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Intake Fraction of Urban Wood Smoke

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Intake fraction (iF), the proportion of emissions inhaled by an exposed population, is useful for prioritizing sources with the greatest impact on population exposure per unit emissions. This article reports iF estimates for urban winter wood smoke emissions. We used two approaches, incorporating spatiotemporal statistical models for (1) winter wood smoke fine particulate matter ($PM_{2.5}$) emissions and concentration and (2) concentrations of levoglucosan (a wood smoke particulate marker). Empirical data used in our models were measured in Vancouver, Canada during 2004−2005. We used Monte Carlo simulations to quantify uncertainty. The estimated geometric mean iF (units: per million) is 13 (one geometric standard deviation range: 6.6−24) for wood smoke $PM_{2.5}$ and 15 (4.5−50) for levoglucosan. These iF estimates are comparable to or slightly larger than iF values for urban vehicle emissions reported in the literature. On average, higher-income areas have lower wood smoke $PM_{2.5}$ concentrations and intake. Our results emphasize the importance of urban wood smoke as a source of $PM_{2.5}$ exposure and highlight the comparatively large population exposure and potential environmental justice benefits from reducing wood smoke emissions.

Introduction

Fine particulate matter (aerodynamic diameter less than 2.5 μm; $PM_{2.5}$) is an important component of urban air pollution and is associated with substantial health impacts (*1*). Residential wood combustion for heating and aesthetic purposes is a significant source of wintertime $PM_{2.5}$ in many regions, including urban areas. Naeher et al. (*2*) indicate that wood burning produces greater than 30% of total annual $PM_{2.5}$ emissions in 8 Canadian provinces. In the Pacific Northwest, 2005 province-/state-wide inventories for British Columbia (BC) (*3*) and Washington (WA) (*4*) attribute 12% and 19%, respectively, of annual $PM_{2.5}$ emissions to residential wood burning, with 7% (BC) and 5% (WA) of households using wood for heating (*5, 6*). In both cases, residential wood burning $PM_{2.5}$ emissions are equivalent to or greater than the total emissions from mobile sources (road, off-road, and marine). At the urban scale, inventories for Metro Vancouver, BC (*7*) and Seattle/King County, WA (*8*) indicate that residential wood burning contributes 6% and 19% of annual $PM_{2.5}$ emissions, respectively. In Vancouver, 34% of households contain wood burning devices; these appliances are used primarily for aesthetic rather than heating purposes (*9*). In the future, the use of wood for residential heating is expected to increase owing to rising fuel costs, the perception of wood as a greenhouse gas neutral fuel, and adaptation to risk of extreme weather events (*10*−*13*).

Studies of wood smoke (WS) exposure have shown a range of adverse health impacts, including decreased lung function, increased respiratory symptoms in children, and increased emergency room visits (*2, 14*). Reviews have indicated that there is insufficient evidence to conclude that $WSPM_{2.5}$ is less harmful than other types of $PM_{2.5}$ (*2, 14*).

Relatively few $WSPM_{2.5}$ exposure studies have focused on large urban areas (*15*). Larson et al. (*16*) previously described a measurement and modeling approach to characterize winter WS concentrations in Vancouver and Victoria, BC. In Vancouver, mobile nephelometer measurements on cold winter nights indicated average $PM_{2.5}$ concentrations of 13 μg m^{-3} in residential neighborhoods. Similar winter night $PM_{2.5}$ concentrations were also measured in Seattle (*17*).

Intake fraction (iF) indicates, for a specific pollutant and source, the total mass intake per mass emitted; this metric facilitates comparisons among sources of a specific pollutant (*18*). Wood smoke is one of many $PM_{2.5}$ sources, and iF can help assess the impact of $WSPM_{2.5}$ exposure relative to other $PM_{2.5}$ sources such as traffic and fossil-fuel electricity generation. We used a rich empirical data set created from previous spatial analysis of WS emissions and concentration in Vancouver (*16, 19*) to estimate winter season iF values for $WSPM_{2.5}$ and fine particle levoglucosan (a biomass combustion molecular marker (*20*)). In addition, we investigated spatial and demographic variation in $WSPM_{2.5}$ exposure within Vancouver to highlight environmental justice aspects of urban wood smoke emissions.

Methods

Calculation. Intake fraction indicates the inhaled proportion of emissions (eq 1) (*18, 21*):

$$iF = \frac{\sum_{(people,time)} \text{mass intake of pollutant by an individual}}{\text{mass released into the environment}} = \frac{\sum_{i=1}^{m}\sum_{j=1}^{n} f(Q_i)\cdot f(D_i)\cdot f(P_j)\cdot f(C_{i,j})\cdot f(F_i)}{\sum_{i=1}^{m}\sum_{j=1}^{n} f(E_{i,j})} \quad (1)$$

Here, Q_i is the average population breathing rate (m^3 person^{-1} hour^{-1}), during time period i, D_i is the duration of time-period i (hours), P_j is the population in geographic area j (people), $C_{i,j}$ is the ambient concentration of WS in time period i and geographic area j (g m^{-3}), F_i is the outdoor-to-indoor infiltration factor in time period i (unitless), and $E_{i,j}$ is the WS emissions in time period i and geographic area j (g). The analysis considers m time periods and n geographic areas. We employed a Monte Carlo approach in which all input values were drawn from probability distributions (e.g., $f(Q_i)$) used to characterize input uncertainty.

We employed two independent approaches for calculating iF: one for $WSPM_{2.5}$ and one for $WSPM_{2.5}$ levoglucosan. Each

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TABLE 1. Summary of Input Parameters for Metro Vancouver Intake Fraction Calculations: Wood Smoke (WS) $PM_{2.5}$ and $WSPM_{2.5}$ Levoglucosan

	parameter	units	value	lower	upper	type	distribution	data sources
			Parameters Common to All Calculations					
	mean breathing rate (Q)	m^3 person^{-1} d^{-1}	14.5	12.5	16.5	SD	normal	*22, 23*
	population (P)[b]	person	1983853	1958856	2088997	limit	triangular	*26*
	infiltration: heating season (F_h)	–	0.49	0.34	0.64	SD	normal	*28*
			Residential $WSPM_{2.5}$ Intake Fraction Based on Spatial Models					
	winter emissions: $WSPM_{2.5}$ ($E_{WSPM1-3}$)	t	306	204	408	limit	triangular	*34*
wood burning nights	number of days (D_1)	d	21.5	7	36	limit	uniform	*32, 33, 45*
	night breathing rate (Q_1)	m^3 person^{-1} d^{-1}	11.6	10.0	13.2	SD	normal	*22, 23, 25*
	concentration (C_{1avg})[a]	μg m^{-3}	8.77[c]	6.74[c]	11.4[c]	GCI[c]	log-normal	*16, 32*
wood burning days	number of days (D_2)	d	21.5	7	36	limit	uniform	*32, 33, 45*
	day-time breathing rate (Q_2)	m^3 person^{-1} d^{-1}	17.4	15	19.8	SD	normal	*22, 23, 25*
	concentration (C_{2avg})[a]	μg m^{-3}	3.86[c]	2.21[c]	6.72[c]	GCI[c]	log-normal	*32, 34*
shoulder season	number of days (D_3)	d	139	110	168	limit	uniform	*32, 33, 45*
	concentration (C_3)	μg m^{-3}	0.26[c]	0.11[c]	0.62[c]	GCI[c]	log-normal	*32, 34–36*
			Residential $WSPM_{2.5}$ Levoglucosan Intake Fraction Based on Measurements					
winter (sampling period)	number of days (D)	d	162					*16*
	winter emissions: WS levoglucosan (E_{levo})	t	11.9[c]	6.16[c]	22.8[c]	GCI[c]	log-normal	*7, 9, 34*
	concentration (C_{levo})	ng m^{-3}	74[c]	29[c]	183[c]	GCI[c]	log-normal	*16*

[a] Spatial average of 385 census tract $WSPM_{2.5}$ concentrations. [b] Population values for each of the 385 census tracts used in calculation. [c] Parameter values were lognormally distributed; values presented are geometric means μ_{geo} and single geometric standard deviation σ_{geo} confidence intervals ($\mu_{geo}\sigma_{geo}^{-1} - \mu_{geo}\sigma_{geo}$).

approach covered the same spatial (Vancouver) and temporal (winter season) extents but incorporated a different number of spatial and temporal divisions, as detailed below. Parameter values are described next and listed in Table 1, with flowcharts and additional calculation details provided in the Supporting Information.

Input Data. *Parameters Common to All Calculations.* Available estimates for population average breathing rate (units: m^3 person^{-1} d^{-1}) vary, with commonly used values ranging from 12.2 (*22*) to 17 (*23, 24*). We used a daily mean breathing rate of 14.5, with an uncertainty range of ±14% (i.e., standard deviation of 2). Following Marshall (*25*), we adjusted this breathing rate up by 20% for daytime and down by 20% for nighttime breathing rates. Aggregate and census tract population data for Vancouver were obtained from the 2001 Canadian Census (*26*), which reports coverage errors of approximately −5.3% to +1.3% (*27*).

Because people spend the majority of time indoors, especially in the winter, it is important to account for systematic differences between measured outdoor $PM_{2.5}$ concentrations and indoor concentrations. The infiltration factor, F, is defined as the fraction of ambient $PM_{2.5}$ that penetrates indoors and remains suspended (*28*). Estimates were obtained from Hystad et al. (*28*), who measured infiltration using indoor/outdoor nephelometers in Seattle and Victoria, locations with similar housing types and climatic conditions to our study region.

Residential $WSPM_{2.5}$: Land-Use Regression. Recent research (*16, 19*) provided a spatially disaggregated wood smoke particle light-scattering land-use regression (LUR) surface for Vancouver, which we used to estimate residential $WSPM_{2.5}$ concentrations during winter wood burning periods. Following eq 1, we calculated $WSPM_{2.5}$ intake (numerator of eq 1, total mass inhaled for a given time period and spatial extent) separately for 385 census tracts in Metro Vancouver (year-2001 Census of Canada (*26*)) and for three time periods: winter daytime, winter nighttime, and shoulder heating-season in spring and fall. Census data provides per-tract Quintile Adjusted Income Per Person Equivalent (QAIPPE), an economic indicator from 1 (low income) to 5 (high income) (*26*) that we used to compare WS intakes among income groups.

The LUR, described in detail by Larson et al. (*16*), was fit to nephelometer light scattering measurements (units: m^{-1}) collected by a mobile monitoring approach during calm, cold nights (6:00p–1:00a) during October 2004–April 2005. The LUR provides a particle light-scattering spatial surface that we converted to $PM_{2.5}$ concentrations using the coefficient developed by Larson et al. during their sampling period: 0.345 g m^{-2} (intercept = 0, range of uncertainly of ±25%). This coefficient value agrees well with other comparisons of light scattering and $PM_{2.5}$ concentration (*29–31*).

Concentrations derived from the LUR model represent $PM_{2.5}$ during calm, cold winter nights, i.e., times when residential WS emissions are comparatively large and are expected to be a significant contributor to $PM_{2.5}$ concentrations. In order to disaggregate the WS and non-WS portions of the LUR $PM_{2.5}$ values, we subtracted geometric mean summer (nonburning season) nighttime regulatory $PM_{2.5}$ concentrations (4.8 μg m^{-3}, range 2.9–8.8) (*32*) from all $PM_{2.5}$ LUR concentration values. LUR $WSPM_{2.5}$ concentrations were then spatially averaged over each census tract (Figure 1b).

Analysis of regulatory $PM_{2.5}$ measurements for night time (7:00p–6:00a) October 2004–April 2005 showed that Larson et al.'s LUR model geometric mean $PM_{2.5}$ concentration (13 μg m^{-3}), applicable to winter nights, is consistent with regulatory monitoring data for between 168 and 800 h per year. Based on correlation between observed two week average levoglucosan levels and meteorological data, Brauer et al. (*33*) concluded that the LUR is valid for roughly 72 (winter) nights annually (heating degree days greater than 12 °C). Combining these two analyses, we assumed a uniform range of LUR model applicability from 168 h (14 12-h nights) to 864 h (72 12-h nights). Regulatory $PM_{2.5}$ measurements (*32*) and survey data (*34*) indicate that wood burning in Vancouver varies diurnally, with nighttime $PM_{2.5}$ peak concentration values at least twice daytime values. As such, for the winter daytimes directly adjoining the winter nights of LUR applicability, we assumed mean daytime $WSPM_{2.5}$

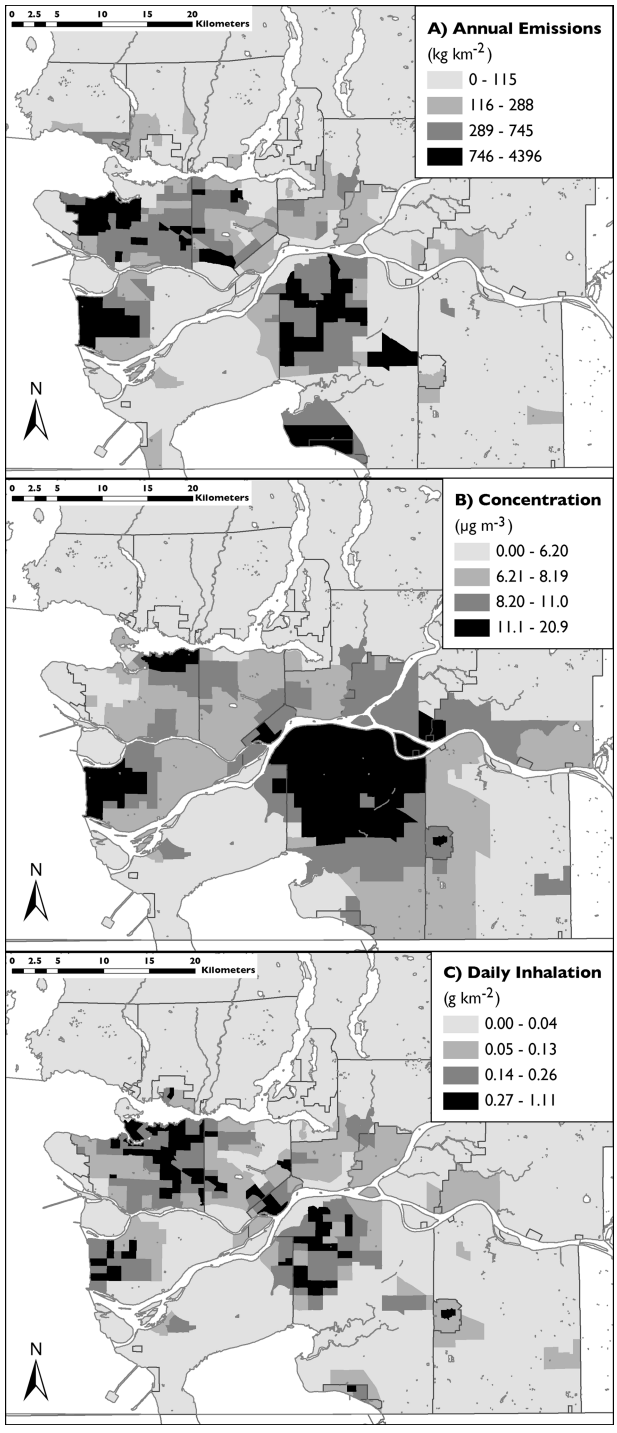

FIGURE 1. a) Estimated annual residential wood smoke $PM_{2.5}$ emissions (per km^2) in Vancouver (*25*). (b) Land-use regression estimated cold winter night wood smoke $PM_{2.5}$ concentrations in Vancouver. (c) Estimated 24-h residential wood smoke $PM_{2.5}$ mass intake (per km^2) for Vancouver during peak winter wood burning period. For all plots, each legend bin represents 25% of the total tract values.

concentration values of half of the LUR nighttime value. Given the uncertainty in the concentration during winter daytimes, we used the LUR winter nighttime concentration as an upper bound and 0 $\mu g\ m^{-3}$ as a lower bound.

Outside the time range when the LUR model is expected to be applicable (wood burning nights/days), levels of residential wood burning are expected to be low. Wood burning survey data (*9, 34*) suggest two low wood burning time periods: (1) the spring and fall shoulder heating-season period, experiencing moderate wood burning activity (110−175 days) and (2) the summer period (183 days). Because summer wood burning is expected to account for 10% or less of annual emissions (*34*), and because no spatial summer wood smoke concentration data were available, we focused on the winter intake fraction. During the shoulder season period, we estimated the $WSPM_{2.5}$ concentrations as the product of the regulatory $PM_{2.5}$ monitoring concentration (*32*) (adjusted to remove secondary $PM_{2.5}$ (*35*) and background $PM_{2.5}$ from outside the airshed (*36*)) and the fraction of the seasonal $PM_{2.5}$ emission inventory represented by residential wood burning (*7*). Given the uncertainty in these concentration values, we doubled our estimate as an upper bound and used 0 $\mu g\ m^{-3}$ as a lower bound. Outside of time periods when the LUR model is applicable, we have no data regarding the spatial variability of $WSPM_{2.5}$ concentration, so we applied our mean estimates to all 385 census tracts.

Metro Vancouver (*7, 9*) and the BC Ministry of Water, Land and Air Protection (*34*) provide estimates of year-2000 residential wood burning fuel mass and $WSPM_{2.5}$ emissions. Implicit in these estimates are the type and volume of wood burned and the type and number of wood burning devices used, both estimated from survey results. No estimates of uncertainty are provided for the $WSPM_{2.5}$ emissions inventory, so we assume an uncertainty range of ±33%. Based on wood burning survey data (*34*) we allocated 90% of total emissions to winter months (November−April).

Residential $WSPM_{2.5}$: Levoglucosan Measurements. Levoglucosan is a significant component of $WSPM_{2.5}$ (*20, 37*). Because the fate and transport of $WSPM_{2.5}$ is likely to be similar for levoglucosan-containing and for non-levoglucosan-containing $WSPM_{2.5}$ particles, the LUR model and the levoglucosan measurements provide independent estimates for $WSPM_{2.5}$ iF.

Available levoglucosan data for Vancouver (*16*) are spatially less precise than the LUR. To apply eq 1 to levoglucosan we use a single spatial extent (Vancouver) and one time period: the 163 days during which levoglucosan concentrations were measured (November−April). Our levoglucosan emission estimate was obtained by combining wood burning estimates (*7*) with literature-derived levoglucosan emission factors; see the Supporting Information for details. As with $WSPM_{2.5}$, we allocated 90% of total emissions to the winter months (the 163 days in during which levoglucosan was measured) (*9*).

Uncertainty Analysis. All input values listed above were characterized by a central estimate and range of variation (Table 1). All iF calculations were performed as Monte Carlo simulations, with inputs characterized by probability distributions. Measured concentrations were lognormally distributed, while LUR-modeled $PM_{2.5}$ concentrations, breathing rate, and infiltration factors were normally distributed. For all other input variables, we employed triangular distributions. The range of days over which the LUR model was assumed to be valid was characterized by a uniform distribution. Each simulation sampled input distributions 10,000 times to produce the resulting iF distributions reported below.

Results

Table 1 provides a summary of iF calculation inputs, including concentration and emissions values we derived from the literature. Our winter levoglucosan emissions estimate of 10.7 t is based on wood mass of 23,400 t and an emission factor of 460 mg levoglucosan per kg wood burned. Our estimated winter season (November−April) mean $WSPM_{2.5}$ concentration is 1.6 $\mu g\ m^{-3}$, with a cold winter night (≤5 °C) mean concentration of 8.5 $\mu g\ m^{-3}$. Measured levoglucosan winter mean concentration is 74 $ng\ m^{-3}$.

TABLE 2. Summary of Intake Fraction Calculation Results for Metro Vancouver: Wood Smoke (WS) $PM_{2.5}$ and WS Levoglucosan

emission source	winter intake fraction[a] (per million)
wood smoke $PM_{2.5}$	13 (1.9; 6.6−24)
wood smoke levoglucosan $PM_{2.5}$	15 (3.3; 4.5−50)

[a] Intake fraction results are lognormally distributed. Values presented are geometric means μ_{geo}, with geometric standard deviation σ_{geo} and one σ_{geo} confidence range, $\mu_{geo}\sigma_{geo}^{-1} - \mu_{geo}\sigma_{geo}$.

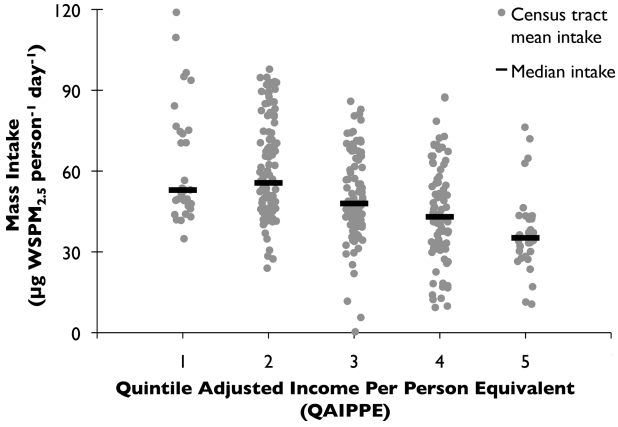

FIGURE 2. Metro Vancouver census tract daily wood smoke (WS) $PM_{2.5}$ mass intake (g $WSPM_{2.5}$ inhaled per person per day) based on winter night $WSPM_{2.5}$ concentrations and quintile-adjusted income per person equivalent (QAIPPE). QAIPPE=1 is lowest-income, QAIPPE=5 is highest-income.

Intake fraction results are presented in Table 2. The $WSPM_{2.5}$ and levoglucosan approaches yield independent estimates for WSiF (units: per million) that exhibit excellent agreement (13 and 15, respectively). Combining the two approaches, we estimated that winter WSiF in Vancouver is in the range 4.5−50, with a best-estimate of 14. Area-normalized annual wood-burning $WSPM_{2.5}$ emissions (kg km^{-2}), LUR $WSPM_{2.5}$ concentration estimates, and LUR-based inhalation estimates (g km^{-2}) are shown in Figure 1.

Wood Smoke Intake and Economic Status. In calculating WSiF, an intermediate step is the calculation of intake, the numerator of eq 1. Figure 2 presents mean winter night $WSPM_{2.5}$ intake (mass $WSPM_{2.5}$ inhaled per person per day) versus census tract QAIPPE. Lower QAIPPE census tracts experience higher WS concentrations and greater WS intake. Median WS intake is 1.5 times higher for lowest vs highest-income tracts. Differences between median WS intake are significant (Mann−Whitney Test, $p < 0.05$) in all cases except between QAIPPE quartiles 1 and 2.

Discussion

Our goal was to quantify WSiF. We hypothesized that because of the timing of WS emissions (winter nights, when air is relatively stagnant) and because of their spatial location (residential areas), iF would be greater for WS than for many outdoor $PM_{2.5}$ emission sources. A one-compartment steady-state model for inert pollutants (following Marshall et al. (*38*)) for the Vancouver region suggests that iF values during winter nights (7:00p−6:00a, November−April) are 1.4 times greater than during winter days and 2.5 times greater than during summer days. Preliminary analysis of the annual-average iF for all-source $PM_{2.5}$ in Vancouver (basis: regulatory monitoring data during 2004−2005, assuming ambient $PM_{2.5}$ is 60% primary (*35*), 2 $\mu g\ m^{-3}$ background contribution from outside Metro Vancouver (*35*); year-2000 emission inventory (*7*)) suggests a median value of 2 per million, or only 14% of the WSiF value reported here. Thus, available evidence indicates that iF is greater for WS than for many sources of $PM_{2.5}$. Further investigation into specific non-WS sources in Vancouver would be required to directly test the hypothesis.

No previous studies of seasonal iF for $PM_{2.5}$ from urban wood burning are available, so direct comparisons with other locations are not possible. However, the iFs determined by this study are consistent with prior research on iFs for area-source urban air pollution, which indicate iFs for an outdoor release in an urban area to be on the order of 1−100 per million (*21, 39, 40*).

Direct comparisons to iFs for other Vancouver $PM_{2.5}$ sources such as traffic emissions are not possible because iFs for these sources are not currently available. Non-Vancouver estimates of iF for nonreactive urban vehicle emissions in urban areas, while not directly comparable given differences in population sizes/densities of study areas, include the following (units: per million; values adjusted to reflect a breathing rate of 14.5 $m^3\ person^{-1}\ d^{-1}$): 2−14 in the US and Germany (*41, 42*); in the US, 12 (range: 7−37) in summer and 18 (range: 10−64) in winter (*43*); and, also in the US, 1.8 (range: 0.1−18) (*44*). Our $WSPM_{2.5}$ iF estimates (14; uncertainty range: 4.5−50) are comparable to or slightly larger than those reported iF values for urban vehicles. In Vancouver, annual $WSPM_{2.5}$ emissions (332 t) are only 20% less than annual traffic $PM_{2.5}$ emissions (417 t) (*7*), suggesting that despite the seasonal nature of WS emissions, wood burning and traffic likely have similar impacts on annual $PM_{2.5}$ exposures.

We focused our WSiF analysis on the winter period in Vancouver, in part because detailed measurement data for $WSPM_{2.5}$ or levoglucosan concentrations outside this period were not available. Evidence supporting our winter focus includes the following: a) survey data indicating 90% of annual residential wood burning for southern BC urban areas occurs in the winter, mainly in the evening (*9*), b) summer temperatures high enough to mitigate the need for home heating (*45*), c) visual observations of significant residential wood smoke plumes on cold winter evenings, d) correlation between high measured levoglucosan concentrations and low winter temperatures (*16*), and e) low measured summer levoglucosan levels (especially given other possible summer levoglucosan sources such as slash burning and forest fires) (*46*). In order to test the impact of focusing on winter, we assumed conservative summer $WSPM_{2.5}$ and levoglucosan concentrations (10% of mean winter concentrations) and performed annual iF calculations, which resulted in geometric mean values of 12 and 16 for $WSPM_{2.5}$ and levoglucosan, respectively - values that are nearly identical to the winter-only iF results (Table 2). Despite the fact that the spatially resolved LUR concentration estimates were only applied to 14−72 days per year, and the measured levoglucosan concentrations to 164 days per year, these high concentrations dominate both seasonal and annual $WSPM_{2.5}$ mass intake fractions, supporting our focus on winter.

We assumed that $PM_{2.5}$ emissions outside our study region do not significantly impact $PM_{2.5}$ concentrations inside the region, especially during winter nights. Weather conditions during winter nights generally involve little or no winds (75% less than 2.1 m s^{-1}), wind direction is predominately east northeast (*45*), and mean mixing heights are under 200 m. Thus, wood burning emissions from nearby population centers (Fraser Valley Regional District and Whatcom County, WA) are unlikely to have a strong impact on Vancouver wood smoke and levoglucosan concentrations. Similarly, because winds are often stagnant during times of wood burning, the

impact of Vancouver wood smoke emissions on adjoining population centers is also expected to be minor.

Our calculations ignored secondary $PM_{2.5}$ from WS emissions since fireplaces and conventional wood stoves emit ~10 times more primary $PM_{2.5}$ than NO_x per unit wood burned (*7*). Therefore, the impact of NO_x-derived secondary $PM_{2.5}$ from WS is expected to be small. As another source of uncertainty, the LUR model was developed using nephelometer measurements converted to $PM_{2.5}$ concentrations by comparing winter daily average light scattering values with average readings for a co-located Tapered Element Oscillating Microbalance (TEOM). TEOMs systematically underestimate $PM_{2.5}$ concentrations, especially in areas impacted by wood burning (*47*). A review of winter season two week average Harvard Impactor (HI) $PM_{2.5}$ measurements from one regulatory monitoring site in Vancouver indicates that TEOM values are systematically lower than HI values by 20−30%, suggesting that our estimated $WSPM_{2.5}$ iF would increase accordingly. However, the available HI data from one site were insufficient to support applying this adjustment to our estimates.

Given that levoglucosan $PM_{2.5}$ and non-levoglucosan $PM_{2.5}$ exhibit similar fate and transport in the environment, we expected (and found) similar central-tendency (geometric mean) iF estimates levoglucosan and $WSPM_{2.5}$. Differences in the confidence intervals − the uncertainty range is a factor of 2 for $WSPM_{2.5}$ and a factor of 3 for levoglucosan − reflect the greater uncertainty in levoglucosan concentrations and emissions. In addition to residential wood burning, other biomass combustion sources can also produce levoglucosan (*20, 37*); however, these sources are not expected to be significant contributors in winter. In addition, levoglucosan emissions factors for residential wood burning vary widely by wood type, wood-burning appliance type, and combustion conditions (see the Supporting Information).

The spatial intake map in Figure 1c provides insight into which areas of Vancouver are expected to have the highest mass intake of $WSPM_{2.5}$ per area. Not surprisingly, population density plays a strong role: intake hotspots (Figure 1c) are associated more closely with areas of high population density than with WS emissions or concentration hotspots (Figure 1a,b). This result reinforces the importance of exposure vs emissions based approaches to understanding and controlling air pollution, especially in cities such as Vancouver that are experiencing urban densification. Modest $WSPM_{2.5}$ emissions in a high-density urban neighborhood may yield only modest nearby ambient concentrations but at a large exposure impact; conversely, significant emissions in a low-density region such as the urban fringe may yield high concentrations but at low exposure impact.

The relationship between income and WS intake (Figure 2) mirrors the relationship between population density and income. No significant relationship was observed between WS emissions and income in Vancouver, so the higher WS intake at low income levels most likely results from higher population densities in lower income neighborhoods rather than a tendency of low-income households to use wood for home heating (see the Supporting Information). The income/intake relationship for WS observed in Vancouver may or may not hold in other cities. A similar relationship was observed in Southern California for several primary pollutants (*48*), reflecting proximity to freeways and industry. Although not accounted for in our analysis, infiltration rates are likely to be greater for lower-income than for higher-income residences (*49*), exacerbating exposure disparities between income groups.

Our study did not consider a potentially important source of exposure to WS: self-pollution. Fireplace and stove chimneys are designed to deliver emissions outdoors, but a small fraction of the emissions may be directly emitted indoors (e.g., when opening the stove for refueling) (*50, 51*), depending on stove design and operation. Even if only a small portion of the total emissions enters the indoor environment directly, self-pollution may be a potentially important contributor to overall iF, because iF is ~100−1000 times greater for indoor than for outdoor releases (*21*).

The approach taken by our study provides guidance for the calculation of iFs in other locations concerned about residential WS emissions. The current study and the previous spatial modeling studies upon which it relies (*16, 19*) provide a comprehensive methodology for modeling $WSPM_{2.5}$ emissions and concentration and for assessing WSiF and spatial/demographic intake patterns. For Vancouver, this approach indicates the importance of residential wood burning as a source of $PM_{2.5}$ exposure, even compared to more commonly studied and highly regulated emissions such as traffic exhaust. More broadly, our results underscore the potential public health benefits from incorporating exposure science into air pollution mitigation (*52*).

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Supporting Information Available

Table of levoglucosan emissions factors, 8 figures (flowcharts; diurnal $PM_{2.5}$ and wood-burning patterns; Monte Carlo sensitivity analysis, wind rose; population density analysis), and text describing calculation methodological details. This material is available free of charge via the Internet at http://pubs.acs.org.

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