessed the effect of building methods/construction type on smoke infiltration. Residential detached homes (n = 28) were classified as certified built green (n = 5), energy efficient retrofitted (n = 13), or non-energy efficient retrofitted (n = 10). Built green homes were designed with improved energy-efficiency, including air-tight construction, rooftop solar panels, all-electric air heating, and water heating systems; whereas specific criteria for energy efficient retrofitted homes were not reported. In the certified green homes, the I/O values were 1.02 for $PM_{0.5\text{-}2.5}$ number concentrations, 0.77 for black carbon, 3.94 for CO, and 1.29 for NO_2. These same values for the energy efficient retrofitted homes were 0.57, 0.77, 2.37, and 1.15, respectively. For the non-energy efficient retrofitted homes, they were 0.66 , 0.60, 2.79, and 1.09, respectively.[26] These results suggest that energy efficient retrofits provided the best protection from smoke infiltration. Although the certified built green program had measures that increased air tightness, two of the five homes had continuously operating exhaust fans, and three had heat recovery ventilators on timer switches. Operation of these mechanical systems during conditions of high outdoor $PM_{2.5}$ concentration would increase infiltration and offset the benefits of the other measures. These results highlight the need to account for potential air pollution events in mechanical system design and operation. Furthermore, it is important to note that built green certifications in Colorado may not be representative of other jurisdictions.

Mechanical ventilation

In the Shrestha et al. study of residential detached homes (n = 28), $PM_{0.5\text{–}2.5}$ particle number and black carbon F_{inf} values were significantly higher by 18% and 4%, respectively, when mechanical ventilation was present.[26] These results suggest that HVAC systems were increasing the concentrations of air pollutants indoors by bringing in more outdoor air that contained elevated levels of pollutants during smoke episodes. Note that the analysis of building type and mechanical ventilation reported in this study included combined data from periods of clear air and heavy smoke plumes (PM concentration ≤ 27 $\mu g/m^3$).

Seven studies examined public buildings with HVAC systems during smoke episodes (Table 4). In two classrooms using fans to draw outdoor air indoors through a MERV 13 filter, relatively low I/O ratios of 0.06 to 0.24[46] and F_{inf} 0.05 to 0.07[45] were reported by Tham et al. for $PM_{0.3-10}$ particle number concentrations during haze episodes. In addition, increased positive pressure was measured in these classrooms when these systems were running. The authors indicated bringing filtered outdoor air indoors would reduce infiltration of unfiltered air, and remove PM, which may explain the low values reported.

In another study, Montrose et al. examined the impact of wildfire smoke on skilled nursing facilities that had HVAC systems with MERV 13 filters and reported a $PM_{2.5}$ F_{inf} of 0.29 (95% CI: 0.28, 0.30) on non-wildfire days and 0.59 (95% CI: 0.49, 0.71) on wildfire days.[33] For context, the overall median $PM_{2.5}$ F_{inf} for the five wildfire-specific studies was 0.59 across all building types. Even though the skilled nursing facilities had HVAC systems, the authors indicated that increased infiltration of smoke $PM_{2.5}$ into skilled nursing facilities may be influenced by seasonality and human behaviours such as door and window use, and that a more detailed analysis of overall building characteristics may explain some of the infiltration variability observed in the study. Lastly Holder et al. examined 23 public buildings in detail during wildfire smoke episodes. Four factors were significantly associated with higher indoor $PM_{2.5}$ during smoke episodes in a multivariate analysis: filter bypass, HVAC condition, window type, and room pressure. Furthermore, there was a trend of increased I/O ratios observed when HVAC filter bypass was higher or when the HVAC system was in poor condition, and I/O ratios were decreased under positive pressure.[38]

Limited evidence on the impact of exhaust fans (i.e., stove hood fans or bathroom fans) during smoke episodes was identified in this review. These systems can have negative impacts on $PM_{2.5}$ infiltration during periods of poor outdoor air quality due to the potential to create negative pressure indoors if more air is exhausted than is drawn inside.[53] Shrestha et al. reported a significant association between the presence of a stove exhaust system and $PM_{0.5-2.5}$ number concentration, black carbon, and CO F_{inf} values. Both $PM_{0.5-2.5}$ number concentration and black carbon F_{inf} were reduced in homes with externally vented exhaust systems, but CO F_{inf} was not. In addition, the built green homes category in Shrestha et al. included five homes, of which three had heat recovery ventilation systems and two had continuously running exhaust fans. $PM_{0.5-2.5}$ number concentration, black carbon, and CO F_{inf} were all significantly higher in the built green homes compared with energy efficient retrofit homes or non-retrofit homes. However, these results represent combined days with and without wildfire smoke impacts and the results may change when looking at wildfire impacted days alone.

Table 4. HVAC systems and combustion-derived air pollution episodes

Reference	Episode	HVAC	Building Type	Outcome
Holder et al. 2025[38]	Wildfire	HVAC MERV 8 filter buildings (n = 18; 78% of the buildings), MERV 11 (n = 1), MERV 13 (n = 1), MERV 14 (n = 1), No HVAC system (n =2)	Church (n = 2), university (n = 2), office (n = 6), hotel (n = 1), fire station (n = 1), museum (n = 2), fitness centre (n = 7), childcare centre (n = 4), community centre (n = 3)	- $PM_{2.5}$ median (range) I/O for building occupied was 0.59 (0.28 to 0.76) and for building unoccupied was 0.60 (0.27, 0.88) during a heavy wildfire period in 2020 - $PM_{2.5}$ median I/O ratios increased when HVAC filter bypass was higher, or if the HVAC system was in poor condition, and decreased room pressure was positive
Montrose et al. 2022[33]	Wildfire	HVAC with MERV 13 filters	Nursing facility (n = 4) 15,000 to 30,000 ft^2	- $PM_{2.5}$ I/O ratio increased (0.27 to 0.56) from non-wildfire to wildfire days - $PM_{2.5}$ F_{inf} increased (0.29 to 0.59) from non-wildfire to wildfire days
Dev et al. 2021[34]	Wildfire	HVAC systems 1) University building: HVAC MERV 8 prefilter and MERV 11 final filter 2) House A: Ventilation off 3) House B: HVAC, MERV 11,	1 public building and 2 residential detached houses	- $PM_{2.5}$ I/O ratios during wildfire: University building: 0.37, House A: 0.14, House B: 0.44 - $PM_{0.3\text{-}10}$ number and mass concentrations during wildfire: University building: significantly elevated, House B: significantly elevated
Mendoza et al. 2021[54]	Wildfire	Two general AHUs, five dedicated exhaust fans for the BSL 3 labs, perchloric acid and radioisotope exhausts	Laboratory and office building (82,000 square feet)	- Indoor $PM_{2.5}$ concentration was 78% of outdoor during wildfire, and 29% during non-wildfire
Kaduwela et al. 2019[24]	Wildfire	HVAC in school equipped with MERV 8 filters	High school	- PM > 0.3µm number concentrations were 10–15 times higher indoors comparing wildfire with prior period
Tham et al. 2021[45]	Haze	FFU with 350 mm diameter axial fan 50 Hz, 1420 RPM, 2559 m3/h, with a MERV13 filter	School building, two classrooms (313 m^3), same location one floor apart	- PM I/O ratios reduced (0.47–0.50 to 0.05–0.07) comparing FFU room with control room - F_{inf} for all PM sizes were reduced (0.2-0.3 to 0.06-0.11) comparing FFU with control - Using a FFU inside during haze events reduced PM_{10}, $PM_{2.5}$, and PM_1 by 75%–85% (56, 48, and 30 µg/m^3)

Reference	Episode	HVAC	Building Type	Outcome
Cao et al. 2016[46]	Haze	Filters attached to fan coil unit, (1) F25: MERV 7, (2) F65: MERV 11, (3) F85: MERV 13, (4) F95: MERV 14	Five neighboring classrooms (60 m^2), Nanyang Technological University	- PM I/O ratios reduced from 0.2–0.62 to 0.06–0.24 comparing MERV 13 or 14 filters with control - MERV 13 filter reduced indoor $PM_{0.3\text{-}2.5}$ by 48% (68.5 $\mu g/m^3$) on heavy haze days (outdoor $PM_{0.3\text{-}2.5}$ 279 $\mu g/m^3$)
Chen et al. 2016[47]	Haze	Air-conditioning and mechanical ventilation system with MERV 7 filter	Staff office (300 m^2), Nanyang Technological University	- $PM_{0.3}$ particle concentration I/O ratio was reduced from 0.64 to 0.0.59 comparing AC on vs. off - Removal efficiencies for particles of 0.37–3.74 µm are significantly higher (4%–25%) in the AC on mode

Abbreviations: **AC:** air-conditioning, **BC:** black carbon, **FFU:** fan filter unit, **NR:** not reported, **$PM_{x\text{-}y}$:** particulate matter x-y µm in diameter, **RWC:** residential wood combustion.

Summary

In this review, the relationship between concentrations of indoor and outdoor air pollutants during smoke episodes was examined in 41 studies, including wildfire events or seasons, residential wood burning, haze events, biomass burning, and a landfill tire fire. Although the original research question of this review focused on non-PM pollutants and included studies on PAHs, trace metals, ions, inorganic gases, and levoglucosan, most studies focused on measuring concentrations of $PM_{2.5}$ (31 studies). These measurements were performed in a variety of settings that included public buildings such as schools, libraries, and care facilities, as well as apartment buildings and detached homes in North America, Australia, Singapore, Brazil, and Spain.

The evidence indicates that outdoor air pollution during smoke episodes strongly influences select indoor pollutant levels. Indoor vapour-phase LMW and HMW PAH concentrations were increased by three and six times respectively during wildfire periods in one study. For inorganic gases, four studies examining concentrations of CO, CO_2, and NO_2 indicated that outdoor smoke had little impact on indoor concentrations, likely because these gases have significant indoor sources compared with ambient levels. In one study where increased indoor CO concentrations were observed during a wildfire smoke episode, they did not reach levels of concern. Only one study examined indoor levels of trace metals, ions, and minerals during haze episodes and periods of good air quality, finding elevated levels of these pollutants indoors during the haze periods. Overall, more research is needed to understand the indoor infiltration of non-PM pollutants during smoke episodes, and their potential health risks.

Outdoor concentrations of PM during smoke episodes strongly influenced indoor concentrations. Accounting for indoor sources of PM, F_{inf} values indicated that outdoor $PM_{2.5}$ contributed to about 50% of the $PM_{2.5}$ measured indoors. Overall, indoor concentrations of $PM_{2.5}$ were consistently lower than outdoors (17/20 studies) but substantially elevated compared with indoor concentrations during periods without smoke (four studies, median 31.5 vs. 4.7 μg/m^3). When comparing against outdoor $PM_{2.5}$ concentrations, studies also showed that window closure reduced median indoor $PM_{2.5}$ concentrations by 32% (9 studies) versus 23% (11 studies) for studies where windows with either open for a portion of the time or window status was not assessed. Portable air cleaners (PACs) were highly effective, lowering indoor $PM_{2.5}$ by 71% (range 18–99%) when windows were closed. Conversely, when windows were left open, both infiltration and indoor concentrations of pollutants were higher (Figure 1).

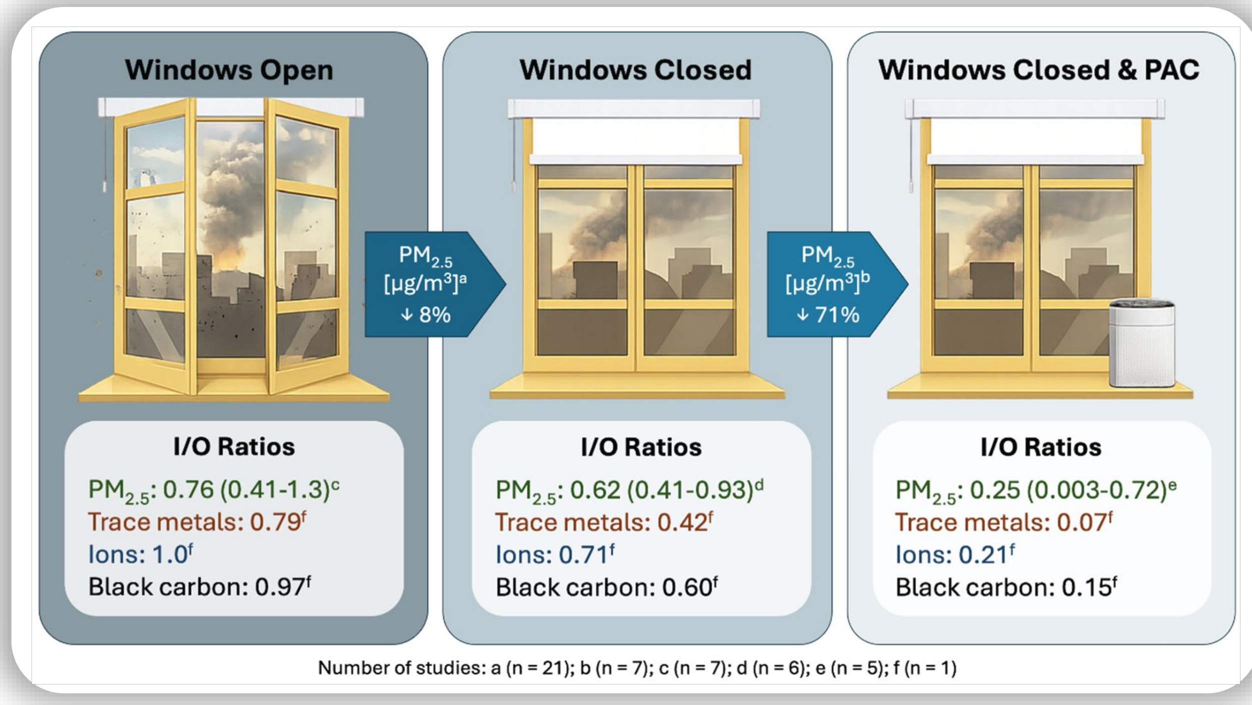
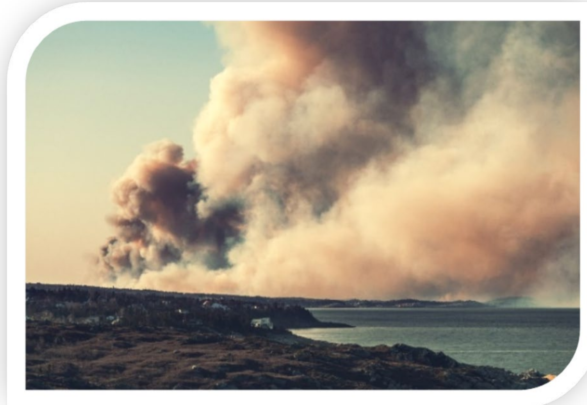


Figure 1: Summary of available evidence on the effect of closing windows and using a PAC on the infiltration of air pollutants indoors during a smoke episode. Indoor $PM_{2.5}$ mass concentrations are indicated as percent reduction from windows open or not reported[19,22,27,31,32,35-38,48,55,56] to windows closed[22,33,40,41,43-45,50,52] conditions, and from windows closed to windows closed while operating a PAC conditions[19,22,40,41,43,44,52]. I/O ratios are reported from available literature with median and range if available for $PM_{2.5}$[10,19,22,33,35,37,38,40,41,45,52,56], trace metals[19], ions[19], and black carbon[19].

Indoor exposures are also modified by building characteristics. Buildings with tighter building envelopes and newer construction provided better passive protection against $PM_{2.5}$ infiltration compared with older or leakier homes. Public buildings with HVAC systems and high-efficiency filters demonstrated some of the lowest infiltration values, especially when indoor positive pressure was maintained, and the HVAC system was in good condition. However, other HVAC-equipped facilities showed that substantial infiltration can occur during smoke events, suggesting that system design and filter type are critical to protecting indoor air quality. Beyond HVAC design, other important factors such as human behaviour likely contributed to elevated levels of pollutants measured in these facilities (i.e., window and door opening).

Several evidence gaps were identified in this review. Although $PM_{2.5}$ was well documented, the factors that influence infiltration of other smoke pollutants such as PAHs are less understood. Another evidence gap identified was the lack of studies on the impact of exhaust fans specifically during combustion-derived air pollution events. One study indicated that the presence of a continually running exhaust system in two homes may have led to higher infiltration of PM during periods of good air quality and during wildfire smoke episodes. However, the same study indicated that homes with an externally vented stove exhaust systems had reduced infiltration.

In summary, the findings of this review highlight that smoke pollution can readily penetrate indoors, but infiltration can be significantly reduced through building features, occupant behaviors, and interventions such as air sealing, PACs and HVAC filtration. Moreover, the most effective strategy to reduce indoor smoke pollution is to combine interventions such as closing windows, using a PAC, and maintaining positive pressure inside the building. Further study of non-PM pollutants is needed to better understand their indoor infiltration, and the potential long-term health impacts of exposure to smoke pollution.



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Appendix A

Table 5. Study details

Reference	Episode	Design	Date/Season	Setting	Pollutants	Sampling Period	Quality
Coker et al. 2025[57]	Wildfire	Analytical Cross Section	Jul–Oct 2022, May–Sep 2023	British Columbia, Canada Licensed childcare facilities (n = 39), Long term residential care facilities (n = 5	$PM_{2.5}$	122 days for 2022, 153 days for 2023	High
Holder et al. 2025[38]	Wildfire	Quasi-experimental	2019–2020	Missoula, Montana, USA Church (n = 2), university (n = 2), office (n = 6), hotel (n = 1), fire station (n = 1), museum (n = 2), fitness centre (n = 7), childcare centre (n = 4), community centre (n = 3)	$PM_{2.5}$	24-hour average, two fire seasons	High
Lee et al. 2024[25]	Wildfire	Analytical Cross Section	Aug–Oct 2022	British Columbia, Canada Licensed childcare facilities (n = 35)	$PM_{2.5}$, CO_2	66 days	High
Prathibha et al. 2024[50]	Wildfire and RWC	Quasi-experimental	Wildfire (Sep–Oct 2021), RWC (Jan–Mar 2022) (Of note: wildfires were extinguished	Hoopa, California, USA Residential detached homes, (n = 8 wildfire smoke study, n = 11 RWC)	$PM_{2.5}$	5–14 days per condition phase, each phase was sampled consecutively	High

Reference	Episode	Design	Date/Season	Setting	Pollutants	Sampling Period	Quality
			prior to data collection in the "wildfire" study; the "RWC" study captured open burning of trees felled during a storm)				
Wheeler et al. 2024[51]	Wildfire (prescribed burns)	Quasi-experimental	Mar–2021	Semirural Victoria, Australia Residential detached homes (n = 10), within 5 km of planned prescribed burns	$PM_{2.5}$	1 to 20 days surrounding a prescribed burn	Moderate
Lunderberg et al. 2023[58]	Wildfire	Analytical Cross Section	2021	USA Residences (n = 3, 977), type not specified	$PM_{2.5}$	2021	High
Walker et al. 2023[31]	Wildfire	Cohort	July–Oct 2022	Missoula, Montana, USA Residential detached homes (n = 14), multi-level apartment/condo (n = 2), other (n = 4)	$PM_{2.5}$	117.7 days maximum per house	High
Willis et al. 2023[40]	Wildfire	Quasi-experimental	Aug–Oct 2021	Butte, Montana, USA 2 comparable sized rooms in 3 public buildings	$PM_{2.5}$	10–11 days	High

Reference	Episode	Design	Date/Season	Setting	Pollutants	Sampling Period	Quality
Burke et al. 2022[59]	Wildfire	Analytical Cross Section	2011–2020	USA Residential homes (n = 1, 520)	$PM_{2.5}$	All available data prior to 2020	High
Ghetu et al. 2022[15]	Wildfire	Cohort	Aug–Nov, 2018–2020	Washington, Oregon, California, and Idaho, USA Residential detached homes (n = 15)	PAHs	Air sampling period 3–4 weeks through study duration, Aug–Nov, 2018–2020	High
He et al. 2022[37]	Wildfire	Analytical Cross Section	Sep 10–21, 2020	Seattle, Washington, USA Residential detached homes (n = 4), apartment (n = 1), office building (n = 2)	$PM_{2.5}$	11 days	Moderate
Montrose et al. 2022[33]	Wildfire	Quasi-experimental	Jan 1–Dec 1, 2020	Idaho, USA Skilled nursing facilities (n = 4)	$PM_{2.5}$	334 days (1326 days across all facilities, 90 wildfire days, 1159 non-wildfire days)	High
O'Dell et al. 2022[60]	Wildfire	Analytical Cross Section	2020	Western USA Buildings and homes (n = 3175), type not specified	$PM_{2.5}$	2020	High
Dev et al. 2021[34]	Wildfire	Quasi-experimental	Jun 2015 and Aug 2017	Fairbanks, AK, USA University building, and residential detached houses (n=2)	Size-resolved and mass $PM_{0.3-10}$ concentrations	10 min, 10 replicate measurements per site	High

Reference	Episode	Design	Date/Season	Setting	Pollutants	Sampling Period	Quality
Liang et al. 2021[32]	Wildfire	Cross-section	Nov 2018, Aug to Sep 2020	San Francisco, California, USA 1274 buildings analyzed, 1112 (87%) buildings were residential: houses (80%), condominiums or multi-family buildings (13%), and apartments (4%)	$PM_{2.5}$	61 days	High
May et al. 2021[41]	Wildfire	Quasi-experimental	Sep 13, 2020	Seattle, Washington, USA Residential detached home (n = 1)	$PM_{2.5}$	5 hours, 3.5 hours PAC on and 1.5 hours PAC off	Moderate
Mendoza et al. 2021[54]	Wildfire	Quasi-experimental	Aug 23–24, 2018	Taylorsville, Utah, USA Laboratory and office building	$PM_{2.5}$	2 days (wildfire period)	High
Wheeler et al. 2021[42]	Wildfire	Quasi-experimental	Aug–Nov, 2019	Port Macquarie, NSW, Australia Library	$PM_{2.5}$	53 days with PAC on and 41 days with PAC off, 140 hours with PAC on and 609 hours with PAC off analyzed	High
Xiang et al. 2021[43]	Wildfire	Quasi-experimental	Sep 16–18, 2020	Seattle, Washington, USA Residential detached homes (n = 5 PAC, n = 2 no filtration)	$PM_{2.5}$	18–24 hours PAC on and 18–24 hours PAC off per house, 71 hours with PAC on and 65.5 hours of PAC off analyzed	High

Reference	Episode	Design	Date/Season	Setting	Pollutants	Sampling Period	Quality
Stauffer et al. 2020[44]	Wildfire	Quasi-experimental	Aug–Sep 2018	Butte, Montana, USA University building	$PM_{2.5}$	10 hours, 6-day (8:00 am–6:00 pm) and 8-night (8:00 pm–6:00 am) sampling periods completed for a total of n = 48-day hours and n = 64 hours	High
Kaduwela et al. 2019[24]	Wildfire	Quasi-experimental	Oct–Nov, 2018	Albany, California, USA High school	PM number concentrations, CO_2	7 days prior to fire, 7 days during fire	High
Messier et al. 2019[61]	Wildfire	Analytical Cross Section	Aug 7–13, 2018	Eugene, Oregon, USA Type NR (n = 6)	PAHs	Air sampled every 24 hours for 7 days	High
Reisen et al. 2019[48]	Wildfire and RWC	Quasi-experimental	2013–2015	Yarra Valley and Gippsland, Victoria, Australia Residential detached homes (n = 7)	$PM_{2.5}$	2–14.5 hour sampling periods	High
Shrestha et al. 2019[26]	Wildfire	Analytical Cross Section	Aug 17–Oct 10, 2016, Jun 28–Sep 12, 2017	Denver, Colorado, USA Residential detached homes (n = 28)	$PM_{0.5\text{-}2.5}$ number concentrations, BC, CO, NO_2	2–7 days	High
Barn et al. 2008[36]	Wildfire and RWC	RCT	Winter 2004 (RWC), Summer 2004–2005 (wildfire smoke)	RWC: Prince George, British Columbia, Canada; residential detached homes (n = 21) Wildfire smoke: Southern British Columbia, Canada;	$PM_{2.5}$	48 hours, 24 hours PAC on and 24 hours PAC off	Moderate

Reference	Episode	Design	Date/Season	Setting	Pollutants	Sampling Period	Quality
				residential detached homes (n = 17)			
Henderson et al. 2005[52]	Wildfire	Quasi-experimental	Oct 2021–Jul 2022	Colorado, USA 4 residential home pairs (n = 8)	$PM_{2.5}$	24 hours per fire event	Moderate
Tham et al. 2021[45]	Haze	Quasi-experimental	Oct 20, 2015	Singapore School building	PM_{10}, $PM_{2.5}$, PM_1	3 hours	Moderate
Tran et al. 2021[19]	Haze	Quasi-experimental	Aug–Sep 2019	Singapore Residential apartment	$PM_{2.5}$, BC, Ions, WSTE	23 days for non-hazy days, 13 days for hazy days, 24-hour averages	High
Sharma et al. 2017[22]	Haze	Quasi-experimental	Sep–Oct, 2015	Singapore Residential apartment	$PM_{2.5}$	24 hours, for PAC condition, 12 hours with PAC on and 12 hours with PAC off	High
Cao et al. 2016[46]	Haze	Quasi-experimental	Sep 17–25, 2015	Singapore University building	$PM_{0.3-0.5}$, $PM_{0.5-1.0}$, and $PM_{1.0-2.5}$	9 days	High
Chen et al. 2016[62]	Haze	Quasi-experimental	Haze: June 14–29, 2013, Clear sky: Aug 13–26 2013	Singapore University building	Size- and time-resolved $PM_{0.01-10}$ concentrations	14 days (Haze period)	High
Yang et al. 2024[63]	RWC	Quasi-experimental	Jan 17 - Feb 25, 2022	Fairbanks, Alaska, USA Residential detached home (n=1)	$PM_{2.5}$, and PM oxidative potential (OP)	24 hours	Moderate
Bravo-Linares et al. 2016[56]	RWC	Analytical Cross Section	Winter, 2014–2015	Los Ríos Region, Chile Type NR (n=135)	$PM_{2.5}$, PAHs	24 hours, 3 to 10 days of sampling	Moderate

Reference	Episode	Design	Date/Season	Setting	Pollutants	Sampling Period	Quality
Kajbafzadeh et al. 2015[35]	RWC	RCT	Dec 2011–Aug 2012	Vancouver, British Columbia, Canada Residential detached homes (n = 20)	$PM_{2.5}$, levoglucosan	7 days	Moderate
Brown et al. 2014[64]	RWC	Analytical Cross Section	Jan–Feb, 2009–2010	Connecticut, USA Residential detached homes (n = 10)	$PM_{0.5,\ 2.5}$ number concentrations	Hourly, 3 days of sampling	High
Wheeler et al. 2014[28]	RWC	RCT	Dec 2009–April 2010	Annapolis Valley, Nova Scotia, Canada Residential detached homes (n = 31)	PM, $PM_{2.5}$, levoglucosan	3 days, 1 day wood burning appliance on, 1 day PAC on, 1 day PAC off per house	High
Allen et al. 2011[29]	RWC	RCT	Nov 2008–Apr 2009,	Smithers, British Columbia, Canada Residential detached homes (n = 25)	$PM_{2.5}$, levoglucosan	7 days	Moderate
Allen et al. 2009[55]	RWC	Quasi-experimental	Nov 2007–Apr 2008	Smithers and Telkwa, British Columbia, Canada Residential detached homes (n = 13), trailer (n = 2)	$PM_{2.5}$, levoglucosan	6 days	High
Weaver et al. 2019[27]	Biomass	Analytical Cross Section	Aug–Sep, Year NR	Mirpur, Dhaka, Bangladesh, India NR (n = 44)	$PM_{2.5}$	24 hours	Moderate

Reference	Episode	Design	Date/Season	Setting	Pollutants	Sampling Period	Quality
Cristale et al. 2012[17]	Biomass	Quasi-experimental	Aug 2007, Jan 2008	Araraquara, São Paulo, Brazil Residential detached home (n = 1)	PAHs	8 hours per day, 25 days	Moderate
Artinano et al. 2017[18]	Landfill fire	Quasi-experimental	May 26 and Jun 3, 2016	Seseña, Toledo, Spain School building	PM_1 number concentrations, BC, minerals, PAHs	24 hours	High

Abbreviations: **BC:** Black carbon, **PAH:** polycyclic aromatic hydrocarbons, **PM_{x-y}**: particulate matter x-y µm in diameter, **RCT**: randomized controlled trial, **RWC:** residential wood combustion, **WSTE:** water soluble trace elements.
Note: Further summary details on data collection and sampling methods are available upon request.

Appendix B

Table 6. Indoor and outdoor pollutant concentrations during combustion-derived air pollution episodes

Reference	Episode	Outdoor pollutant levels	Indoor pollutant levels	Indoor/Outdoor ratios	Infiltration factor (F_{inf})
Coker et al. 2025	Wildfire	2022 Wildfire Season: $PM_{2.5}$ median (IQR) (µg/m^3): 5.47 (3.26 to 11.14) 2023 Wildfire Season: $PM_{2.5}$ median (IQR) (µg/m^3): 5.41 (3.30 to 10.34) Non-Wildfire Season: $PM_{2.5}$ median (IQR) (µg/m^3): 4.57 (2.16 to 9.61)	2022 Wildfire Season: $PM_{2.5}$ median (IQR) (µg/m^3): 4.48 (2.93 to 7.96) 2023 Wildfire Season: $PM_{2.5}$ median (IQR) (µg/m^3): 4.40 (2.91 to 7.51) Non-Wildfire Season: $PM_{2.5}$ median (IQR) (µg/m^3): 4.31 (3.23 to 6.16)	2022 Wildfire Season: $PM_{2.5}$ median (IQR): 0.83 (0.60 to 1.12) 2023 Wildfire Season: $PM_{2.5}$ median (IQR): 0.82 (0.60 to 1.11) Non-Wildfire Season: $PM_{2.5}$ median (IQR): 0.97 (0.58 to 1.73)	NR
Holder et al. 2025	Wildfire	2019 Season: $PM_{2.5}$ median (range) (µg/m^3): 12.54 (9.41 to 17.16) 2020 Season: $PM_{2.5}$ median (range) (µg/m^3): 71.16 (64.39 to 82.97)	2019 Season: $PM_{2.5}$ median (range) (µg/m^3): 9.68 (4.91 to 11.64) 2020 Season: $PM_{2.5}$ median (range) (µg/m^3): 39.2 (22.73 to 61.61)	2019 Season: $PM_{2.5}$ median (range): Building occupied 0.84 (0.56 to 1.12), Building unoccupied 0.75 (0.58, 1.14) 2020 Season: $PM_{2.5}$ median (range): Building occupied 0.59 (0.28 to 0.76), unoccupied 0.60 (0.27, 0.88)	NR
Lee et al. 2024	Wildfire	$PM_{2.5}$ median (range) (µg/m^3): Wildfire days 20.7 (3.8 to 144.6), Non-wildfire days 6.3 (1.0–35.2)	$PM_{2.5}$ median (range) (µg/m^3): Wildfire days 14.4 (3.1 to 54.1), Non-wildfire days 4.3 (1.0–26.0)	$PM_{2.5}$ median (range): Wildfire days 0.64 (0.17 to 1.45), Non-wildfire days 0.76 (0.13–3.17)	NR

Reference	Episode	Outdoor pollutant levels	Indoor pollutant levels	Indoor/Outdoor ratios	Infiltration factor (F_{inf})
			CO_2 (ppm): Wildfire days 809.2, Non-wildfire days 689		
Prathibha et al. 2024	Wildfire and RWC	**Wildfire study** $PM_{2.5}$ mean ± SD (µg/m^3): Baseline 9.1 ± 8.4, DIY PAC 5.0 ± 1.7, commercial PAC 5.8 ± 3.1 **Wood stove study** $PM_{2.5}$ Mean ± SD (µg/m^3): Baseline 38.1 ± 24.4, DIY PAC 60.1 ± 35.3, commercial PAC 58.5 ± 38.5	**Wildfire study** $PM_{2.5}$ mean ± SD (µg/m^3): Baseline 14.0 ± 13.9, DIY PAC 10.5 ± 13, commercial PAC 10.5 ± 8.9 **Wood stove study** $PM_{2.5}$ Mean ± SD (µg/m^3): Baseline 34.1 ± 63.9, DIY PAC 23.1 ± 17.9, commercial PAC 22.6 ± 19.7	NR	**Wildfire study** $PM_{2.5}$ mean ± SD: Baseline 0.7 ± 0.2 , DIY PAC 0.7 ± 0.2, commercial PAC 0.8 ± 0.2 (*Wildfire smoke exposure only during the baseline phase and was extinguished by rainfall*) **Wood stove study** $PM_{2.5}$ mean ± SD: Baseline 0.7 ± 0.2, DIY PAC 0.6 ± 0.2, commercial PAC 0.6 ± 0.2 *(Calculated from PM-indoor data with indoor peaks removed and PM-outdoor data)*
Wheeler et al. 2024	Wildfire (prescribed burns)	$PM_{2.5}$ (µg/m^3): No summary data provided, levels peaked at approximately 250	NR	NR	NR
Lunderberg et al. 2023	Wildfire	Median outdoor PM2.5 concentrations increased to more than 20 µg/m^3	Increased by about 4 µg/m^3 during a wildfire impacted month	NR	NR
Walker et al. 2023	Wildfire	$PM_{2.5}$ mean ± SD (µg/m^3): wildfire period 36.8 ± 26.4, non-wildfire period 3.9 ± 3.0	$PM_{2.5}$ mean ± SD (µg/m^3): wildfire period 15.9 ± 14.7, non-wildfire period 3.6 ± 5.2		$PM_{2.5}$ mean (95% CI): wildfire period 0.32 (0.28, 0.36), non-wildfire period

Reference	Episode	Outdoor pollutant levels	Indoor pollutant levels	Indoor/Outdoor ratios	Infiltration factor (F_{inf})
					0.39 (0.37, 0.42) (*Calculated from I/O ratio with indoor peaks removed*)
Willis et al. 2023	Wildfire	$PM_{2.5}$ mean (µg/m^3): 1) Homeless shelter 42.17 2) Senior assisted living 44.79 3) School: 42.65	$PM_{2.5}$ mean (µg/m^3) 1) Homeless shelter: PAC 22.67, no PAC 31.52 2) Senior assisted living complex: PAC 21.95, no PAC 35.88 3) School: PAC 30.66, no PAC 37.20	$PM_{2.5}$ mean: 1) Homeless shelter: PAC 0.54, no PAC 0.75 2) Senior assisted living complex: PAC 0.49, no PAC 0.8 3) School: PAC 0.72, no PAC 0.87	NR
Burke et al. 2022	Wildfire	NR	No smoke: Median outdoor $PM_{2.5}$ concentrations (6 µg/m^3), infiltration declines by 0.0281 per 10 µg/m^3 increase in outdoor $PM_{2.5}$ (95% CI, −0.02925, −0.02810; $P < 0.001$). Smoke present: infiltration declines by 0.0209 per 10 µg/m^3 increase in outdoor $PM_{2.5}$ (95% CI, −0.02141, −0.02043; $P < 0.001$)	NR	NR
Ghetu et al. 2022	Wildfire	Average outdoor vapour-phase LMW PAH concentrations were three times higher during wildfires than before (not significant)	Average indoor vapour-phase LMW PAH concentrations were three times higher during wildfires than before (not significant)	Indoor vapour-phase PAH concentrations were significantly higher than outdoor concentrations before wildfires (8/9	NR

Reference	Episode	Outdoor pollutant levels	Indoor pollutant levels	Indoor/Outdoor ratios	Infiltration factor (F_{inf})
		Average outdoor vapour-phase HMW PAH concentrations were 86 times higher during wildfires than before (significant at most locations)	Average indoor vapour-phase HMW PAH concentrations were six times higher during wildfires than before (not significant)	locations), and significantly higher than outdoor concentrations at locations with an average AQI of less than 115 during wildfires Outdoor HMW PAH concentrations exceeded indoor concentrations at locations with an average AQI exceeding 115	
He et al. 2022	Wildfire	$PM_{2.5}$ mean (μg/m^3): 108.8 PAC & HVAC, 112.9 no filtration	$PM_{2.5}$ mean (μg/m^3): 57.4 PAC & HVAC, 92 no filtration	$PM_{2.5}$ mean: 0.43 PAC & HVAC, 0.82 no filtration	NR
Montrose et al. 2022	Wildfire	$PM_{2.5}$ mean ± SD (μg/m^3): 56.1 ± 46.5 wildfire days, 4.9 ± 4.8 non-wildfire days	$PM_{2.5}$ mean ± SD (μg/m^3): 31.1 ± 32.3 wildfire days, 1.3 ± 1.9 non-wildfire days	$PM_{2.5}$ mean (μg/m^3): 0.56 wildfire days, 0.27 non-wildfire days	$PM_{2.5}$ mean (95% CI): 0.59 (0.49, 0.71) wildfire days, 0.29 (0.28, 0.30) non-wildfire days *(Calculated from paired I/O ratio data with indoor peaks removed)*
O'Dell et al. 2022	Wildfire	$PM_{2.5}$ median (μg/m^3): Smoke Days: San Francisco - 21.37 Los Angles - 24.4 Seattle & Portland - 23.54 Salt Lake City - 18.27 Denver - 16.69 Smoke-free Days	$PM_{2.5}$ median (μg/m^3): Smoke Days San Francisco - 8.87 Los Angles - 10.88 Seattle & Portland - 6.8 Salt Lake City - 7.58 Denver - 9.88 Smoke-free Days	$PM_{2.5}$: Smoke Days San Francisco - 0.39 Los Angles - 0.44 Seattle & Portland - 0.17 Salt Lake City - 0.43 Denver - 0.58 Smoke-free Days	NR

Reference	Episode	Outdoor pollutant levels	Indoor pollutant levels	Indoor/Outdoor ratios	Infiltration factor (F_{inf})
		San Francisco - 5.22 Los Angles - 7.09 Seattle & Portland - 4.07 Salt Lake City - 4.91 Denver - 5.15	San Francisco - 4.83 Los Angles - 5.29 Seattle & Portland - 4.52 Salt Lake City - 4.65 Denver - 5.51 Median indoor $PM_{2.5}$ 82% (IQR: 43%–135%) or 4.3 μg/m^3 (IQR: 2.0–7.2 μg/m^3) higher on smoke-impacted days	San Francisco - 0.95 Los Angles - 0.81 Seattle & Portland - 1.2 Salt Lake City - 0.93 Denver - 1.0	
Dev et al. 2021	Wildfire	$PM_{0.3-10}$ mean ± SD (μg/m^3) 1) University: wildfire period 90.8 ± 13.32, non-wildfire period 8.3 2) House A: wildfire period 114.15 ± 8.85 , non-wildfire period 6.3 3) House B: wildfire period 59.1 ± 9.2 , non-wildfire period 5.1 $PM_{0.3-10}$ number concentration mean ± SD (number/cm^3) 1) University: wildfire period 1589.9 ± 202.2, non-wildfire period 8.3 ± 0.71 2) House A: wildfire period 1583.6 ± 16.35, non-wildfire period 6.5 ± 0.39 3) House B: wildfire period	$PM_{0.3-10}$ mean ± SD (μg/m^3) 1) University: wildfire period 33.5 ± 1.08, non-wildfire period 11.7 2) House A: wildfire period 16.7 ± 2.54 , non-wildfire period 5.5 3) House B: wildfire period 26.1 ± 1.78, non-wildfire period 4.4 $PM_{0.3-10}$ number concentration mean ± SD (number/cm^3) 1) University: wildfire period 1195.1 ± 19.9, non-wildfire period 8.02 ± 1.04 2) House A: wildfire period 206.9 ± 3.71, non-wildfire period 10.6 ± 0.55 3) House B: wildfire period	$PM_{0.3-10}$ mean ± SD 1) University: wildfire period 0.37 ± 0.06, non-wildfire period 1.7 ±0.3 2) House A: wildfire period 0.14 ± 0.01, non-wildfire period 0.6 ± 0.37 3) House B: wildfire period 0.44 ± 0.04, non-wildfire period 1 ± 0.96 $PM_{0.3-10}$ number concentration mean ± SD 1) University: wildfire period 0.76 ± 0.11, non-wildfire period 1± 0.17 2) House A: wildfire period 0.13 ± 0.001, non-wildfire period 1.6 ± 0.18 3) House B: wildfire period	NR

Reference	Episode	Outdoor pollutant levels	Indoor pollutant levels	Indoor/Outdoor ratios	Infiltration factor (F_{inf})
		1184.8 ±24.2, non-wildfire period 5.3 ± 0.3	721.7 ± 13.49, non-wildfire period 6.8 ± 0.39	0.61 ± 0.02, non-wildfire period 1.3 ± 0.03	
Liang et al. 2021	Wildfire	Non-fire days: $PM_{2.5}$ mean ± SD (µg/m^3): 9.1 ± 4.0 Fire days: $PM_{2.5}$ mean ± SD (µg/m^3): 45.4 ± 17.0	Non-fire days: $PM_{2.5}$ mean ± SD (µg/m^3): 4.1 ± 2.5 Fire days: $PM_{2.5}$ mean ± SD (µg/m^3): 11.1 ± 8.3	Non-fire days: $PM_{2.5}$ mean ± SD (µg/m^3): 0.9 ± 0.88 Fire days: $PM_{2.5}$ mean ± SD (µg/m^3): 0.41 ± 0.44	Non-fire days: $PM_{2.5}$ mean ± SD (µg/m^3): 0.45 ± 0.15 Fire days: $PM_{2.5}$ mean ± SD (µg/m^3): 0.27 ± 0.14
May et al. 2021	Wildfire	$PM_{2.5}$ mean ± SD (µg/m^3): 127 ± 9	$PM_{2.5}$ mean ± SD (µg/m^3) 1) Room A: PAC on 28 ± 2, PAC off 64 ± 2 2) Room B: PAC on 0.4 ± 0.4, PAC off 40 ± 2	$PM_{2.5}$ mean 1) Room A: PAC on 0.22, PAC off 0.50 2) Room B: PAC on 0.003, PAC off 0.31	NR
Mendoza et al. 2021	Wildfire	$PM_{2.5}$ mean ± SD (µg/m^3): no summary data reported	$PM_{2.5}$ mean ± SD (µg/m^3): no summary data reported	I/O ratio of readings significant ($p < 0.001$), Indoor concentration 78% of outdoor during wildfire, 29% during normal weekday	NR
Wheeler et al. 2021	Wildfire	$PM_{2.5}$ median (25%–75%) (µg/m^3): PACs on 23.3 (12.0–49.1), PACs off 30.7 (12.2–85.9)	$PM_{2.5}$ median (25%–75%) (µg/m^3): PACs on 5.7 (5.5–8.5), PACs off 20.0 (10.5–39.0)	NR	F_{inf} $PM_{2.5}$ median (25%–75%) (µg/m^3): PACs on 0.17, PACs off 0.32 Calculated from paired I/O data with indoor peaks removed
Xiang et al. 2021	Wildfire	$PM_{2.5}$ mean ± SD (µg/m^3): 64 ± 17	$PM_{2.5}$ mean ± SD (µg/m^3): PAC 14 ± 7, PAC off 47 ± 24	NR	$PM_{2.5}$ mean ± SD: PAC 0.19 ± 0.09, PAC off 0.56 ± 0.13

Reference	Episode	Outdoor pollutant levels	Indoor pollutant levels	Indoor/Outdoor ratios	Infiltration factor (F_{inf})
					(Estimated based on $PM_{2.5}$ mass balance models)
Stauffer et al. 2020	Wildfire	$PM_{2.5}$ mean ± SD (µg/m^3):17.47 ± 13.07	$PM_{2.5}$ mean ± SD (µg/m^3) Daytime: PAC on 2.95 ± 2.39, PAC off 11.09 ±9.70 Nighttime: PAC on 0.50 ± 0.39, PAC off 6.55 ±7.10	NR	NR
Kaduwela et al. 2019	Wildfire	PM number concentration: no summary data reported CO_2 (ppm): no summary data reported	PM number concentration: no summary data reported CO_2 (ppm): no summary data reported	NR	NR
Messier et al. 2019	Wildfire	No summary data reported	No summary data reported	I/O ratios not reported, indoor median and maximum concentrations ≥ than outdoor for most individual PAHs	NR
Reisen et al. 2019	Wildfire and RWC	Prescribed Burns: $PM_{2.5}$ mean ± SD (µg/m^3): 101.5 ± 83.28 RWC: $PM_{2.5}$ mean ± SD (µg/m^3): 18.54 ± 6.46	Prescribed Burns: $PM_{2.5}$ mean ± SD (µg/m^3): 73.35 ± 66.75 RWC: $PM_{2.5}$ mean ± SD (µg/m^3): 9.06 ± 3.09	Median $PM_{2.5}$ I/O range: 0.13 to 2.9	Prescribed Burns: $PM_{2.5}$ mean ± SD (µg/m^3): 0.58 ± 0.19 RWC: $PM_{2.5}$ mean ± SD (µg/m^3): 0.30 ± 0.13
Shrestha et al. 2019	Wildfire	$PM_{0.5\text{-}2.5}$ median (range) (#/cm^3) Wildfire plume density: none 3.36 (0.240–224), low 4.81 (0.650–58.3), medium 7.49 (0.750–168), high 24.9 (8.44–	$PN_{0.5\text{-}2.5}$ median (range) (#/cm^3) Wildfire plume density: none 2.15 (0.140–44.8), low 3.25 (0.280–52.4), medium 6.42 (0.710–563), high 9.85	$PN_{0.5\text{-}2.5}$ median (range) (#/cm^3) Wildfire plume density: none 0.81, low 0.80, medium 1.06, high 0.80	$PN_{0.5\text{-}2.5}$ median (range) (#/cm^3) Wildfire plume density: none 0.64, low 0.68, medium 0.86, high 0.40

Reference	Episode	Outdoor pollutant levels	Indoor pollutant levels	Indoor/Outdoor ratios	Infiltration factor (F_{inf})
		165) BC median (range) (ng/m^3) Wildfire plume density: none 590 (98.3–6900), low 745 (97.0–8280), medium 857 (54.5–39100), high 948 (176–21500) CO median (range) (ppm) Wildfire plume density: none 0.224 (0–4.10), low 0.206 (0–14.0), medium 0.253 (0–1.90), high 0.278 (0–2.80) NO_2 median (range) (ppb) Wildfire plume density: none 5.56 (4.48–9.24), low 9.60 (6.55–11.1), medium 9.39 (7.19–10.2), high 12.9 (5.21–20.6)	(4.08–30.4) BC median (range) (ng/m^3) Wildfire plume density: none 422 (82.3–3490) low 568 (144–4700), medium 632 (83.3–5080), high 495 (98.3–4320) CO median (range) (ppm) Wildfire plume density: none 0.526 (0–3.23), low 0.605 (0–10.5), medium 0.528 (0–5.30), high 0.88 (0.26–4.46) NO_2 median (range) (ppb) Wildfire plume density: none 6.39 (3.58–10.1), low 11.8 (8.00–13.8), medium 8.96 (6.48–11.9), high 10.1 (6.73–13.4)	BC median (range) (ng/m^3) Wildfire plume density: none 0.72, low 0.76, medium 0.75, high 0.57 CO median (range) (ppm) Wildfire plume density: none 5.01, low 3.96, medium 2.70, high 4.13 NO_2 median (range) (ppb) Wildfire plume density: none 1.18, low 1.03, medium 1.17, high 1.01	BC median (range) (ng/m^3) Wildfire plume density: none 0.72, low 0.80, medium 0.74, high 0.52 CO median (range) (ppm) Wildfire plume density: none 2.35, low 2.94, medium 2.09, high 3.17 NO_2 median (range) (ppb) Wildfire plume density: none 1.14, low 1.22, medium 0.96, high 0.78 *(Calculated from I/O ratio with indoor peaks removed)*
Barn et al. 2008	Wildfire and RWC	**Wildfire smoke** $PM_{2.5}$ mean ± SD (µg/m^3): PAC on 11.4 ± 10.0, PAC off 10.6 ± 6.8 **RWC** $PM_{2.5}$ mean ± SD (µg/m^3): PAC on 18.7 ± 19.4, PAC off 16.2 ± 14.2	**Wildfire smoke** $PM_{2.5}$ mean ± SD (µg/m^3): PAC on 4.9 ± 1.6, PAC off 8.2 ± 5.0 **RWC** $PM_{2.5}$ mean ± SD (µg/m^3): PAC on 3.9 ± 8.6, PAC off 5.8 ± 7.0	NR	**Wildfire smoke** $PM_{2.5}$ mean ± SD: PAC on 0.19 ± 0.2, PAC off 0.61 ± 0.27 **RWC** $PM_{2.5}$ mean ± SD: PAC on 0.10 ± 0.08, PAC off 0.28 ± 0.18 *(Calculated from paired I/O*

Reference	Episode	Outdoor pollutant levels	Indoor pollutant levels	Indoor/Outdoor ratios	Infiltration factor (F_{inf})
					data with indoor peaks removed)
Henderson et al. 2005	Wildfire	$PM_{2.5}$ mean (µg/m^3): **Prescribed burn:** House 1 (PAC): 21.7 House 2: 37.5 **Snaking wildfire**: House 1 (PAC): 7.52 House 2: 5.54 **Schnoover wildfire:** House 1 (PAC): 20.7 House 2: 19.6 **Hayman wildfire:** House 1 (PAC): 32.7 House 2: 32.9	$PM_{2.5}$ mean (µg/m^3): **Prescribed burn**: House 1 (PAC): 2.0 House 2: 21.8 **Snaking wildfire:** House 1 (PAC) 2.61 House 2: 5.16 **Schnoover wildfire:** House 1 (PAC): 1.43 House 2: 11.4 **Hayman wildfire:** House 1 (PAC) 3.02 House 2: 24.5	$PM_{2.5}$ mean: **Prescribed burn**: House 1 (PAC): 0.09 House 2: 0.85 **Snaking wildfire:** House 1 (PAC): 0.35 House 2: 0.93 **Schnoover wildfire:** House 1 (PAC): 0.06 House 2: 0.58 **Hayman wildfire:** House 1 (PAC): 0.09 House 2: 0.74	NR
Tham et al. 2021	Haze	PM_{10} mean (µg/m^3): 110 $PM_{2.5}$ mean (µg/m^3): 85 PM_1 mean (µg/m^3): 70	PM_{10} mean (µg/m^3): 66 $PM_{2.5}$ mean (µg/m^3): 58 PM_1 mean (µg/m^3): 40	PM_{10}: FFU 0.05, control room 0.50 $PM_{2.5}$: FFU 0.06, control room 0.49 PM_1: FFU 0.07, control room 0.47	F_{inf} PM_{10}: FFU 0.06, control room 0.20 F_{inf} $PM_{2.5}$: FFU 0.08, control room 0.29 F_{inf} PM_1: FFU 0.11, control room 0.30 (*Estimated based on $PM_{2.5}$ mass balance models*)
Tran et al. 2021	Haze	$PM_{2.5}$ mean ± SD (µg/m^3): Hazy days: PAC 60.93 ± 5.65, AC 37.22 ± 10.62, NV 49.78 ± 8.84 Non-hazy days: PAC 25.26 ±	$PM_{2.5}$ mean ± SD (µg/m^3): Hazy days: PAC 15.12 ± 3.79, AC 26.80 ± 7.64, NV 48.93 ± 9.72 Non-hazy days: PAC 5.26 ±	$PM_{2.5}$ mean ± SD: Hazy days: PAC 0.25, AC 0.72, NV 0.98 Non-hazy days: PAC 0.21, AC 0.63, NV 0.99	NR

Reference	Episode	Outdoor pollutant levels	Indoor pollutant levels	Indoor/Outdoor ratios	Infiltration factor (F_{inf})
		5.14, AC 25.48 ± 6.44, NV 25.77 ± 2.71 BC mean ± SD (µg/m^3): Hazy days: PAC 5.99 ± 2.01, AC 4.95 ± 1.78, NV 4.55 ± 1.76 Non-hazy days: PAC 3.86 ± 1.82, AC 3.94 ± 2.21, NV 4.01 ± 1.84 Total Ions mean (µg/m^3): Hazy days: PAC 18.96, AC 13.71, NV 14.93 Non-hazy days: PAC 8.89, AC 7.26, NV 9.47 Total WSTE mean (µg/m^3): Hazy days: PAC 2.24, AC 1.79, NV 1.84 Non-hazy days: PAC 0.96, AC 0.68, NV 0.95	1.36, AC 16.07 ± 2.90, NV 25.42 ± 2.70 BC mean ± SD (µg/m^3): Hazy days: PAC 0.90 ± 0.34, AC 2.97 ± 1.07, NV 4.40 ± 1.24 Non-hazy days: PAC 0.83 ± 0.18, AC 2.43 ± 0.68, NV 3.85 ± 1.31 Total Ions mean (µg/m^3): Hazy days: PAC 3.91, AC 9.68, NV 14.98 Non-hazy days: PAC 0.86, AC 3.97, NV 8.77 Total WSTE mean (µg/m^3): Hazy days: PAC 0.21, AC 0.70, NV 1.53 Non-hazy days: PAC 0.06, AC 0.29, NV 0.75	BC mean ± SD: Hazy days: PAC 0.15, AC 0.60, NV 0.97 Non-hazy days: PAC 0.22, AC 0.62, NV 0.96 Total Ions mean: Hazy days: PAC 0.21, AC 0.71, NV 1.00 Non-hazy days: PAC 0.1, AC 0.55, NV 0.93 Total WSTE mean: Hazy days: PAC 0.07, AC 0.42, NV 0.79 Non-hazy days: PAC 0.21, AC 0.51, NV 0.99	
Sharma et al. 2017	Haze	$PM_{2.5}$ mean ± SD (µg/m^3) PAC 72, Windows closed 157 ± 107, Windows open 94 ± 34	$PM_{2.5}$ mean ± SD (µg/m^3) PAC 23, Windows closed 98 ± 54, Windows open 71 ± 20	$PM_{2.5}$ mean ± SD PAC 0.32, Windows closed 0.62, Windows open 0.76	NR
Cao et al. 2016	Haze	$PM_{2.5}$ mean (µg/m^3): 88	$PM_{2.5}$ range (µg/m^3): Control: ~32 to ~130	$PM_{0.3\text{-}0.5}$ particle concentration mean: Control: 0.62, F25: 0.65, F65: 0.43, F85: 0.24, F95 0.21	NR

Reference	Episode	Outdoor pollutant levels	Indoor pollutant levels	Indoor/Outdoor ratios	Infiltration factor (F_{inf})
				$PM_{0.5-1.0}$ particle concentration mean: Control: 0.40, F25: 0.39, F65: 0.25, F85: 0.13, F95 0.13 $PM_{1.0-2.5}$ particle concentration mean: Control: 0.2, F25: 0.19, F65: 0.15, F85: 0.06, F95: 0.06	
Chen et al. 2016	Haze	$PM_{2.5}$ mean (µg/m^3): 96	$PM_{0.3-1.0}$ particle volume concentration mean (µg/cm^3): Hazy: 43.6, Clear sky: 4.8	$PM_{0.3}$ particle concentration mean: ACMV on 0.59, ACMV off 0.64	NR
Yang et al. 2024	RWC	$PM_{2.5}$: No summary data reported OP mass-normalized: No summary data reported	$PM_{2.5}$ mean ± SD (µg/m^3): 2.45 ± 0.58 OP mass-normalized mean ± SD (pmol/min/µg): 19.2 ± 16.5	OP mass-normalized mean ± SD: 0.53 ± 0.37	NR
Bravo-Linares et al. 2016	RWC	$PM_{2.5}$ mean (range) (µg/m^3): 85 (5, 367) PAH mean (range) (ng/m^3): 71 (3, 365)	$PM_{2.5}$ mean (range) (µg/m^3): 72 (6, 194) PAH mean (range) (ng/m^3): 51 (2, 291)	$PM_{2.5}$ regression: I/O 0.44, R2 = 0.48	NR
Kajbafzadeh et al. 2015	RWC	$PM_{2.5}$ mean ± SD (µg/m^3): PAC on 3.9 ± 2.1, placebo 5.0 ± 2.5 Levoglucosan mean ± SD	$PM_{2.5}$ mean ± SD (µg/m^3): PAC on 3.4 ± 1.9, placebo 6.5 ± 2.7	$PM_{2.5}$ mean ± SD: PAC on 0.87 ± 0.77, placebo 1.3 ± 0.65	NR

Reference	Episode	Outdoor pollutant levels	Indoor pollutant levels	Indoor/Outdoor ratios	Infiltration factor (F_{inf})
		(ng/m3): PAC on 20.6 ± 19.9, placebo 13.2 ± 13.5	Levoglucosan mean ± SD (ng/m3): PAC on 11.8 ± 14.4, placebo 29.3 ± 67.1		
Brown et al. 2014	RWC	NR	$PM_{0.5}$ mean ± SD (#/cm^3): OWF homes 0.302 ± 0.305, control homes 0.0718 ± 0.077 $PM_{2.5}$ mean ± SD (#/cm^3): OWF homes 6.58 ± 4.85, control homes 1.91 ± 2.40	NR	NR
Wheeler et al. 2014	RWC	PM median (range) (µg/m^3): PAC on 7.66 (0.91, 65.28), placebo 5.9 (0.51, 35.66) $PM_{2.5}$ median (range) (µg/m^3): PAC on 2.51 (0.42, 22.6), placebo 3.67 (0.34, 115.97) Levoglucosan median (range) (ng/m^3): PAC on 0.169 (0.055, 3.258), placebo 0.096 (0.021, 1.458)	PM median (range) (µg/m^3): PAC on 3.17 (0.9, 21.26), placebo 8.58 (2.6, 64.42) $PM_{2.5}$ median (range) (µg/m^3): PAC on 1.92 (0.35, 11.28), placebo 3.87 (0.37, 30.19) Levoglucosan median (range) (ng/m^3): PAC on 0.034 (ND, 0.189), placebo 0.050 (ND, 0.448)	PM median (range): PAC on 0.42 (0.08, 2.29), placebo 1.42 (0.24, 19.72) $PM_{2.5}$ median (range): PAC on 0.63 (0.08, 5.78), placebo 1.2 (0.03, 24.0) Levoglucosan median (range): PAC on 0.17 (0.00, 0.77), placebo 0.36 (0.00, 14.62)	F_{inf} PM median (range): PAC on 0.26 (0.07, 0.84), placebo 0.56 (0.16, 1.00) *(Calculated from I/O ratio with indoor peaks removed)*
Allen et al. 2011	RWC	$PM_{2.5}$ mean ± SD (µg/m^3): PAC on 9.8 ± 4.2, placebo 10.8 ± 4.2 Levoglucosan mean ± SD (ng/m^3): PAC on 530 ± 358, placebo 613 ± 548	$PM_{2.5}$ mean ± SD (µg/m^3): PAC on 4.6 ± 2.6, placebo 11.2 ± 6.1 Levoglucosan mean ± SD (ng/m^3): PAC on 33 ± 39, placebo 127 ± 191	NR	$PM_{2.5}$ mean ± SD: PAC on 0.2 ± 0.17, placebo 0.34 ± 0.17 *(Estimated based on $PM_{2.5}$ mass balance models)*

Reference	Episode	Outdoor pollutant levels	Indoor pollutant levels	Indoor/Outdoor ratios	Infiltration factor (F_{inf})
Allen et al. 2009	RWC	$PM_{2.5}$ range (5 to 95 percentiles) (µg/m^3): pre-exchange ~5 to 50, post-exchange ~5 to 18 Levoglucosan range (5 to 95 percentiles) (ng/m^3): pre-exchange ~200 to 2500, post-exchange ~50 to 1500	Indoor $PM_{2.5}$ median (µg/m^3): pre-exchange 12.8, and post-exchange 12.2 µg/m3 Levoglucosan median (ng/m^3): pre-exchange 113, post-exchange 109	NR	F_{inf} $PM_{2.5}$ mean ± SD: pre-exchange 0.32 ± 0.17, post-exchange sampling 0.33 ± 0.11
Weaver et al. 2019	Biomass	$PM_{2.5}$ mean ± SD (µg/m^3): 57.1 ± 19.9 CO mean ± SD (ppm): 0.006 ± 0.008	$PM_{2.5}$ mean ± SD (µg/m^3): Index home (stove) 51.9 ± 16.8, neighbour home 47.5 ± 14.9 CO mean ± SD (ppm): Index home (stove) 0.03 ± 0.04, neighbour home 0.006 ± 0.01	NR	NR
Cristale et al. 2012	Biomass	NR	Total PAHs mean (range) (ng/m^3) Harvest season: 22.9 (4.82 - 44.8), Non-harvest season: 2.35 (0.79–5.53)	NR	NR
Artinano et al. 2017	Landfill fire	PM_1 number concentrations maximum (#/cm^3): 3.8 × 10^5 BC maximum (µg/m^3): ~4 Total minerals mean (µg/m^3): 5639	PM_1 number concentrations maximum (#/cm^3): 3.9 × 10^4 BC maximum (µg/m^3): 1.8 Total minerals mean (µg/m^3): 3075	NR	NR

Reference	Episode	Outdoor pollutant levels	Indoor pollutant levels	Indoor/Outdoor ratios	Infiltration factor (F_{inf})
		Total PAHs maximum (µg/m^3): 0.13	Total PAHs maximum (µg/m^3): 0.02		

Abbreviations: **AC:** air-conditioning, **ACMV**: air-conditioning and mechanical ventilation, **AQI:** air quality index, **BC:** black carbon, **F25, F65, F85, and F95:** filters with dust-spot efficiencies of 25%, 65%, 85% and 95%, **FFU:** fan filter unit, **HMW**: high molecular weight, **LMW:** low molecular weight, **NR:** not reported, **NV:** naturally ventilated, **OP:** oxidative potential, **OWF**: outdoor wood furnaces, **PM_{x-y}**: particulate matter x-y µm in diameter; **PN:** particle number, **RWC:** residential wood combustion, **WSTE:** water soluble trace elements.

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