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Pollutant Control Strategies for Acceptable Indoor Air Quality and Energy Efficiency in Retail Buildings

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Pollutant Control Strategies for Acceptable Indoor Air Quality and Energy Efficiency in Retail Buildings

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Dedication

To my beloved grandmother, Fatima Hachem

To your time spent memorizing lessons from the neighbor,
To your patience teaching them back to your children,
You instilled in all of us the love of what you were denied,
And defended our right for a better future.

I raise my glass high in your direction,
And celebrate your love for education.

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Pollutant Control Strategies for Acceptable Indoor Air Quality and Energy Efficiency in Retail Buildings

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Indoor air is associated with substantial health risks and is estimated to be responsible for the loss of over 4.7 million healthy life years (years lost due to morbidity and mortality) annually in the U.S. The highest indoor air-related health benefits can be expected from policies and strategies that efficiently target pollutants having the greatest contribution to the burden of disease. This burden is caused by indoor sources as well as by outdoor pollutants transported to the indoors. The diversity of pollutants, pollutant sources, and the resulting health effects challenge the comparison of the impacts of different control strategies on energy consumption and indoor air quality. To address this challenge, this work presents a quantitative framework for reaching the optimal energy cost for the maximum achieved exposure benefits, specifically for retail buildings and their understudied energy, economic, and health risk influence. The main objectives of this dissertation are to 1) determine pollutants of concern in retail buildings that contribute the greatest to the burden of disease, and 2) determine energy-efficient, exposure-based control strategies for different retail types and locations. The research in this dissertation is divided into four specific aims that fulfill these two objectives.

The first specific aim (Specific aim 1.a) addresses Objective 1 by applying available disease impact models on pollutant concentrations taken from 15 literature studies (150 stores, a total of 34 pollutants). Of those pollutants, there was little data reported on particulate matter (PM) concentrations and none on emission rates for PM, limiting our understanding of exposure to this pollutant. The second specific aim (Specific aim 1.b) also addresses Objective 1 by characterizing particulate matter (PM) concentrations, emission rates, and fate of ambient and indoor-generated particles in retail buildings. The tasks of this specific aim consisted of particulate matter and ventilation measurements in 14 retail buildings. Among the findings of Objective 1, $PM_{2.5}$ and acrolein are the main contaminants of concern for which control methods should be prioritized, contributing to 160 disability-adjusted life years (DALYs; years lost due to premature mortality and disability) per 100,000 persons annually. Employees in grocery stores mainly drove this burden. An efficient indoor exposure reduction strategy should take into account all mechanisms that influence pollutant concentrations: indoor and outdoor sources (highlighting the importance of retail type and location), infiltration, ventilation, and filtration.

The remaining specific aims address Objective 2 by investigating the energy and air quality impact of two commonly used exposure control scenarios, ventilation (Specific aim 2.a) and filtration (Specific aim 2.b). The tasks of Specific aim 2.a consisted of modeling the impact of multiple ventilation strategies on contaminants of concern for six major U.S. cities and two retail types. The tasks for Specific aim 2.b consisted of conducting field measurements on 15 rooftop units to determine the fan energy impacts of filter pressure drop. These results are used in combination with a large dataset of 75 filters commonly installed in commercial buildings to estimate the energy consequences of filtration. Results for Objective 2 are presented from the quantitative

comparison of the impact on energy usage and DALYs lost of three main approaches: (1) adjusting ventilation only; (2) adjusting filtration only; and (3) adjusting ventilation and filtration together. All approaches were able to provide substantial reductions in the health risks (19-26% decrease in DALYs lost); the magnitude of the reductions depended on the ventilation/filtration scenario, the retail type, and the city. The magnitude of energy cost to achieve the maximum health benefits depended on the city and the retail type (for example for a 10,000 m^2 grocery store, the energy cost ranged from $1,100 for the annual cost of filtration energy in Los Angeles to $24,000 for the annual cost of ventilation in Austin). The uncertainties of the estimates driving these findings are discussed throughout the results section. The finding that emerges from this analysis is the pollutant exposure control ventilation (PECV) strategy. This strategy is superior to the ventilation rate procedure (VRP; ASHRAE Standard 62.1-2010) and the indoor air quality procedure (IAQP; ASHRAE Standard 62.1-2010) as it decides on a range of ventilation rates by weighing the exposures of contaminants of concern found in retail buildings. Then, among the range of ventilation rates identified, the PECV recommends the optimal ventilation rate that leads to energy usage savings in the climate considered.

Overall, the work presented here prioritizes specific contaminants of concern in retail buildings and proposes an exposure-based, energy-efficient control strategy for different retail types and locations. Policy makers, engineers, and building owners can use these results to decide amongst appropriate control strategies that will lead to minimum energy consumption and, at the same time, will not compromise occupant health. This work can be repeated for different types of buildings, notably for residences, schools, and offices where abundant information is available on both pollutant concentrations and ventilation rates, but where information is lacking on how to optimize the control strategies for better indoor air quality.

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Chapter 1: Introduction and Motivation

Commercial buildings represent 46% of building energy consumption and 19% of U.S. energy consumption. Among commercial buildings, retail buildings account for the largest energy costs, rank second in greenhouse gas emissions, and have the most immediate opportunity to realize energy efficiency savings (Zhang et al., 2011, Kennedy et al., 2012). Within retail stores, HVAC systems have been identified as one of the largest energy users, accounting for 28% to 48% of total building energy use (Alhafi, 2012). In addition to energy concerns, the indoor air quality (IAQ) of retail buildings is an important occupational exposure consideration: the retail sector employs 15 million workers, approximately 10% of the U.S. workforce (NRF, 2010), and the average American above the age of fifteen spends 0.48 hours per day purchasing goods and groceries (ATUS, 2011). Balancing air quality concerns and energy usage in retail buildings is key to reducing energy consumption without compromising occupants' health, as well as minimizing negative sales impacts from poor perceived air quality.

Identifying the pollutants that contribute the greatest to the burden of disease, i.e., the contaminants of concern, is the first step to ensure efficient allocation of energy used by pollutant control strategies. These contaminants of concern (COCs) are not yet investigated for retail buildings. Identifying COCs upon the understanding of exposures to pollutants found in these buildings, which in turn depend upon the knowledge of pollutants concentrations. Among the pollutants found in retail buildings, $PM_{2.5}$ is of particular importance due to the strong association between exposure to $PM_{2.5}$ and serious health effects (e.g. lung cancer, cardiovascular morbidity and mortality, ischemic heart disease mortality, non-fatal myocardial infarction, heart rate variability and systemic inflammation; Pope et al., 2002; Pope & Dockery, 2006; Weichenthal et al., 2007; Brook

et al., 2010). Although short in duration, exposures to $PM_{2.5}$ occurring during time spent in retail buildings can contribute significantly to 24-h personal exposures (Chang et al., 2000; Rea et al., 2001). Rea et al. (2001) suggested that up to 35% of $PM_{2.5}$ exposure happens in microenvironments (primarily retail buildings, restaurants, and coffee shops) where people only spend 4-13% of their time. Despite its importance, knowledge of PM indoor concentrations, emissions, and sources in retail buildings is not complete, limiting our understanding of exposure to PM. This serves as the motivation for the first part of this dissertation.

Ventilation is commonly used to dilute indoor-generated pollutants. However, the effectiveness of different ventilation strategies in targeting the contaminants of concern are not well-known for retail buildings. Additionally, ventilation have a great impact on overall building energy consumption: just considering the retail sector, eliminating ventilation would decrease the total energy use index (i.e., building's energy use as a function of its size) by 8.4% on average, with the gas energy use index decreasing by 27.8% (Benne et al., 2009). As energy conservation in buildings became crucial, one area that received significant attention was determining the minimum ventilation rate that sufficiently maintains an acceptable indoor air quality (IAQ) in buildings. Wargocki et al. (2002), Seppänen et al. (2004), and Sundell et al. (2011) performed an exhaustive review of papers (n= 134) published in peer-reviewed scientific journals that provided sufficient information on both ventilation rates and health effects in non-industrial environments. These researchers found a positive correlation between increased ventilation rates and improved perceived air quality and/or decreased sick building syndromes. While perceived air quality is not a universal measure of adverse effects; nonetheless, it has likely an impact on retail sales (Chebat and Michon, 2003). What remains less investigated in the reviewed studies is the impact of outdoor air quality on indoor

exposures for different ventilation rates. This is important because ventilation can also bring ambient air pollution indoors (e.g., Hänninen et al., 2005). Therefore, the determination of the optimal ventilation rate is yet to be fully characterized and understood.

Another common pollutant control mechanism is the use of high-efficiency heating ventilation and air-conditioning (HVAC) filters to decrease particulate matter concentrations (PM) from indoor or outdoor origins. Similar to ventilation, filtration is also tied to energy consumption: high-efficiency HVAC filters generally have higher-pressure drops and are widely presumed to have large energy penalties. The magnitudes and even the signs of the energy impacts of high filter efficiencies are not known for commercial buildings. It is important to assess these impacts because fan energy necessary to move the air throughout commercial buildings equipped with rooftop units accounts for at least 7% of total site energy consumption (261 trillion BTUs annually; EIA, 2003; DOE, 2012).

To address these knowledge gaps, my dissertation explores the entire chain of contaminant concentrations, the impacts of exposures, and the energy consequences of different approaches to reducing these exposures in retail buildings. The work in this dissertation is grouped to fulfill two objectives:

- Determine contaminants of concern in retail buildings for which control methods should be prioritized.
- Determine the optimal combination of ventilation rate and filtration efficiency that will warrant a decreased exposure to contaminants of concern with minimal impact on energy usage in different retail types and locations.

This dissertation is organized into two major sections: (1) a Research Summary, which includes research objectives (Chapter 2), a literature review (Chapter 3), a

summary of methods (Chapter 4), a summary of findings (Chapter 5), and a conclusion (Chapter 6); and (2) Appendices A through D, which include four full-length articles addressing the two main objectives.

Ultimately, this work provides new knowledge about contaminants of concern found in retail environments, viable removal processes, and their impact on indoor air quality and HVAC energy use in retail buildings.

Chapter 2: Summary of Research Objectives

The overall goal of the proposed work is to decrease exposure to contaminants of concern while achieving energy efficiency in retail buildings. In this dissertation, there are two research objectives with four associated specific aims, as illustrated in Figure 1.

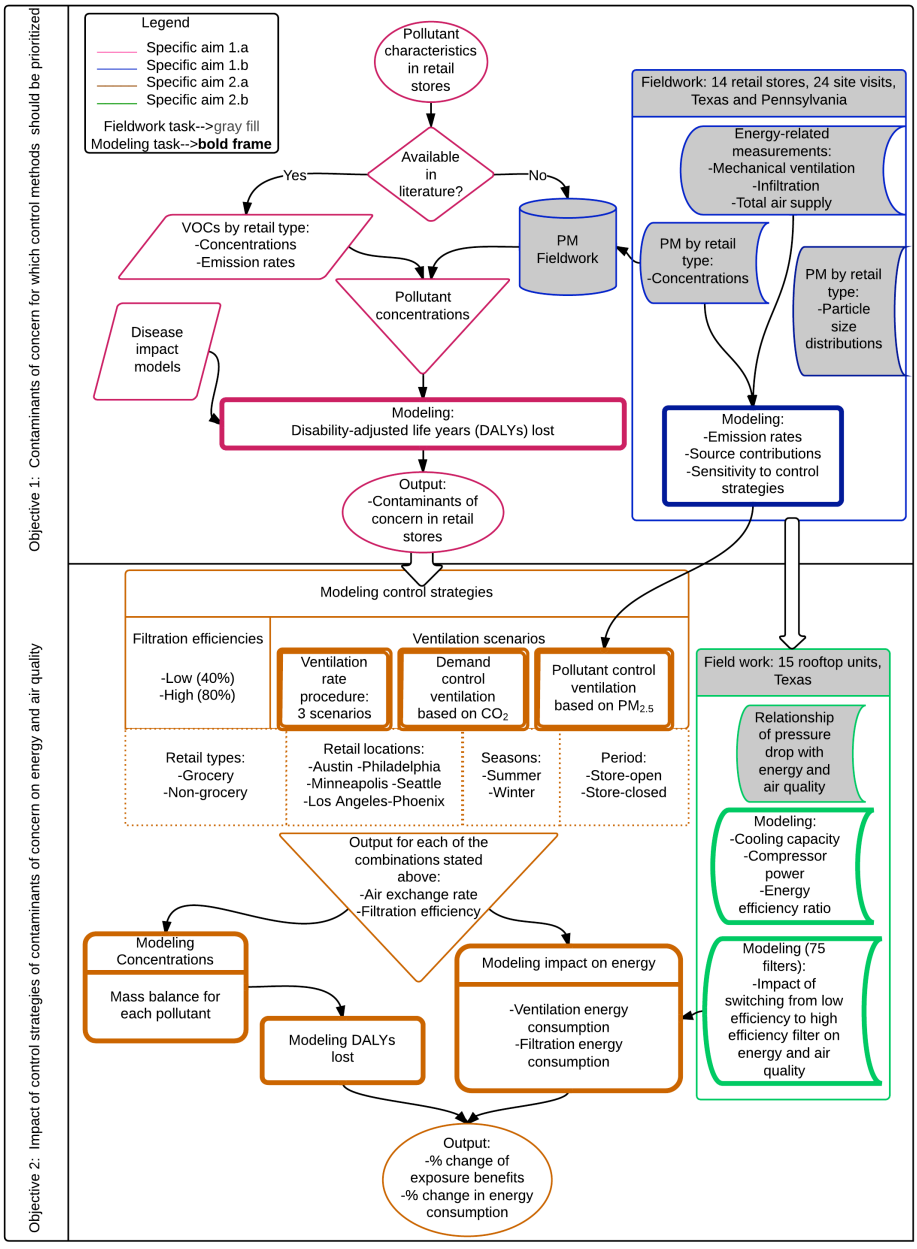

Figure 1: Schematic of research objectives.

The goal of the first objective is to determine the contaminants of concern in retail buildings for which control methods should be prioritized. This objective is divided into two specific aims (Specific aims 1.a and 1.b), described below.

2.1 SPECIFIC AIM 1.A EXPOSURE MODELING OF CONTAMINANTS OF CONCERN IN RETAIL BUILDINGS

Quantifying chronic health risks from inhalation exposures to different pollutants in retail environments is key to select contaminants of concern (COCs). The selection of COCs will allow prioritization of pollutant control approaches. Simply comparing the pollutants measured indoors to reference or regulatory levels is problematic. Published health standards or guidelines vary widely between different organizations (e.g., more than two orders of magnitude for formaldehyde exposure limit). Additionally, many pollutants such as volatile organic compounds (VOCs) do not have reference exposure levels yet. Thus, the ability to identify and compare harmful pollutants at existing indoor concentrations requires a common metric of harm such as disability-adjusted life year (DALY), expressing morbidity and mortality in one number. This metric is commonly used and is endorsed by the World Health Organization (WHO). No study has yet considered chronic health risks from exposure to pollutants in retail stores.

This specific aim quantified pollutant-specific impacts of 34 pollutants by using pollutant concentrations reported in 15 literature studies (150 stores) in Monte Carlo simulations to sample from uncertainties of parameters used in the available disease models.

Among the pollutants considered in this aim, there was little data available in the literature about particulate matter concentrations. Additional characterization of particles indoors is essential for accurate PM exposure assessments. Specific aim 1.b focuses on filling this gap.

2.2 SPECIFIC AIM 1.B PARTICLE CHARACTERIZATION IN RETAIL ENVIRONMENTS

Retail stores are important sites of exposure to PM, yet few existing studies provide useful data about particle concentrations. Quantifying exposure to particles and developing exposure mitigation strategies is not yet possible because the fate of ambient and indoor-generated particles and the effectiveness of removal mechanisms are not yet well understood for retail stores. Specific aim 1.b completes the understanding of particle sources and losses in retail stores by conducting simultaneous measurements of particulate matter concentrations and particle size distributions both indoors and outdoors, coupled with ventilation measurements, in 14 retail stores (a total of 24 site visits) located in Austin, TX and State College, PA.

The results of this specific aim strengthen the knowledge base of PM concentrations and indoor-to-outdoor ratios of different particle sizes by retail type. Also, this specific aim introduced distributions of indoor emission rates for PM, which have not yet been reported in the literature. Emission rates were calculated through a parametric analysis taking into account the uncertainties of parameters involved when applying a mass balance analysis of particle mass concentrations. This specific aim also explored the importance of indoor versus outdoor sources, presented potential indoor sources by retail type, and investigated the impact of changing ventilation rates by experiment and by modeling, and the impact of changing infiltration rates and filtration efficiency by modeling.

Specific aim 1.a uses particles concentrations measured in specific aim 1.b when quantifying exposure to particles in retail buildings, as illustrated in Figure 1. The overall results presented in Objective 1 are the COCs found in retail buildings.

The goal of the second objective is to evaluate the energy and exposure consequences of different control strategies for the contaminants of concern determined

from Objective 1. Objective 2 is divided into two specific aims (Specific aims 2.a and 2.b), described below.

2.3 SPECIFIC AIM 2.A ASSESSMENT OF VENTILATION AS A METHOD TO CONTROL CONTAMINANTS OF CONCERN FOUND IN RETAIL ENVIRONMENTS

Ventilation is mainly used to control the quality of indoor environment, including thermal and humidity conditions, as well as odor levels. Ventilation used for contaminant dilution can play a two-sided role with respect to indoor air quality. The positive role is the dilution of emissions from indoor sources, and the negative role is the transport of ambient pollution indoors. IAQ in buildings is prescribed by existing minimum standards for ventilation and the result is far too often suboptimal from energy and/or air quality perspectives. There is no study to date that has evaluated the energy and exposure consequences of different ventilation strategies prescribed in standards or industry practices.

The results of this specific aim provide a quantitative approach to reduce exposure to contaminants of concern and energy consumption in retail buildings. Best estimates for distributions of inputs across the retail sector were modeled in Monte Carlo simulations to find pollutant concentrations for multiple combinations of cities, seasons, store types, and time of day. The resultant pollutant concentrations were then used in Monte Carlo simulations of DALY to determine the exposure impact of different ventilation scenarios. The calculated air exchange rate from each ventilation scenario was used in an energy model to determine the energy impact.

Ventilation strategies presented in this specific aim (Specific aim 2.a) represents the first exposure control scenario. In addition to ventilation, Objective 2 also incorporated PM filtration as a second exposure scenario: the energy and exposure consequences calculations of the different ventilation scenarios were repeated for both

low and high levels of PM filter efficiency. However, the energy and air quality consequences of different efficiency filters are not well-known for commercial buildings. Specific aim 2.b focuses on fulfilling this gap, and the data from this aim are used as a model input for Specific aim 2.a, as illustrated in Figure 1.

2.4 SPECIFIC AIM 2.B ENERGY AND IAQ CONSEQUENCES ASSOCIATED WITH HIGH FILTER EFFICIENCY INSTALLED IN COMMERCIAL SYSTEMS

High efficiency filters are commonly used to decrease exposure to particles and, unlike ventilation, they are effective even when outdoor concentrations of PM are high. However, improved filtration is thought to increase energy consumption by increasing resistance to airflow. Theoretically, increasing the pressure drop may increase or decrease the fan power; the magnitude and the direction of change depend on the fan control mode (i.e., with or without fan speed control) as well as the fan performance curve. Only a limited number of studies have explored the effect of high-pressure drop filters on fan and compressor power in real buildings and research in this area is especially needed for mid-sized and large commercial buildings.

The results from this specific aim provide a comprehensive understanding of the impact of better filtration by quantifying the relationship between filter pressure drop and system parameters (fan pressure, airflow rate, fan power, and system efficiency) through fieldwork conducted on 15 rooftop units equipped with and without speed control installed in retail buildings. The results of the field measurements were used as input for the vapor compression model applied to characterize the impact of high efficiency filters on cooling capacity and compressor power. In addition, the results of both fieldwork and modeling were extended in the context of comparing the energy and air quality performance of a large dataset of 75 filters divided into four different filtration efficiencies.

The results presented in this specific aim (Specific aim 2.b) are combined with Specific aim 2.a to calculate: (1) the overall energy cost of ventilation, coupled with cost of low- or high-efficiency PM filters, and (2) the exposure benefits translated from this energy cost.

This dissertation takes an integrated approach with consideration to energy, and indoor air quality, and includes all contaminants of concern found in retail environments. Specific aims 1.a, 1.b, 2.a, and 2.b improve our understanding of how contaminants of concern concentrations, exposures, and energy consumption are related. The comprehensive fieldwork output resulting from Specific aims 1.a and 2.b also provide a valuable database that can be used as input in other studies.

The complete methodology description and results of these specific aims are included in Appendices A through D, which include four full-length journal articles. The titles of these papers are as follows:

Appendix A: Paper 0: "Ventilation and Indoor Air Quality in Retail Stores: A Critical Review (RP-1596)". This paper was submitted to ASHRAE *HVAC&R Research*. A group of researchers contributed to this paper and I was the first author.

Appendix B: Paper 1: "Particle Characterization in Retail Environments: Concentrations, Sources, and Removal Mechanisms". This paper was submitted to *Indoor Air*.

Appendix C: Paper 2: "The Relationship between Filter Pressure Drop, Indoor Air Quality, and Energy Consumption in Rooftop HVAC Units". This paper was submitted to *Building and Environment*.

Appendix D: Paper 3: "Impact of ventilation and filtration strategies on energy consumption and exposures in retail stores". This paper will be submitted to ASHRAE *HVAC&R Research*.

Chapter 3: Summary of Literature Review

The purpose of this chapter is to distinguish the present study from the relevant literature. The chapter is divided into three sections. Section 3.1 reviews research on chronic health risk estimates. Section 3.2 then provides a summary of the investigations that measured and linked concentrations of particulate matter to building characteristics, ventilation, and occupant satisfaction specifically in retail environments. For brevity, other pollutants found in retail environments will not be described in this section: Paper 0 (Appendix A) provides a critical review of these pollutants. Lastly, Section 3.3 discusses the available methods of pollutant control (i.e., ventilation and filtration) and the effect of these methods on IAQ and energy consumption in retail environments.

3.1 CHRONIC HEALTH RISK ESTIMATION

Epidemiological and toxicological research has contributed to the development of metrics that link the intake of pollutants to disease incidences, and disease incidences to health costs in disability-adjusted life years (DALYs). The DALY metric measures health gaps as opposed to health expectancies (ideal health scenario). The main strength of the DALY metric is that it combines the fatal and non-fatal health outcomes into a single value (i.e. years of life lost due to death and disability). Multiple studies provided health impact assessments for different pollutants found indoors using the DALY metric. Huijbregts et al. (2005) presented a method to derive human damage and effect factors, and provided input for 1,192 toxic pollutants typically found indoors (i.e., VOCs, aldehydes, and ozone). These damage factors are expressed as DALYs for cancer and non-cancer effects. Lvovsky et al. (2000) studied DALYs lost by different outcomes for CO, NO_2, O_3, and SO_2. Pope (2002), Lvovsky et al. (2000), Pope et al. (2002; 2009), Hong et al. (2010), and Melse et al. (2010) studied DALYs lost by death, chronic

bronchitis, and nonfatal stroke from exposure to $PM_{2.5}$. Burnett et al. (1999) developed a concentration response function for hospital admissions associated with long term $PM_{2.5}$ exposure. However, its impact is negligible compared to the impact of mortality, chronic bronchitis and stroke. Logue et al. (2012) demonstrated the use of an approach to synthesize all the available disease incidences and disease impact models to estimate the chronic health impacts from exposure to indoor pollutants.

Even though the most recent scientific knowledge and data were used in estimating the DALYs lost, many uncertainties and controversies remain. Comparing the DALYs lost due to exposure to different pollutants gives only a crude ranking of environmental health impacts and need to be interpreted with caution. The following limitations must be taken into account when interpreting the DALYs lost results:

- The DALY model takes into account only health impacts for which sufficient evidence exists in a quantitative format. This means that the total DALYs lost do not include burden of disease for which as yet incomplete or only qualitative evidence exists. For example, there is evidence in the literature that exposure to $PM_{2.5}$ is associated with decreasing lung function and diabetes (Reis et al., 2009; Pearson et al., 2010; da Silva et al., 2012). However, more epidemiological studies are needed to quantify the correlation of incidences with these diseases. Also, there is a there is growing concern that some semi-volatile organic compounds and ultrafine particles may cause substantial adverse health effects at typical environmental levels. However, there is currently insufficient toxicological data to quantify that impact (e.g., ultrafine particles; Crüts et al., 2008; Knol et al., 2009).

- The DALY model allows for the estimation of the total national impacts, but does not allow for identifying gender differences or health impacts in specific population groups, e.g. highly exposed or those especially vulnerable due to a lower background health status.
- The DALY inputs regarding the years lost due to disease severity are subject to biases from self-reported health data (Koltjob et al., 2005). More validation related to the disease severity is warranted.
- For pollutants considered in this work other than $PM_{2.5}$, their health impact is studied based on animal toxicity literature rather than on epidemiological concentration-response (C-R) relationships. Animal toxicity data requires interspecies extrapolations that generally involve larger uncertainties than the epidemiologically based C-R functions. This will result in uncertainty bounds larger than one order of magnitude and ultimately an overlap between the confidence intervals of many of the pollutants considered.
- Currently, epidemiological studies upon which the $PM_{2.5}$ concentration-response (C-R) relationships are quantified are based solely on ambient epidemiology: a small number of monitoring stations can be used to assess the exposures of a large population. However, this is not the case for indoor epidemiological studies where substantial challenges arise due to the diversity of indoor spaces and the targeted population. The implication of using ambient rather than indoor $PM_{2.5}$ concentration-response relationships is further discussed in the results section (Section 5.2.1).
- The identified methods (either based on epidemiology or animal toxicity data) are accepted health impact models; they are, nevertheless, simplifications of population wide responses to chronic inhalation exposure. These methods rely on

non-threshold models (i.e., there is no concentration below which adverse health consequences do not happen). While this is true for $PM_{2.5}$ (Schwartz et al. 2002), it is not true for some VOCs (e.g., Salthammer and Bahadir 2009). Instead of only using the mean of the concentration distribution in the model, existence of a threshold requires the knowledge of the shape of the population intake distribution, an area with very little to no information. Multiple threshold values have been suggested in the literature, although these thresholds vary widely. For example, reported formaldehyde threshold levels vary from 120 $\mu g/m^3$ (German Federal Institute for Risk Assessment; WHO) to 12 $\mu g/m^3$ (Naya & Nakanishi, 2005). Less threshold information is available for other pollutants (e.g., acrolein). Considering no threshold will probably overestimate DALYs lost for VOCs, and this point is further addressed in the results section (Section 5.2.1).

Overall, when interpreting DALYs lost, the methodological complexities and underlying assumptions need to be kept in mind. Despite the large uncertainties, the DALY metric is regarded as the most practical population health status measure. This metric is commonly used to identify contaminants of concern in buildings (e.g., Logue et al., 2012; Hänninen et al., 2005). As mentioned previously, no study has yet identified the contaminants of concern in retail buildings. Therefore, there remains a need to use the DALY as a metric to rank exposures to pollutant and evaluate the contaminants of concern found in retail environments.

3.2 INDOOR PARTICLE DYNAMICS IN RETAIL ENVIRONMENTS

The ability to link PM indoor concentrations to personal exposure in retail stores depends upon understanding the indoor fate of ambient and indoor-generated particles. Only a limited number of researchers have investigated particle concentrations in retail

spaces (Grimsrud et al., 1999; Li et al., 2000; Rea et al., 2001; Sakai et al., 2002; Liu et al., 2004; Lee & Hsu, 2007; Bennett et al., 2011; Brown et al., 2012; Chan et al., 2012; Dong et al., 2012). Only two studies have reported measurements in big box retail stores (Grimsrud et al., 1999; Liu et al., 2004), and only three of the existing studies (Lee & Hsu, 2007; Bennett et al., 2011; Chan et al., 2012) have reported the ventilation rates of the retail spaces that were tested. However, there is no broad study that has evaluated the relationship between particle concentrations, air exchange rate, or with different type of retail stores and other building factors. Additionally, no previous studies are available that included the indoor emission rates for particles in retail environments. Therefore, predicting the effectiveness of control mechanisms in different types of retail stores is not feasible.

Despite these limitations, there are useful data on particle concentrations in retail environments. PM concentrations reported in the literature are summarized in Figure 2. The solid horizontal line represents the median PM_{10}, $PM_{2.5}$ and submicron particle concentrations from all studies. The dotted line represents the average PM reported in all studies, and the dashed line represents the average taken only from the U.S. studies.

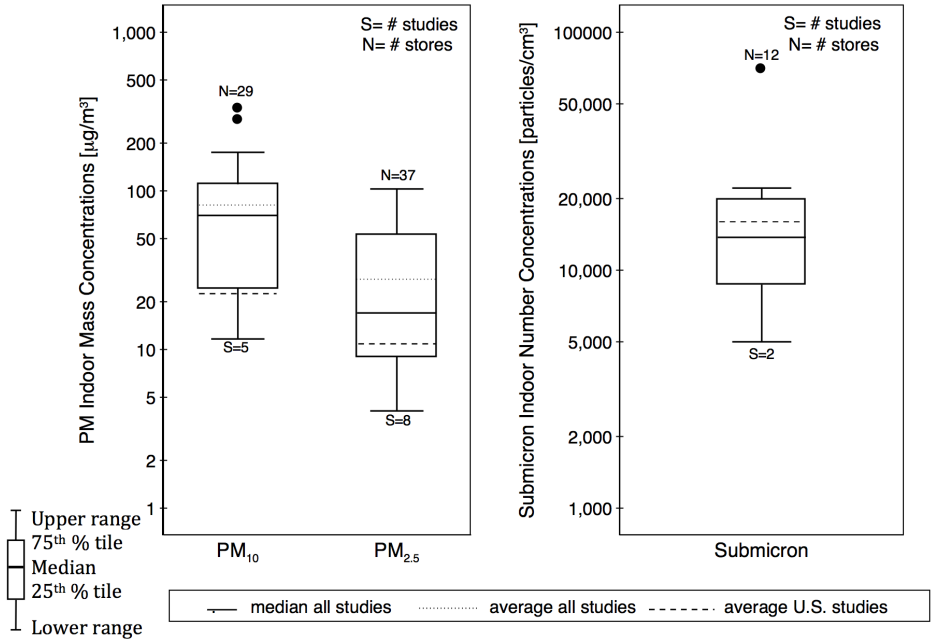

Figure 2: Summary of PM_{10} and $PM_{2.5}$ mass concentrations and submicron particle number concentrations. Data for PM_{10} are from Li et al., 2000; Sakai et al., 2002; Liu et al., 2004; Bennett et al., 2011; and Chan et al., 2012. Data for $PM_{2.5}$ are from Grimsrud et al., 1999; Rea et al., 2001; Liu et al., 2004; Lee & Hsu, 2007; Bennett et al., 2011; Brown et al., 2012; Chan et al., 2012; and Dong et al., 2013. Data for submicron particles are from Bennett et al., 2011; and Brown et al., 2012.

PM_{10} and $PM_{2.5}$ concentrations. Median PM_{10}, as shown in Figure 2, is dominated by studies that were conducted in Hong Kong, Taiwan, Japan, and China, where outdoor air is an important contributor to indoor PM_{10} (Li et al., 2001; Lee & Hsu 2007; Dong et al., 2013). Bennett et al. (2011) and Chan et al. (2012) are the only studies that have reported PM_{10} concentrations in U.S. retail stores. These two studies reported an average PM_{10} mass concentration for 15 retail stores of 24.3 µg/m^3. Using data from Grimsrud et al. (1999), Rea et al. (2001), Bennett et al. (2011), Brown et al. (2012), and Chan et al. (2012,), the average $PM_{2.5}$ mass concentration calculated for U.S. retail sites was 11.6 µg/m^3. Approximately half of all the stores tested had an indoor-to-outdoor

(I/O) ratio larger or equal to one and the rest had an I/O ratio less than unity. This finding suggests that indoor PM_{10} and $PM_{2.5}$ can originate indoors, outdoors, or both. The reported indoor sources include cleaning (Grimsrud et al., 1999), cooking (Li et al., 2001; Bennett et al., 2011; Brown et al., 2012), and illegal smoking (Li et al., 2001). Additionally, there was evidence that grocery stores exhibit higher particle concentrations than other types of stores: $PM_{2.5}$ mean concentration recorded in U.S. grocery stores was 31.7 $\mu g/m^3$ with a maximum concentration of 90 $\mu g/m^3$ (Bennett et al., 2011; Brown et al., 2012; Chan et al., 2012). The main contributor to elevated concentrations in these grocery stores was cooking events. Besides grocery stores, the data available for U.S. sites are insufficient to draw conclusions about specific indoor particle sources for different store types.

Submicron particles concentrations. Figure 2 contains a graph showing the submicron particle number concentrations reported by two studies conducted in the U.S. (Bennett et al., 2011; Brown et al., 2012). The average submicron concentration was 17,000 $\#/cm^3$. The submicron concentration measured in these studies is a good proxy for ultrafine particles because the particle number concentration is typically dominated by small particles (Apte et al., 2011). Another non-U.S. study reported only the highest ultrafine number concentrations observed during photocopying of 10^8 particles/cm^3 (Lee & Hsu 2007); thus the number reported in Lee and Hsu (2007) is not included in the figure. These investigations associated submicron particle number concentrations with in-store cooking, cleaning, photocopying, and specific activities (e.g., presence of silk screening area) (Lee & Hsu 2007; Bennett et al., 2011; Brown et al., 2012).

Generally, PM concentrations found in retail stores can be significant and may pose health concerns. However, little is known about the factors that affect these PM indoor concentrations. There still remains a need to characterize PM concentrations,

source strengths, contribution of indoor versus outdoor sources, and the effectiveness of removal mechanisms.

3.3 POLLUTANT CONTROL STRATEGIES

HVAC systems in retail establishments generally are equipped with mechanical ventilation to remove indoor contaminants by dilution with outdoor air. Also, both outdoor air and recirculated air pass through filters located upstream of the cooling coils to remove particulate matter contaminants from both indoor and outdoor environments. Both ventilation and filtration are potential pollutants removal processes and they present concerns related to energy consumption. The next two sections highlight gaps in the literature that limit our understanding of the compromise between energy and indoor air quality for pollutant removal processes, first for ventilation, then for filtration.

3.3.1 Ventilation

Over the past two decades, a lot of effort has been exerted to find the minimum ventilation rates that will reduce energy consumption while maintaining an acceptable indoor air quality. ASHRAE Standard 62.1-2010 provides two alternative procedures for selecting the minimum ventilation rate for commercial buildings: 1) a prescriptive approach: the ventilation rate procedure (VRP); 2) a performance-based approach: the indoor air quality procedure (IAQP).

The VRP is the more widely-used procedure. The prescribed minimum ventilation rates are the sum of two quantities: the minimum rate of outdoor air supply per unit floor area, and the minimum rate of outdoor air supply per occupant. The VRP is assumed to maintain an acceptable indoor air quality as perceived by at least 80% of occupants. Bluyssen et al. (1996) tested 44 buildings with average ventilation rates of 25 L/s (far above the current ventilation rates specified by the VRP) and found that air quality in

64% of these buildings did not satisfy 80% of the occupants. In addition, a review of ventilation measurements in retail stores found that half of the stores tested met or exceeded the VRP; nonetheless, these ventilation rates were not sufficient to keep all pollutants below their most conservative limits (see Appendix A Paper 0 for details). Specifically, there is no documentation of the adequacy of VRP in maintaining an acceptable indoor air quality in retail buildings.

One variation to the VRP is the use of demand control ventilation (DCV), often based on CO_2 concentrations in buildings with variable occupancy, to further save energy. The impact of DCV on indoor air quality remains less investigated. Only eight literature studies investigated whether controlling ventilation by measuring occupancy (CO_2-based DCV) could keep pollutants (e.g., formaldehyde, TVOC, radon) below their reference or regulatory limits (Gabel et al., 1986; Donnini et al., 1991; Carpenter, 1996; Enermodal, 1995; Persily et al., 2003; Chao & Hu, 2004; Jeong et al., 2010; Mui & Chan, 2006; Herberger & Ulmer 2012). Five of these studies found that DCV was not sufficient to control the measured pollutants below their established limits; additional studies would have found that DCV did not control these pollutants if they considered the current reference exposure or standard limits for these pollutants. DCV-based CO_2 does not control outdoor-generated pollutants, nor does it account for pollutants generated indoors but independently of human activities. Thus, the ability of VRP or DCV to maintain an acceptable IAQ in buildings depends highly on the source strengths, pollutant sources, and infiltration rates, which are specific to building type and location.

Another approach to control ventilation is to follow the performance-based approach specified in ASHRAE Standard 62.1-2010, where the minimum ventilation rate is defined based on contaminants of concern's (COCs) emission rates and concentration limits. However, according to Mendell and Apte (2010), given the limitations of the

current performance-based procedure, implementation of this standard for big box retail stores would not save energy while providing reliably an acceptable IAQ. The main limitation is the knowledge of source strengths and the understanding of how different sources of emissions should be added together.

In summary, the appropriate ventilation strategy that will minimize energy consumption and health burden (i.e., number of DALYs lost) is not identified yet in retail buildings. Specifically, there remains a need to compare the impacts of VRP, DCV-based CO_2, and IAQP on pollutants of concern, whether generated indoors, outdoors, or both, and observe the energy consumption in different retail types and locations.

3.3.2 Filtration

Filtration is another important PM control strategy and may be less energy intensive than ventilation. Quantifying the relationship between filter pressure drop, indoor air quality, and energy cost of air filters has been the focus of a number of studies. Field, lab, and computer simulation work have mostly focused on small systems (<30 kW) with no fan speed control (Palani et al., 1992; Rodriguez et al., 1996; Parker et al., 1997; Yang et al., 2006; Stephens et al., 2010[a&b]; Noh & Hwang, 2010; Proctor et al., 2011; Nassif et al., 2012). Only one recent field study tested small systems equipped with and without fan speed control in residences (Walker et al., 2013). Several of those studies are summarized in Figure 3. The horizontal line represents the median change in total power, cooling capacity, and energy efficiency ratio for HVAC systems equipped with no fan speed control.

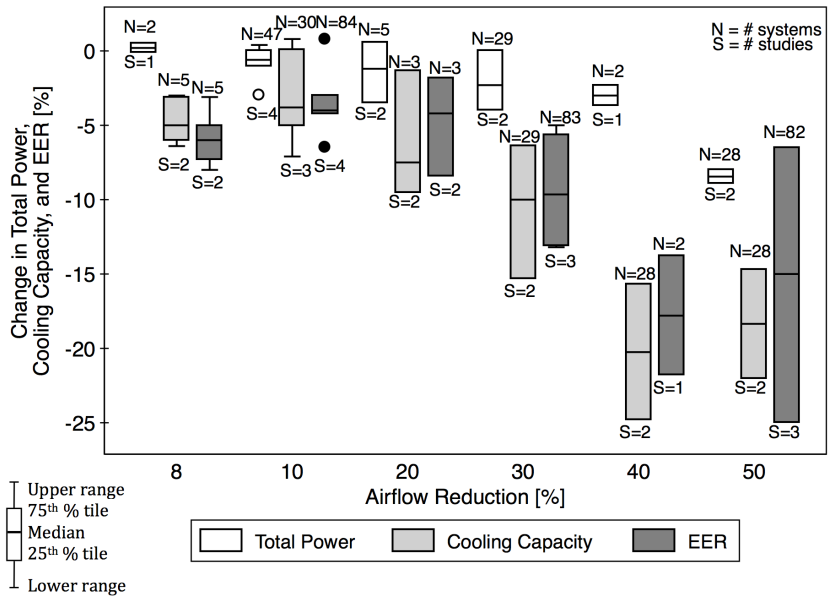

Figure 3: Summary of the effect of airflow reduction on total power, cooling capacity, and energy efficiency ratio (EER) for HVAC systems equipped with no fan speed control. Data are from Palani et al., 1992; Rodriguez et al., 1996; Parker et al., 1997; Yang et al., 2006; Stephens et al., 2010a&b; Proctor et al., 2011.

Figure 3 shows that the consequences of a high pressure-drop filter installed in residential and light commercial systems with no fan speed control are likely reduction in airflow, total power, and system efficiency. The magnitudes of these reductions vary between studies as they depended on the overall studied system characteristics (i.e. fan and system performance curves). As for energy consumption, the sign of the change depended on the system runtime and the magnitude of decrease of system total power.

Three small-scale studies of large commercial systems investigated the effect of reducing the pressure drop across filters on energy consumption (Chimack & Sellers, 2000; Lam et al., 2006; Nassif, 2012). Of those studies, however, only Lam et al. (2006) performed field measurements. They conducted a field study on a 40-floor building with a centralized HVAC system; however, the results from this study are difficult to apply to

most of buildings in the U.S. that are smaller because the filter pressure drop on the studied 40-floor building is negligible when compared to the high static pressure drop of the entire ducting system of this building. Through modeling, Chimack and Sellers (2000) and Nassif (2012) found that reducing the average static pressure drop across filters 1.5 times reduces the cooling energy consumption by 21% over 40 weeks.

Other studies have focused on determining an approximate cost of filter operation (life cost analysis) by applying a filter model (Fisk et al., 2002; Arnold et al., 2005; Bekö et al., 2008; Mayer et al., 2008; Sun & Woodman, 2009; Sun, 2010; Montgomery et al., 2012). In these studies, the filter models used assumed the airflow rate through the filter to be constant over the life of the filter and, therefore, the impact on power for systems with and without fan speed control was not captured. In addition, the actual system operating point of the fan and duct curves and the resulting system efficiency can significantly impact the fan power consumption. Fan efficiency has also been shown to vary widely with values that range from 22-49% for residential and light-commercial buildings (e.g., Stephens et al., 2011) and 30-80% for modeled commercial systems (Chimack & Sellers, 2000; Fisk et al., 2002; Yang et al., 2006; Bekö et al., 2008).

In summary, there is lack of measured data and analysis of energy and indoor air quality consequences of filters used in big commercial systems (rated cooling capacity >30 kW), equipped with or without fan speed control. There still remains a need to quantify the energy and air quality performance of different filtration efficiencies installed in commercial systems.

Chapter 4: Summary of Methods

The purpose of this chapter is to summarize the methods used in this dissertation. A detailed description of these methods is presented in the papers and the supporting information appended at the end of this dissertation. The investigation methods are grouped in two major sections: 1) data collection and 2) modeling approaches, both described below.

4.1 DATA COLLECTION

Data collection included PM sampling, HVAC characterization, and impact of pressure drop on HVAC system parameters.

4.1.1 PM sampling

PM sampling included measurements of indoor and outdoor mass concentrations of PM_{10} and $PM_{2.5}$, number concentrations of submicron particles (0.02–1 μm), and PM number concentrations in six different size bins ranging from 0.3 μm to >10 μm in a total of 14 retail buildings (24 site visits, some sites visited multiple times per year) in Texas and Pennsylvania, monitored between April 2011 and July 2012. Table 1 describes the sample in more detail. The first column of Table 1 is a unique site identifier where the first letter is the store type (H = home improvement, M = general merchandise, F = furniture, E = electronics, O = office supplies, G = mid-sized grocery with floor area >2,000 m^2, and S = small grocery with floor area <2,000 m^2), the second letter is a brand identifier, and the third character indicates whether the site was located in Pennsylvania (P) or Texas (T). In the result section, for those stores that were visited more than once, a number is added to the site identifier to differentiate between visits to the same site. The maximum number of register transactions, often used with a multiplier as a proxy for

store occupancy, was obtained from the store records for each given visit (Column 5 of Table 1). The last column summarizes potential outdoor sources based on the store location.

Table 1: Characteristics of the retail sites visited.

Site	Months tested	Floor area [m^2]	Volume [m^3]	Max Transactions [#/h]	Potential Outdoor Sources
HaP	May Nov	11,700	93,400	207-380	Heavy traffic, near highway
HaT	Jun	12,400	91,800	265	Heavy traffic, near railway
MbP	Sep Jan May Jul	16,600	99,500	474-605	Rural area, near highway
MbT	Jul Oct Feb Apr	8,740	61,200	285-521	Heavy traffic, in a strip-mall
MiP[1]	Apr	11,300	66,800	400	Heavy traffic, in a strip-mall
MiT[1]	Apr	12,100	55,200	560	Heavy traffic, next to a highway
EgP	Feb Jun	2,700	20,300	141-157	Moderate traffic, next to bus stops
OhT	May	2,280	20,700	76	Heavy traffic
FfP	Mar	1,970	8,170	38	Moderate traffic, in a strip-mall
FfT	Oct Feb	3,250	19,800	48-55	Heavy traffic, near highway
GeP	Jul	3,100	25,300	370	Moderate traffic, next to a railway
GeT	Sep Jan	3,250	14,900	260-467	Heavy traffic, in a strip-mall
ScP	Aug	836	3,340	151	Moderate traffic, next to bars/restaurants
SdT	Aug	1,180	5,390	157[2]	Heavy traffic, entrance next to bus stop

[1] Indicates intervention sites tested for two consecutive weeks, one week at the normal ventilation rate and another week at the maximum ventilation rate allowed by the store.
[2] Indicates people count rather than number of transactions.

In a site, test experiments typically took five workdays and data collection activities included fixed and mobile sampling. For mobile sampling, portable particle instruments were carried in baskets or shopping carts and moved around in the store in order to collect more representative spatial data within the retail space. The data presented in the results section includes only results about $PM_{2.5}$ mass concentrations collected during mobile sampling. The methods and the results of other PM sizes can be found in Paper 1, Appendix B.

$PM_{2.5}$ real-time mass concentration was measured using TSI Sidepak and Met One Aerocet with a sampling duration of five minutes. A major effort in this work was spent

on calibrating these real-time instruments by collocating them with integrated particle measurements (gravimetric samplers and/or Thermo Scientific tapered element oscillating microbalance; TEOM) during each site visit to obtain calibration factors that corrected for the difference between the physical/optical properties of aerosols tested and the dust used in the calibration. A detailed description of the calibration procedure and the resulting calibration factors is presented in Paper 1 and its supporting information (Appendix B).

PM concentrations were used as inputs to (1) estimate exposure to PM in retail stores (Specific aim 1.a); (2) calculate PM emission rate distributions (Specific aim 1.b), used to test different control strategies (Specific aim 2.a). PM distributions were used as input to calculate integrated PM filter efficiency (Specific aims 1.b and 2.b).

4.1.2 HVAC characterization

HVAC characterization was accomplished simultaneously with PM sampling measurements described above on the 14 retail stores. All tested stores were conditioned and mechanically ventilated with rooftop air conditioning units (RTUs) that ranged in size from 3 to 55 tons. Total building ventilation rates (ACH) were measured using the SF_6 tracer decay method. SF_6 was released into different sections of the store to obtain a uniform concentration. After waiting approximately one hour (depending on the site), three field technicians simultaneously collected air samples every four to six minutes for a total of four hours with 500 mL plastic syringes at 15-18 sampling locations. In addition to the sample bags, QA/QC bags, including trip blanks and trip standard bags, were prepared and mixed in with the sample bags prior to analysis. All samples were analyzed using a Lagus Autotrac SF_6 analyzer, generally within 48 hours of collection.

System airflow rates of all rooftop units were measured at least once with an Energy Conservatory TrueFlow metering plate and DG-700 digital manometer. For the measurement, the TrueFlow plates were installed in place of the system filter (removed for measurement). To account for differences in pressure drop across the TrueFlow plates and the system filter, the static pressure was measured for both cases and used to adjust the supply flow, following the calculation procedure in the instrument manual.

Carbon dioxide concentrations were measured in the indoor space, RTU supply air streams, and outdoors using Telaire CO_2 sensors connected to a HOBO data logger or Q-Trak sensors.

Total building ventilation and supply airflow rates were used as inputs to calculate PM emission rate distributions (Specific aim 1.b). CO_2 concentrations were used to calculate mechanical ventilation rates, used as inputs to (1) calculate PM emission rate distributions (Specific aim 1.b), and (2) calculate integrated PM filter efficiency (Specific aims 1.a and 2.b).

The instruments used for each type of measurement, along with the manufacturer reported resolution and uncertainty are summarized in the Supporting Information of Paper 1 (Appendix B).

4.1.3 Impact of pressure drop on HVAC system parameters

From the sample of retail buildings in Austin, TX described above, 14 rooftop units (RTUs) equipped with no fan speed control and one unit equipped with fan speed control were selected for further testing. Cooling capacity ratings on these units ranged from 30 to 84 kW. For each sampled system, system airflow, filter and coil pressure drop, fan pressure rise, and power draw were measured for at least four different filter pressure drops (induced by blocking airflow at the filter). System airflow was measured as

explained in Section 4.1.2. An Onset HOBO Energy Logger was used to record the power draw of the air handler fan measured by a Continental Control Systems (CCS) Wattnode AC true power meter for approximately 30 minutes at 10-second intervals (±3% measurement uncertainty). The Energy Logger box was connected to voltage taps and 0 to 20 amp CCS current transducers. For the unit equipped with fan speed control, fan speed was adjusted manually using the variable frequency drive installed on the unit to maintain approximately the same airflow rate delivered by the fan for each measured filter pressure drop.

The relationships established between pressure drop and system parameters were used as inputs to assess the energy consequence of filtration (Specific aim 2.b), and used also as model input to assess impact of different filtration strategies on energy an exposures (Specific aim 2.a).

Two additional parameters, outdoor temperature and relative humidity, were also measured during fieldwork and were used as inputs for the vapor compression model (Specific aim 2.b).

4.2 MODELING APPROACHES

Five models are used in this dissertation. These models include the IAQ model, the DALY model, the system cooling capacity and compressor model, the filtration model, and the energy model, as described below.

4.2.1 IAQ model

Considering the retail space to be a single zone, the relationship between indoor and outdoor concentrations of particulate matter can be described by the time-averaged mass-balance shown in Equation 1, based on that provided by Riley et al. (2002) and adding an indoor source emission term.

$$C_{out}\left(\frac{Q_{OA}(1-\eta)}{V}+\frac{pQ_i}{V}\right)+E-C_{in}\left(\frac{Q_R\eta}{V}+\beta+\frac{Q_{EX}}{V}\right)=0 \qquad (1)$$

Where C_{out} is the outdoor and indoor concentrations [$\mu g/m^3$], Q_{OA} is the mechanical outdoor air airflow rate [m^3/h], p is the penetration factor of PM through leaks in building envelopes and major openings (dimensionless, ranging from 0 to 1), Q_i is the infiltration airflow rate [m^3/h], V is the volume of the store [m^3], E is the indoor emission rate [$\mu g/m^3 \cdot h$], Q_R is the recirculation airflow rate [m^3/h], η is the filter efficiency (dimensionless, ranging from 0 to 1), β is the first-order indoor loss rate of the pollutant by deposition to surfaces and/or reactions [per hour], and Q_{EX} is the exhaust airflow rate.

Applying this model assumes a well-mixed space and no phase change processes (gas-to-particle transformations and vice versa). The well-mixed assumption was confirmed by examining the SF_6 concentrations (and the resultant mean age of air) collected in different locations of the store as described in Section 4.1. The no gas-to-particle transformation assumption was verified by examining ozone and VOC concentrations (collected simultaneously with particle measurements). These concentrations were on average low at all sites – conditions that were not conducive to ozone chemistry, particularly for particle formation.

All the parameters in Equation 1 were measured or calculated from measured parameters, with the exception of filter efficiency, deposition rate, and penetration factor, which are not known for commercial buildings. Best-input parameters reported in the literature were used to determine efficiencies and deposition rates by particle size bin. Then, size-resolved efficiency and deposition rate for integrated particle sizes (e.g. $PM_{2.5}$) were calculated using Equation 2; where the efficiency or deposition for each particle

size was combined with indoor ($N_{in,i}$) and outdoor ($N_{out,i}$) particle size distribution, particle geometric mean (GM_i), and outdoor air fraction (OA). Particle size distributions were measured as part of Specific aim 1.b (using TSI Aerotraks). Outdoor air fraction was calculated by applying a mass balance of CO_2 concentrations measured as part of Specific aim 1.b, following the work of Persily et al., 1997.

$$\eta = OA \times \sum(\eta_i \times N_{out,i} \times GM_i^3) / \sum N_{out,i} \times GM_i^3 + (1 - OA) \times \sum(\eta_i \times N_{ind,i} \times GM_i^3) / \sum N_{ind,i} \times GM_i^3 \quad (2)$$

Particle penetration factor can be assumed to be equal to unity because particle infiltration in retail stores is dominated by large openings such as entrance, exit, and truck loading doors.

Note that a similar, more simplified, mass balance model exists for VOCs, described in Equation 3. For gas phase contaminants, depositional, filtration and penetration losses are negligible.

$$(C_{in} - C_{out})\frac{(Q_{OA} + Q_i)}{V} - E = 0 \quad (3)$$

Throughout this work, the IAQ model (Equations 1 and 3) was re-arranged to calculate PM emission rate distributions by retail type using a parametric analysis approach to sample from uncertainty distributions of the influencing parameters (Specific aim 1.b). In the parametric analysis, 13,122 samples by site were drawn from values that correspond to the mean and mean ± sd for each measured indoor and outdoor concentration, particle size distribution, infiltration, ventilation, and recirculation rates, as well as three different estimated filter efficiencies, three different deposition rates, and two surface to volume ratios. Figure 4 contains a summary of measured, calculated, and estimated parameters across the 24 sites. Additional simulation runs were completed to

test the importance of varying efficiency and deposition values on $PM_{2.5}$ emission rates, described in details in the Supporting Information of Paper 1 (Appendix B).

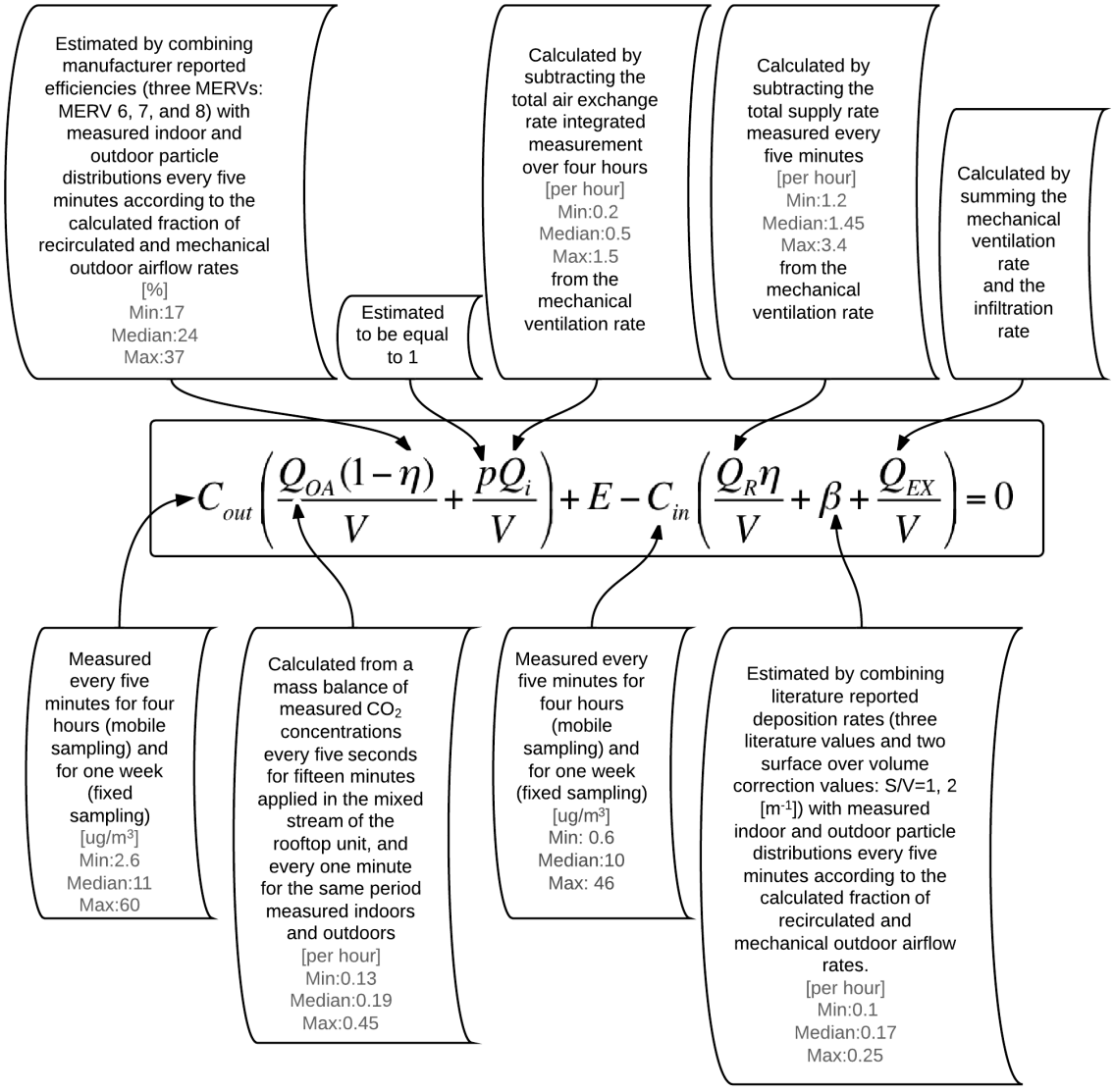

Figure 4: Schematic detailing the measured, calculated, and estimated parameters across the 24 sites.

In addition, the IAQ model was used to show how indoor pollutants concentrations change with different removal mechanisms (Specific aim 2.a) or to calculate the air exchange required to keep pollutants at a certain given concentration

(Specific aim 2.a). For the analysis conducted as part of Specific aim 2.a, the IAQ model was used in Monte Carlo simulations by sampling from uncertainties of parameters that cover the retail sector, rather than just the measured parameters. Monte Carlo simulations were performed using Stata version 12.1 (StataCorp, College Station, USA) and convergence was obtained by replicating the simulation 20,000 times for each of the ventilation scenarios, cities, seasons, periods, filter efficiency, and pollutants considered. Specifically, different ventilation rates were used in the IAQ model to test their impacts on pollutant concentrations; these rates corresponded to those calculated from the VRP, DCV-based CO_2, and IAQP-based $PM_{2.5}$ (as described in ASHRAE Standard-62.1-2010). Two additional strategies were investigated as alternatives to the VRP (VRP-C and VRP-NG) and are only reported for some of the analyses. For the two additional strategies, VRP-C differ from the VRP by setting the ventilation rate zero at night, and VRP-NG differ from the VRP by reducing the ventilation rate specified for non-grocery stores to be equal to that specified for grocery stores. Strategies other than the IAQP scenario are described in Figure 5.

As indicated in Figure 5, the ventilation rates prescribed by the VRP or DCV are the sum of two quantities: the minimum rate of outdoor air supply per occupant (denoted as Rp in Figure 5), and the minimum rate of outdoor air supply per unit floor area (denoted as Ra in Figure 5). These rates are specified in ASHRAE Standard 62.1-2010 according to building use. While the VRP uses the maximum number of people (also specified in ASHRAE Standard 62.1-2010) to calculate the minimum rate of outdoor air supply required for occupants, the DCV uses the actual number of people at the store (taken from people counters, or deduced from register transaction numbers, or calculated from CO_2 concentrations measured in the space).

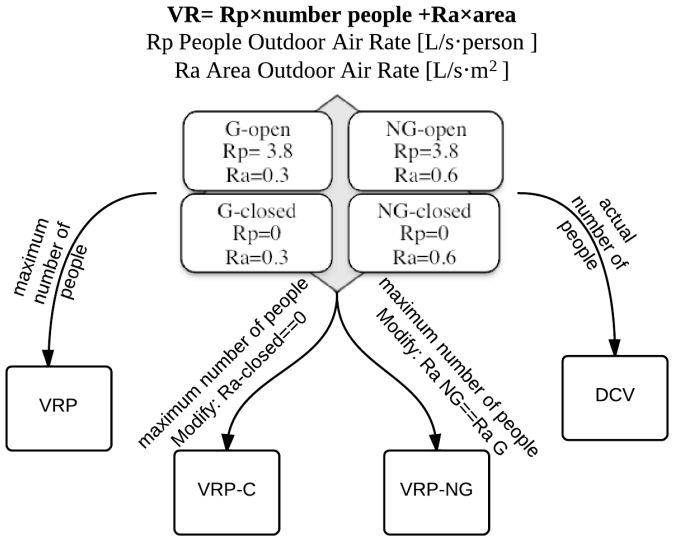

Figure 5: Summary of ventilation scenarios for grocery stores (designated as G), and non-grocery stores (designated as NG).

For IAQP, the IAQ model was applied to find the ventilation rate that will result in the desired $PM_{2.5}$ concentration. This concentration is defined in ASHRAE Standard 62.1-2010 (Appendix B; ASHRAE, 2010) and corresponds to the National Ambient Regulatory limit (NAAQS) of 12 $\mu g/m^3$ over a one-year average. Detailed descriptions of the distributions of other inputs to the IAQ model are summarized in Paper 3 (Appendix D).

4.2.2 DALY model

Two different approaches were used to estimate DALYs lost from exposure to contaminants: (1) the intake-incidence-DALY (IND) approach based on epidemiology-based concentration-response (C-R) functions following the work of Logue et al. (2012); and (2) the intake-DALY (ID) approach based on animal toxicity literature following the work of Huijbregts et al. (2005). Only information on C-R functions in humans are

available for $PM_{2.5}$. For the rest of pollutants, animal toxicity data was available in the literature to apply the ID approach.

For both methods, Monte Carlo simulations were used to quantify the central estimates and 95% confidence intervals (CI) by sampling from uncertainty distributions of DALY factors. Monte Carlo simulations were performed using Stata version 12.1 (StataCorp, College Station, USA) and convergence was obtained by replicating the simulation 30,000 times for each of the ventilation scenarios, cities, seasons, periods, filter efficiency, and pollutants considered. Equations and input parameter distributions used in DALY calculations collected from the literature are summarized in Table 2. Retail occupancy characteristics and cancer age adjustment factor used in DALY equations are presented in Table 3.

Table 2: Summary of DALY formulas, disease outcomes, and inputs used in the calculation

DALY Formula	Outcome	Input (95% CI)	Source
$PM_{2.5}$ DALYs = $y_0 \times (\exp(-\beta \times \Delta C_{exposure}) - 1) \times population \times DI$ Where: y_0 is the baseline incidence of outcome of interest, β is $PM_{2.5}$ coefficient, $\Delta C_{exposure}$ is the exposure concentration, and DI is the DALYs lost per incidence.	Mortality	β=0.0058(0.002,0.01) y_0=0.0074 DI=1.4 (0.14, 14)	Pope et al., 2002; 2009
	Chronic Bronchitis	β=0.091(0.078,0.105) y_0=0.0004 DI=1.2(0.12,12)	Logue et al., 2012
	Non-fatal Stroke	β=0.025(0.002,0.048) y_0=0.0002 DI=11.7 (11.1, 12.4)	Hong et al., 2010; Logue et al., 2012
VOCs, aldehydes, ozone DALYs = $C_{exposure} \times V \times (\frac{\partial DALY_{cancer}}{\partial intake} \times ADAF + \frac{\partial DALY_{non-cancer}}{\partial intake})$ Where: $C_{exposure}$ is the exposure concentration, V is the volume of air breathed (shown in Table 3). ∂DALY/∂intake are the cancer and non-cancer	Cancer	Example: Formaldehyde: ∂DALY/∂intake=0.76 (y. kg^{-1}) K=26	Huijbregts et al., 2005

mass intake-based DALY factors, and ADAF is the age-dependent cancer adjustment factor (shown in Table 3*)*. Uncertainty K was assumed to follow the log-normal distribution and was calculated as: $K = (\frac{97.5 percentile}{2.5 th percentile})^{0.5}$	Non-cancer	Example: Acrolein: $\partial DALY/\partial intake=50$ ($y.\ kg^{-1}$) K=277	Huijbregts et al., 2005

Table 3: Retail occupancy characteristic and age-dependent cancer adjustment factors.

Category	Age (years)	Cancer ADAF[a]	Percent of Day Spent at Retail[b]	Air Intake (m^3/day)[a]
Employees	≥16	1	29	15
Customers	2-16	3	0.5	13
Customers	≥16	1	2	15

Sources: [a] U.S. EPA exposure factors handbook (2011); [b] ATUS (2011).

4.2.3 System cooling capacity and compressor power model

The impact of filter pressure drop on cooling capacity, sensible heat ratio, and compressor power were modeled using two vapor compression models: the ACHP vapor compression system model (Braun, 2013) and the Secondary HVAC Toolkit software (Nassif, 2013). Other inputs included temperature and relative humidity (measured during field work, described in Section 4.1.3), geometry of coils, refrigerant type (obtained from manufacturer specifications), and fan power and airflow rate (measured during field work, described in Section 4.1.3). Comparison of the modeling results of the two models showed very similar results, and in this work the results from the ACHP vapor compression system model are reported.

4.2.4 Filtration model

The established relationships between (1) filter pressure drop and (2) system parameters (described in Sections 4.1.3 and 4.2.3, including airflow, fan speed, fan power, cooling capacity, compressor power, and duty cycle) study results were

synthesized in the context of comparing the energy and particle reduction performance of different filters (Specific aim 2.b). The energy consequences of duct leakage due to high pressure drop across filters were excluded from the model because HVAC distribution systems in retail stores typically have almost no ducts and instead deliver conditioned air directly to the space.

A filter database consisted of 75 different filters. Test reports of 60 of those filters were obtained from 15 different manufacturers, and test reports for the rest of filters were obtained from Rivers and Murphy (1999). The test reports obtained from the manufacturers or from Rivers and Murphy (1999) provided particle fractional efficiency (η) for each filter. These efficiencies were averaged within three particle size range groups (E_1: 0.3 μm -1 μm; E_2: 1 μm - 3 μm; E_3: 3 μm - 10 μm). Then, these filters were classified with an ASHRAE Standard 52.2-2012 Minimum Efficiency Reporting Value (MERV): the sample consisted of MERV 8 (n=16), MERV 11 (n=21), MERV 13 (n=24), MERV 14 (n=14) filters. Next, the integrated filter efficiency for $PM_{2.5}$ was calculated using the same procedure described in Section 4.2.1.

In addition to efficiency, the filter test reports provided several data points linking the filter resistance (i.e. pressure drop) to airflow or face velocity. Using this information, a second-degree polynomial fit was established for each filter between pressure drop and the face velocity of the system ($p = \text{constant}_1 \times V^2 + \text{constant}_2 \times V$, where p is the pressure in Pa and V is the face velocity in m/s). A detailed discussion of the face velocity chosen in this work is detailed in Paper 2 (Appendix C).

The energy performance of filters was calculated for cooling mode by incorporating the fan power and the compressor power; it was calculated for fan-only mode by considering only the fan power to depict economizer mode and ventilation modes. The percentage contribution of fan power to the total power was calculated from

the collected dataset for each measured unit. The resulting total power draw was then multiplied by the additional time (i.e., duty cycle) that the system would run to achieve the same cooling capacity when system airflow is decreased. Figure 6 details the methodology to calculate the power during cooling mode.

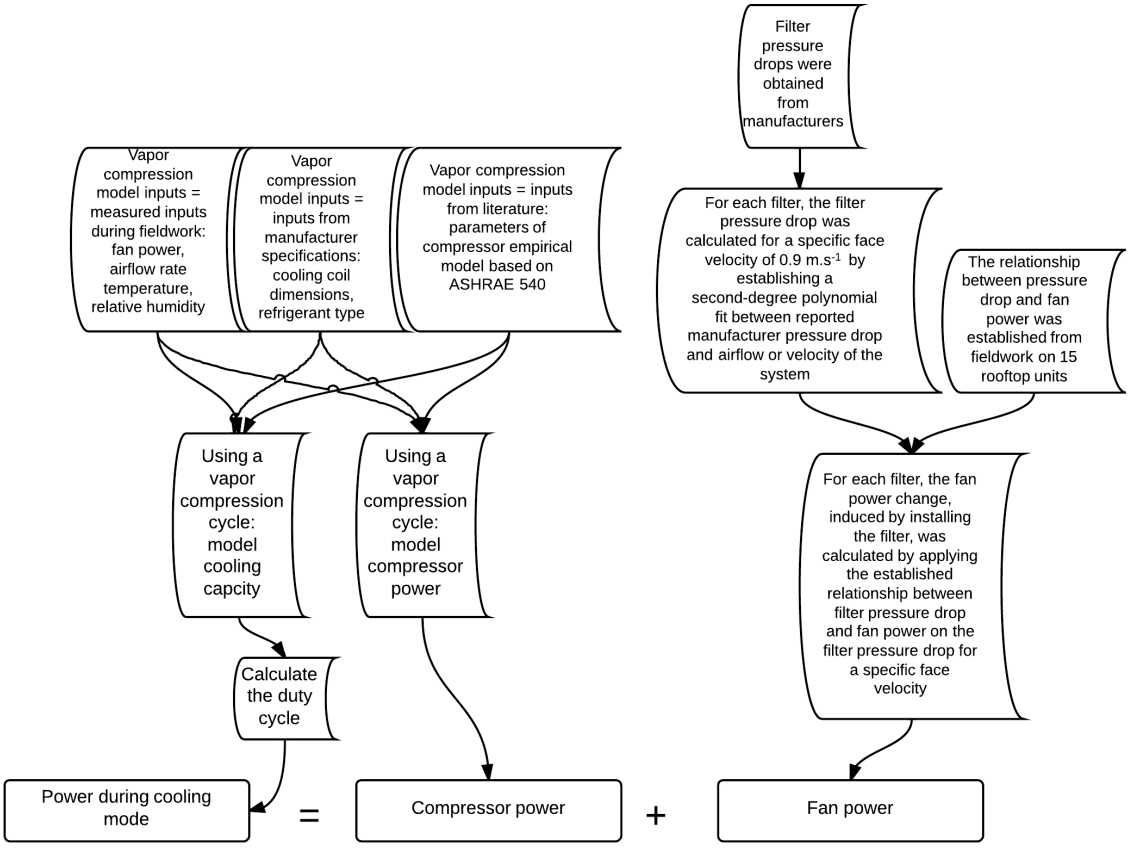

Figure 6: Schematic detailing the methodology to obtain the power during cooling mode.

The indoor air quality performance of filters was calculated using the clean air delivery rate (CADR; e.g., MacIntosh et al., 2008), which takes into account the change in efficiency (as calculated above), as well as the change in airflow through the filter (as measured in the fieldwork, Section 4.1.3). Figure 7 details the methodology to calculate the clean air delivery rate (CADR).

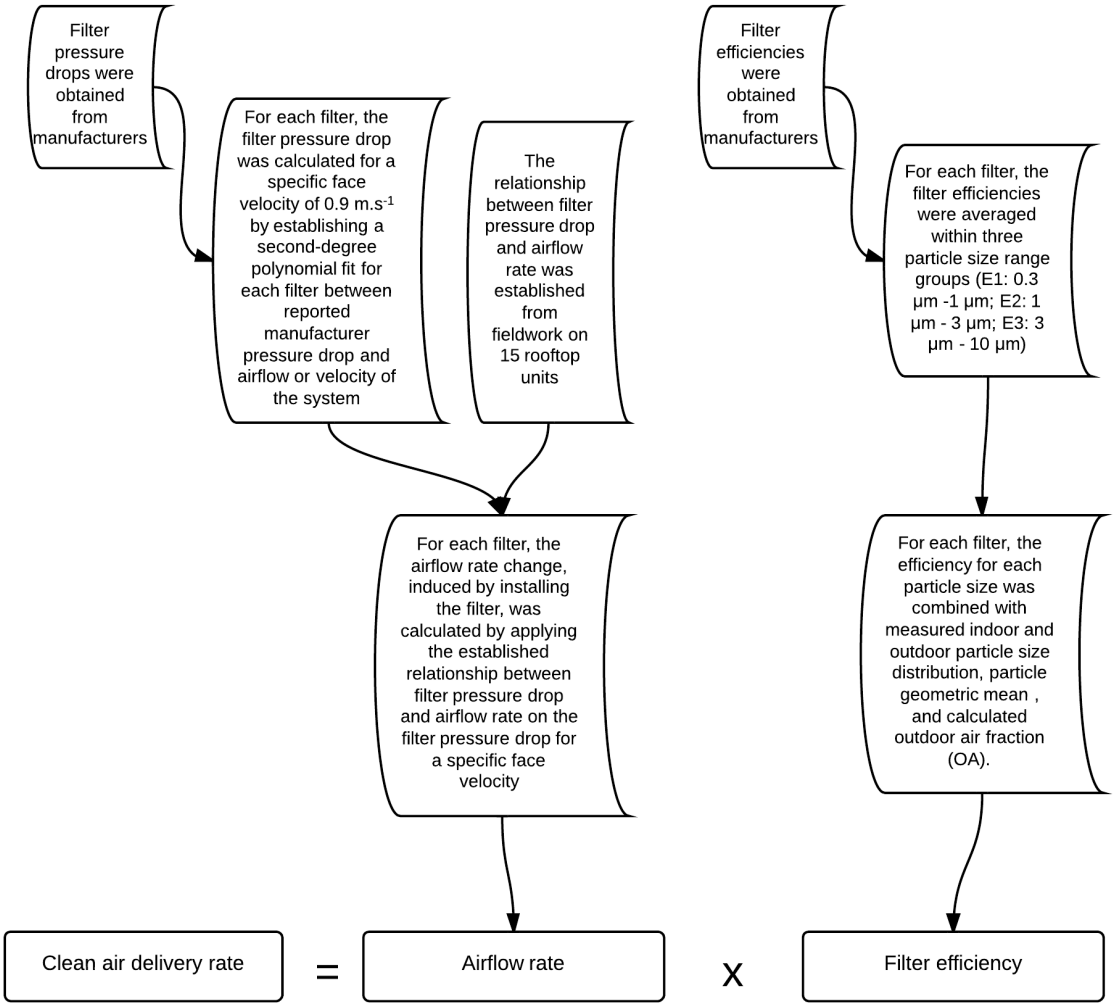

Figure 7: Schematic detailing the methodology to obtain the clean air delivery rate (CADR).

4.2.5 Energy model

To assess the energy consumption of different ventilation and filtration scenarios, an energy model for a typical retail store was built in e-Quest software. The 10,000 square meter (representative of a big box retail store), single story building contains retail sales floor, and stock storage areas. The retail sales area includes a food service component and a grocery component that includes refrigerator cases. The HVAC system

used in the store model is a set of commercial rooftop direct expansion (DX) cooling units. Independent units located on the roof provide grocery refrigeration cooling. The major electricity usages of the store that are independent of the climate considered are lighting, refrigerator cases, food preparation, and others lumped under miscellaneous equipment.

Using this model, comparisons were made of the energy required to cool and distribute the air through the building, over a range of different ventilation scenarios (described in Section 4.2.1), and for U.S. six cities (Austin, Philadelphia, Minneapolis, Seattle, Los Angeles, Phoenix) representative of different climate zones and outdoor air quality. The analysis takes into account cooling only as the filtration fieldwork analysis was conducted for cooling mode. The cost of electricity (used later when comparing different control strategies) was assumed to be eleven cents per kilowatt-hours (EIA, 2013). The price considered corresponds to the average retail price for electricity across the commercial sector and it does not include peak power charges as they vary by city, and inclusion would inhibit comparison between sites, causing the energy cost results to depend more on the energy policy rather than climate and outdoor/indoor air pollution.

Chapter 5: Summary of Results

This dissertation addresses two main research objectives using four investigations. Two of these investigations resulted from fieldwork (Specific aim 1.b and 2.b), and it is important first to describe an overview of these results, as they will be used in the rest of the results section. This overview constitutes the first part of the results (Section 5.1). The second part combines all the fieldwork results and the relevant data from literature studies to determine the main contaminants of concern found in retail stores and the energy, air quality, and health impacts of different control strategies (Section 5.2).

5.1 FIELDWORK RESULTS

For brevity, only $PM_{2.5}$ related results are presented in this section. Details about other PM sizes can be found in Paper 1 and Paper 2 (Appendices B and C).

5.1.1 $PM_{2.5}$ characterization

This section summarizes $PM_{2.5}$ concentrations, indoor-to outdoor ratios, emission rates by retail type, relationship of indoor concentrations with air exchange rates and filtration efficiencies. These results used sampled $PM_{2.5}$ indoor and outdoor concentrations (methodology described in Section 4.1.1), HVAC characterization (methodology described in Section 4.1.2), and the IAQ model (methodology described in Section 4.2.1). The detailed results can be found in Paper 1 (Appendix B).

Indoor concentrations, indoor-to-outdoor ratios, and indoor sources

Figure 8 shows measurements of $PM_{2.5}$ indoor concentrations and in indoor-to-outdoor (IO) ratio for the 22 sampled sites. These sites are grouped by store types between dashed lines. Within the same type of stores, dotted lines are used to separate

different brands of stores. The maximum concentration measured at each site is displayed in parentheses.

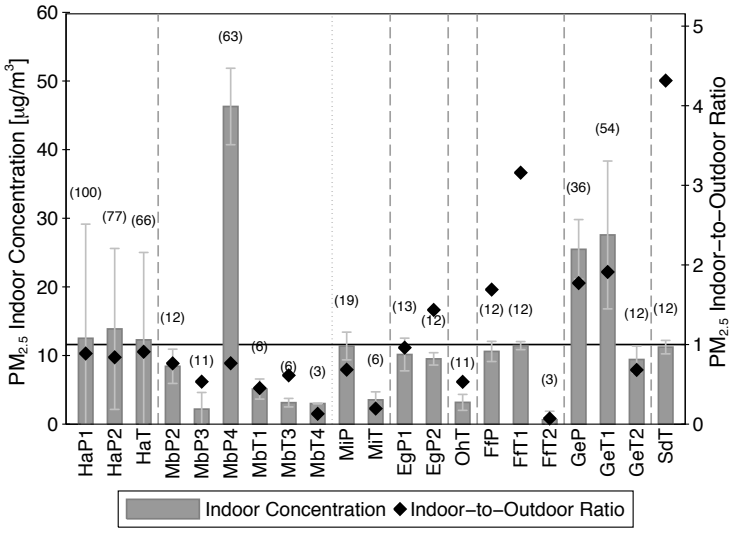

Figure 8: $PM_{2.5}$ indoor mass concentration and indoor-to-outdoor ratio by site. Maximum concentration ($\mu g/m^3$)is presented in parentheses. Horizontal black line represents $PM_{2.5}$ IO ratio equal to 1.0 (right axis).

$PM_{2.5}$ concentrations (mean ± sd) were 11±10 $\mu g/m^3$, and ranged from 0.6 $\mu g/m^3$ at Site FfT2 to 46.3 $\mu g/m^3$ at Site MbP4. Across all stores, the average $PM_{2.5}$ concentrations reported in this study were slightly lower than the average $PM_{2.5}$ (11.6 $\mu g/m^3$) reported for retail stores measured in the U.S. (Grimsrud et al., 1999; Rea et al., 2001; Bennett et al., 2011; Brown et al., 2012; Chan et al., 2012). The average $PM_{2.5}$ reported for grocery stores measured in the U.S. was 31.7 $\mu g/m^3$, which is higher than the results of this investigation (17.5 $\mu g/m^3$). Dissimilarities may be due to differences in outdoor concentrations, occupant density, ventilation rate, the amount and frequency of cooking activities and associated use of exhaust fans, as well as differences in collection and/or calibration techniques. For other environments, generally PM concentrations

reported for retail stores (this work and previous studies) were lower than concentrations reported for restaurants (Buonanno et al., 2010; Bennett et al., 2011; Brown et al., 2012), residences (Clayton et al., 1993; Wallace et al., 1994), and classrooms (Branis et al., 2005), but higher than the concentration reported in offices (Burton et al., 2000).

$PM_{2.5}$ indoor-to-outdoor (IO) ratios were 0.88±1.0. Sites with $PM_{2.5}$ IO ratios larger than one (e.g., grocery stores for any particle size) suggest the presence of a strong indoor source. However, explicit conclusions are difficult to make when $PM_{2.5}$ IO ratios are less than or equal to unity as this may indicate one or a combination of the following factors: low indoor emission rates, effective filtration, and/or high outdoor concentrations.

Across all sites, there was no obvious association of $PM_{2.5}$ indoor concentrations with store type. Generally, $PM_{2.5}$ concentrations for all sites were elevated in grocery stores with cooking activities (i.e., frying, baking, grill/burner use), which was consistent with data reported in the literature for residential and hospitality environments (e.g., Buonanno et al., 2010; Wallace & Ott, 2011; Bennett et al., 2011). Outdoor sources, associated with nearby agriculture burning, were significant at Site MbP4 (Outdoor $PM_{2.5}$ = 60 $\mu g/m^3$) and likely contributed to the high indoor $PM_{2.5}$ concentration measured in these stores.

For all sites with high indoor concentrations, these concentrations were generally associated with high activity levels for customers and/or employees or high outdoor concentrations, with the exception of home improvement stores. Home improvement stores had indoor particle sources independent of consumer activities; the average mass concentration for $PM_{2.5}$ measured during business hours (7:00 AM till 10:00 PM) were 30% less than the 24-hour average concentration. During this time, outdoor concentrations were lower than indoors. These indoor sources were potentially caused by

diminished ventilation rates or the use of forklifts at night. There was very limited evidence for secondary particle formation from precursor emissions such as volatile organic compounds.

Indoor emission rates

Modeled distributions of indoor emissions of $PM_{2.5}$ were calculated by arranging Equation 1 for a total of n=13,122 runs and are plotted by site in Figure 9. Median $PM_{2.5}$ indoor emission rates ranged from 0.42 $\mu g/m^2 \cdot h$ at Site FfT2 to 193 $\mu g/m^2 \cdot h$ at Site GeP. The highest emission rates were observed at two grocery stores (GeP, GeT1), likely due to the presence of intense cooking activities and heavy foot traffic. The low indoor emission values observed at some sites indicate either that the indoor concentrations were low or they were dominated by outdoor sources.

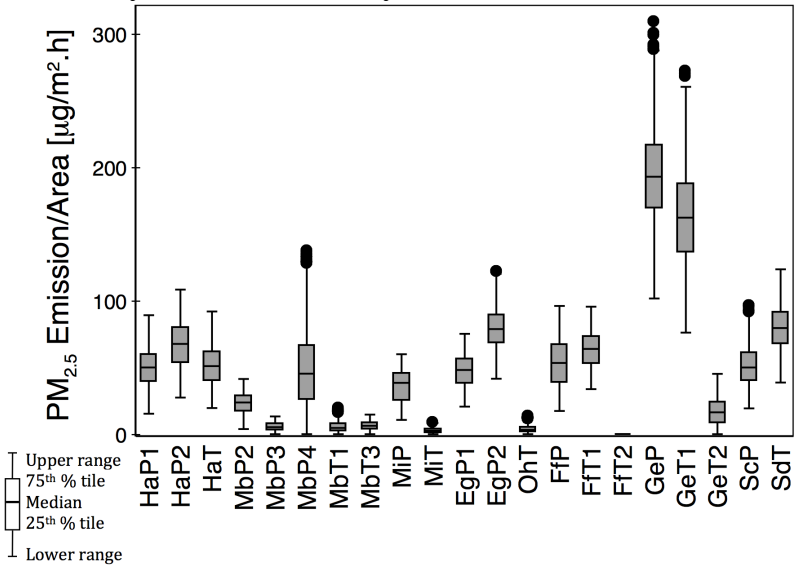

Figure 9: $PM_{2.5}$ indoor emission source rates normalized by area.

For stores measured more than once, indoor emission rates serve to explain the difference in the measured indoor concentration, given the measured ventilation rates and outdoor concentrations. An example is the grocery store Brand e (i.e., GeT1 and GeT2),

which was measured twice in Texas. The outdoor concentration and the total ventilation rate were comparable in both visits. The median indoor $PM_{2.5}$ emission rate calculated for the first visit (Site GeT1) was 162 $\mu g/m^2 \cdot h$ and for the second visit (Site GeT2) was 14 $\mu g/m^2 \cdot h$. Because the number of transactions (a proxy for human activity) was a factor of 1.8 higher at the second visit (Site GeT2), the difference of emission rates between these visits may be better explained by differences in the amount of cooking activities. These results suggest the complexity of particle sources and sinks in these environments.

Contribution of indoor sources to indoor concentrations

Using the modeled median emission rate, the contribution of indoor sources (versus the contribution of outdoor sources) was estimated by isolating the portion of the indoor concentration generated from indoor sources. Median contribution of indoor sources to $PM_{2.5}$ was 53%. For grocery Sites GeP and GeT1, indoor sources such as cooking activities contributed significantly to the elevated concentration (70-75%). These findings show the importance of indoor sources and are consistent with findings reported in the literature (Abt et al., 2000; Wu et al., 2012). Outdoor sources can also be significant. Site MbP4 had the largest $PM_{2.5}$ concentration among all the tested sites; here, outdoor sources contributed to 76% of the indoor concentration.

Effect of ventilation and filtration on $PM_{2.5}$ concentrations

Across all sites, the mean and standard deviation of the ventilation rate (ACH) were 0.63 ± 0.37 per hour (ACH for stores located in Pennsylvania: 0.66 ± 0.41 per hour; ACH for stores located in Texas: 0.60 ± 0.33 per hour) and ranged from 0.2 to 1.5 per hour. The Shapiro-Wilk test showed that PM concentrations and indoor-to-outdoor ratio variables across all sites did not follow the normal distribution. Spearman Rank Correlation tests performed across all sites showed that particle concentrations and IO

ratios measured for all particle sizes did not exhibit a statistically significant relationship with air exchange rates (p values of all the tests was higher than 0.05). Several factors may have contributed to this finding. These mainly include variability in indoor and outdoor sources and removal mechanisms: the small sample size and the fact that the air exchange rates were generally low likely played a role as well. To account for some of these differences, the impact of changing the ventilation rate within the same store on PM concentrations was investigated experimentally in two of the measured general merchandise stores (intervention sites, MiP and MiT). Each intervention site was tested for two consecutive weeks, one week at the normal ventilation rate and another week at the maximum ventilation rate allowed by the store. Figure 10 contains a scatter plot of indoor $PM_{2.5}$ concentration as a function of ACH. Arrows on the plot shows the impact of change in $PM_{2.5}$ concentration when going from normal to elevated ACH at these sites.

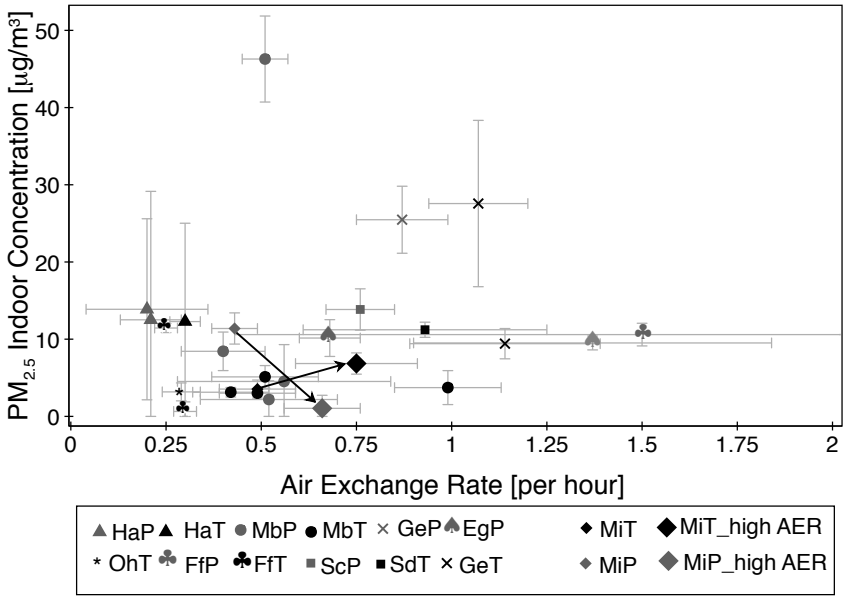

Figure 10: $PM_{2.5}$ indoor mass concentration by air exchange rate (ACH). Arrows visualize the impact on $PM_{2.5}$ concentration when going from normal ACH to high ACH for two Sites: MiP and MiT.

Figure 10 shows that increasing the ventilation rate by 54% at the intervention Site MiP resulted in a reduction in $PM_{2.5}$ mass concentrations of 90%. At Site MiT, $PM_{2.5}$ mass concentration exhibited the opposite relation: 94% increase in $PM_{2.5}$ concentration with 53% increase in ventilation rate. Indoor emission rates and outdoor concentrations were different between both weeks for both stores. This suggests that determining the minimum ventilation rate to lower indoor PM concentrations should be based on the knowledge of indoor emission rates and outdoor concentrations.

Besides the experimental investigation, indoor $PM_{2.5}$ concentrations were modeled for different scenarios of ventilation rates, infiltration rates, and filtration efficiencies. The focus was to assess possible control strategies for the sites that exceeded the $PM_{2.5}$ concentration of interest defined in ASHRAE Standard 62.1-2010 (Appendix B; ASHRAE, 2010). Note that this concentration corresponds to the National Ambient Regulatory limit (NAAQS) of 12 $\mu g/m^3$ over a one-year average. The detailed modeling results are described in Paper 1 (Appendix B). The main conclusions were (1) the direction and magnitude of changing ventilation rate to lower $PM_{2.5}$ indoor concentrations depends on the indoor-to-outdoor ratio; (2) infiltration levels and high emission rates present at some sites diminished the effect of high filtration efficiency, (3) eliminating the infiltration rates (i.e., with positive pressurization) and using a highly efficient filter would drive $PM_{2.5}$ concentrations in most retail stores, including stores with elevated outdoor concentrations, below the $PM_{2.5}$ concentration of interest of 12 $\mu g/m^3$. These results show that $PM_{2.5}$ concentrations can only be reduced effectively if we account for all pollutant influencers: indoor and outdoor sources (hence, the importance of retail type and location), infiltration, ventilation, and filtration. Most importantly, there remains a need to identify the energy and air quality impacts of different reduction strategies.

5.1.2 Impact of higher efficiency filters on energy and indoor air quality

This section summarizes the impact of increased pressure drop on rooftop unit performance equipped with or without fan speed control and the potential energy impact and air quality consequences when switching from low MERV to high MERV filters. These results used measured impact of pressure drop on HVAC system parameters (methodology described in Section 4.1.3), modeled system cooling capacity and compressor power (methodology described in Section 4.2.3), and the filtration model (methodology described in Section 4.2.4). The detailed results can be found in Paper 3 (Appendix C).

Impact of increased pressure drop on rooftop unit performance

Commercial rooftop units equipped with no fan speed control generally behaved like residential systems when subjected to an increase in pressure drop. Increasing the pressure drop across filters led to system curve changes in a way that induced flow decrease and fan static pressure increase, as shown in Figure 11. As a result of the flow decrease, the fan power, cooling capacity, and compressor power also decreased.

Field results, across all the 14 RTUs tested, showed that doubling the pressure across filters resulted in a median airflow decrease of 15.9%, and a median fan power decrease of 12.8%. The reason for this significant power decrease is the larger impact of flow decrease compared to the impact of pressure rise while the fan static efficiency increases only marginally (from the fan power equation: $W = \Delta p \times Q/\eta$; where W is the fan power, Δp is the static fan pressure rise, Q is the airflow rate, and η is the static fan efficiency).

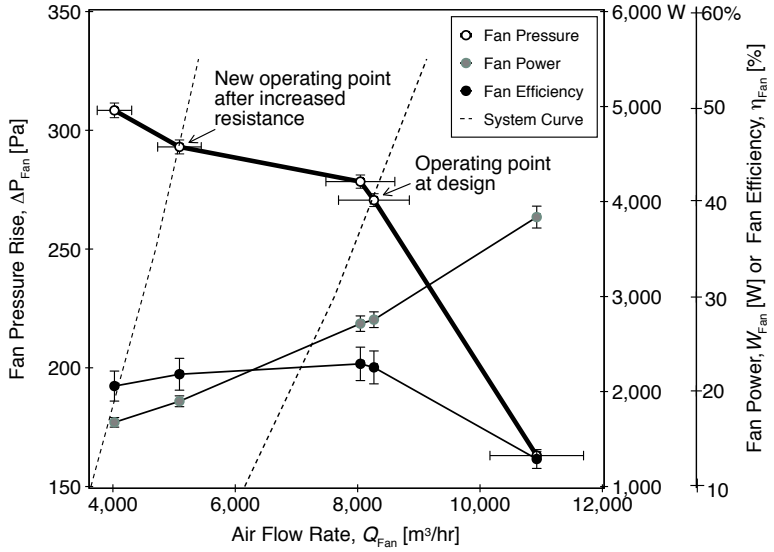

Figure 11: Measured fan and two theoretical system curves (assuming p = constant × Q^2) for a 70 kW nominal capacity unit.

For units with fan speed control, increasing the pressure drop across filters yielded fan curve changes that maintain the flow constant, as shown in Figure 12. As a result of pressure drop increase, the speed and the power of the fan increased. Doubling the pressure drop across filters resulted in the same magnitude change of fan power observed for units with no fan speed control but in the opposite direction. Other parameters were unchanged.

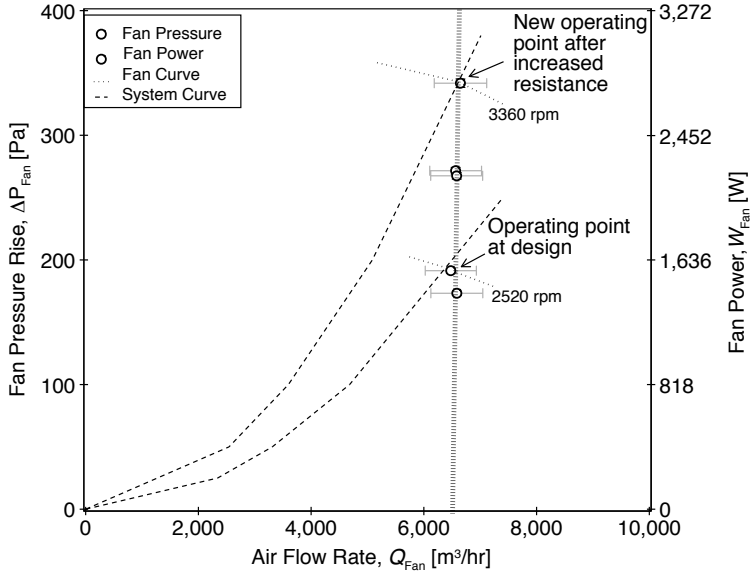

Figure 12: Measured fan operating points and two theoretical system curves (assuming p = constant × Q^2) for a 84 kW nominal capacity unit.

Potential energy impact and air quality consequences when switching from low MERV to high MERV filter

The energy and air quality performance of MERV 8 filters was compared to that of higher efficiency filters (MERV 11, MERV 13, and MERV 14). The MERV 8 category was chosen as the basis of comparison as it is the most commonly used category in rooftop units in commercial buildings. Different MERV 8 filters (i.e., different filter pleating, depth, and area) have different resistances and efficiencies and do not perform the same way on all particle size ranges. To capture the performance of different filters with the same MERV 8 category, five base cases were chosen to cover a wide range of pressure drop and efficiencies encountered in the marketplace for this filter category. Figure 13 summarizes the pressure drop and integrated $PM_{2.5}$ and PM_{10} particle efficiencies for these five filters.

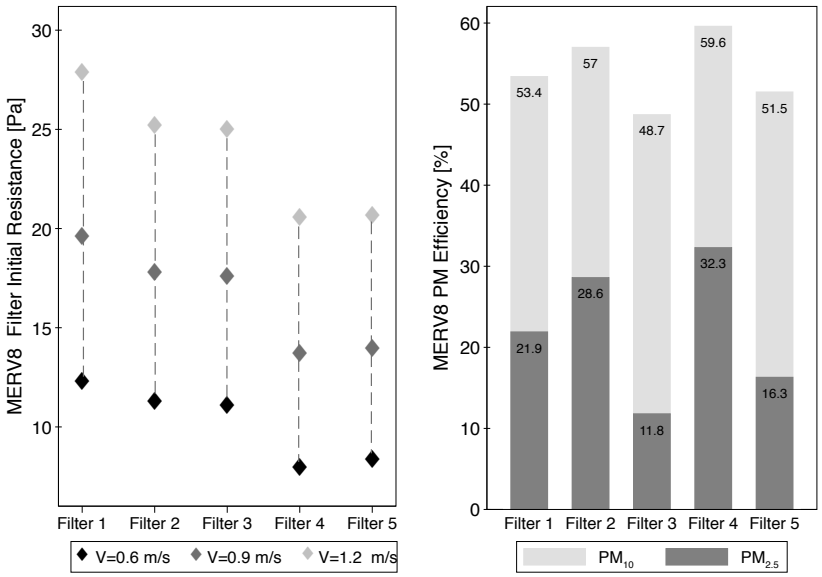

Figure 13: Initial resistances at face velocities of 0.6, 0.9, and 1.2 m/s (left), and $PM_{2.5}$ and PM_{10} efficiencies (right) for five MERV 8 base case filters.

As described in Section 4.2.4, the impact of replacing a MERV 8 filter with a higher MERV filter on power consumption was calculated for cooling mode by incorporating the fan power, compressor power, and duty cycle. For fan-only mode, this analysis considered only the fan power. The impact on particle concentrations was investigated using the CADR metric (which takes into account the change in filter efficiency as well as the change in airflow, as explained in Section 4.2.4). For brevity, only the impact of switching MERV 8 filter with higher MERV filters under clean conditions (i.e., the filter pressure drop corresponds to the pressure drop of a particle-free filter) are discussed below. The effect of filter loading is discussed in details in Paper 2 (Appendix C).

Figure 14 summarizes these findings for units equipped with no fan speed control. There are three columns in each of these two figures; each column corresponds to replacing MERV 8 filter with a higher MERV filter: MERV 11 (left), MERV 13

(middle), and MERV 14 (right). The distribution represents the range of impact for different filters with the same MERV category. The number of filters for each MERV is indicated at the top of each column. There are five distributions in each sub-figure that correspond to the impact on energy and air quality relative to five base case MERV 8 scenarios.

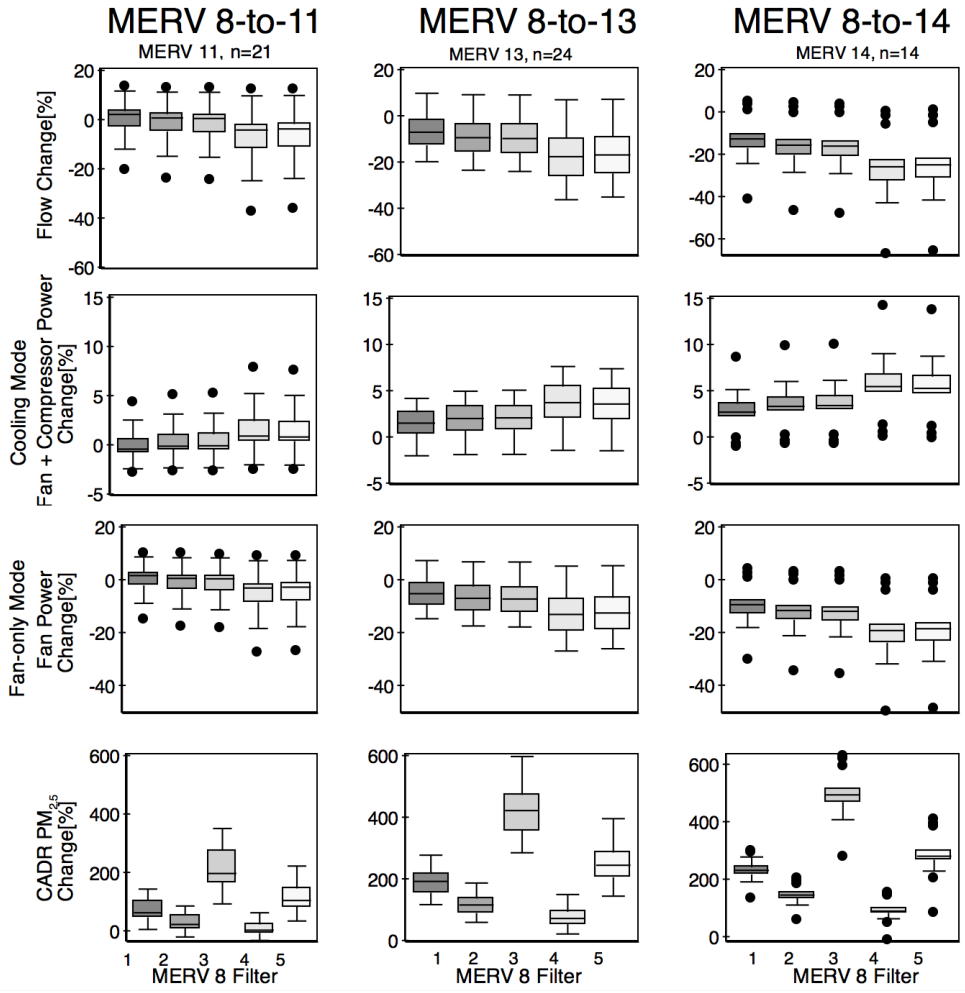

Figure 14: Box plots showing the impact of replacing MERV 8 with MERV 11 (left), 13 (middle), 14 (right) on flow, total power (compressor and fan), fan power, and CADR $PM_{2.5}$ for five MERV 8 base cases for a face velocity of 0.9 $m \cdot s^{-1}$ for units equipped with no fan speed control.

Results displayed in Figure 14 show that replacing MERV 8 with MERV 11 marginally decreased the airflow; across all the MERV8 base cases, the median decrease in airflow was 1.9%. As a result of the marginal flow decrease, the power during cooling mode increased by 0.4% and during fan-only mode decreased by 1.4%. This indicates that the energy consequences of MERV 11 compared to MERV 8 filter were minimal. As for CADR, MERV 11 offered a moderate increase in $PM_{2.5}$ CADR (median increased by 63%). Note that for three MERV 11 filters, CADR was decreased because the diminished flow caused by installing MERV 11 was not compensated for by the increased efficiency of MERV 11.

Results in Figure 14 also shows that replacing MERV 8 with MERV 13 and MERV 14 filters had a larger effect on the studied parameters than those of a MERV 11 replacement. The median decrease in flow was 10.8% when using MERV 13 and 17.6% when using MERV 14. For cooling mode, the increased duty cycle to meet the same cooling load requirements caused a median total power draw increase of 2.4% for MERV 13 and 3.7% for MERV 14. For fan-only mode, the fan power decreased 8.4% for MERV 13 and 13% for MERV 14, suggesting a potential benefit when switching from a low to a higher-pressure drop filter. The total energy impact of high efficiency filters (i.e., MERV 13, MERV 14) depends on the time the system runs in cooling mode and fan-only mode. For example, if the system was running 100% in fan-only mode (i.e., with no cooling through cooling coils), then replacing a low efficiency filter with a higher efficiency filter may result in energy savings, especially given that fan energy contributes to at least 7% of total site energy consumption, as previously mentioned. As another example, if the system was running in 25% fan-only mode, then replacing a MERV 8 filter by MERV 14 may result in energy neutral consumption. Besides the energy consequences, replacing

MERV 8 with high MERV filters offered largely better indoor air quality (at least 2.9 times increase in $PM_{2.5}$ CADR).

Comparing MERV 13 with MERV 14 filter revealed that the median $PM_{2.5}$ CADR resulting from installing a MERV 14 filter was only 1.2 times higher than that of a MERV 13 filter. The increased $PM_{2.5}$ CADR for MERV 14 filter was accompanied with 1.5% increase in power consumption during cooling mode and similar percentage decrease in fan-only mode.

As for units with fan speed control, replacing MERV 8 with a higher MERV filter caused an increase in fan power to maintain a constant flow, as mentioned previously (figure excluded for brevity, see Figure 11 of Paper 2 Appendix C for more details). In contrast to units without fan speed control, there was no savings in fan power; the fan power draw increased in both cooling mode and fan-only mode. For MERV 11, the fan power increased 2%. For MERV 13 and MERV 14, the increase in fan power was more important: 11% for MERV 13 and 18% for MERV 14. Accounting for the change in total power during cooling mode, the increase in power draw for units with and without fan speed control was comparable. However, during fan-only mode, using high MERV filters incurred additional fan cost when using units with fan speed control and incurred energy savings when using a fan without speed control.

Since the flow in the units equipped with fan speed control remained the same, the CADR improvement was simply an efficiency improvement: $PM_{2.5}$ CADR increased 1.7 times with MERV 11, 3.1 times with MERV 13, and 3.8 times with MERV 14. The constant flow for these units caused the CADR to be greater than the CADR in units without fan speed control.

Two previous studies performed field measurements in residences equipped with small systems to test the impact of high MERV filters on energy use (Stephens et al.,

2010: 15 units without fan speed control and two units with fan speed control; Walker et al., 2012, which also included simulation work: seven units without fan speed control and three units with fan speed control). Both studies found minimal effects of mid/high MERV (range mid MERV: 6-8; range high MERV: 11-13) on fan power (<5%) when replacing low MERV filters (range low MERV 2-5). Additionally, Walker et al. (2013) found that fan power draw became significant when MERV 16 was used (about 20%), although these results were for thick filters. Also, Stephens et al. (2010[b]) found minimal energy reduction (0.07 $kWh/kW_{capacity}$·day) following the switch to high MERV, although the uncertainty was relatively large (±1.3 $kWh/kW_{capacity}$·day). Direct comparison of findings form these two previous studies with the results presented in this work is challenging due to the differences in the filters considered and the equipment. Also, other factors that are reasoned by these studies may affect the results (e.g., duct leakage, initial airflow resistance, capacity rating of the system, etc.). Yet the general consensus in the literature (including this work) is that using a mid MERV filter (up to MERV 11) has minimal impact on energy consumption in cooling or fan-only mode; this is for both units equipped with or without fan speed control.

5.2 EFFICIENT REDUCTION OF INDOOR EXPOSURES TO CONTAMINANTS OF CONCERN

This section synthesizes all the fieldwork results and the relevant data from literature studies with the models described in Section 4.2 first to determine contaminants of concern found in retail stores and, second, to conclude effective methods to decrease the exposure to these pollutants and the energy consumption required to control them, as described below.

5.2.1 Exposure modeling of contaminants of concern in retail buildings

Contaminants of concern (COCs) found in retail stores were evaluated using the DALY metric (methodology described in Section 4.2.2). The pollutant dataset used in this calculation contains pollutant concentrations data from 150 stores in the U.S. $PM_{2.5}$ concentrations were taken from Grimsrud et al., 1999; Rea et al., 2001; Bennett et al., 2011; Brown et al., 2012; Chan et al., 2012; Dong et al., 2013, and this work (see Section 5.1.1). VOCs concentrations were taken from Kim et al., 2001; Tang et al., 2005; Hotchi et al., 2006; Loh et al., 2006; Lee & Hsu 2007; Bruno et al., 2008; Caselli et al., 2009; Bennett et al., 2011; Chan et al., 2012; and Siegel et al., 2013. Ozone concentrations were taken from Chan et al., 2012; and Siegel et al., 2013. From these studies, average concentrations for 34 pollutants (e.g., $PM_{2.5}$, acrolein, formaldehyde, acetaldehyde, trichloroethylene, toluene, benzene, xylene, styrene, carbon tetrachloride, chloroform, naphthalene, ozone, etc.) were included in the analyses. For each pollutant, the average DALYs lost were estimated for customers and employees. Figure 15 shows the estimated number of DALYs lost due to indoor inhalation intake of $PM_{2.5}$, ozone, and selected VOCs and the corresponding pollutant concentrations. The health burden (i.e., number of DALYs lost) reported for employees in retail stores dominated the DALYs lost.

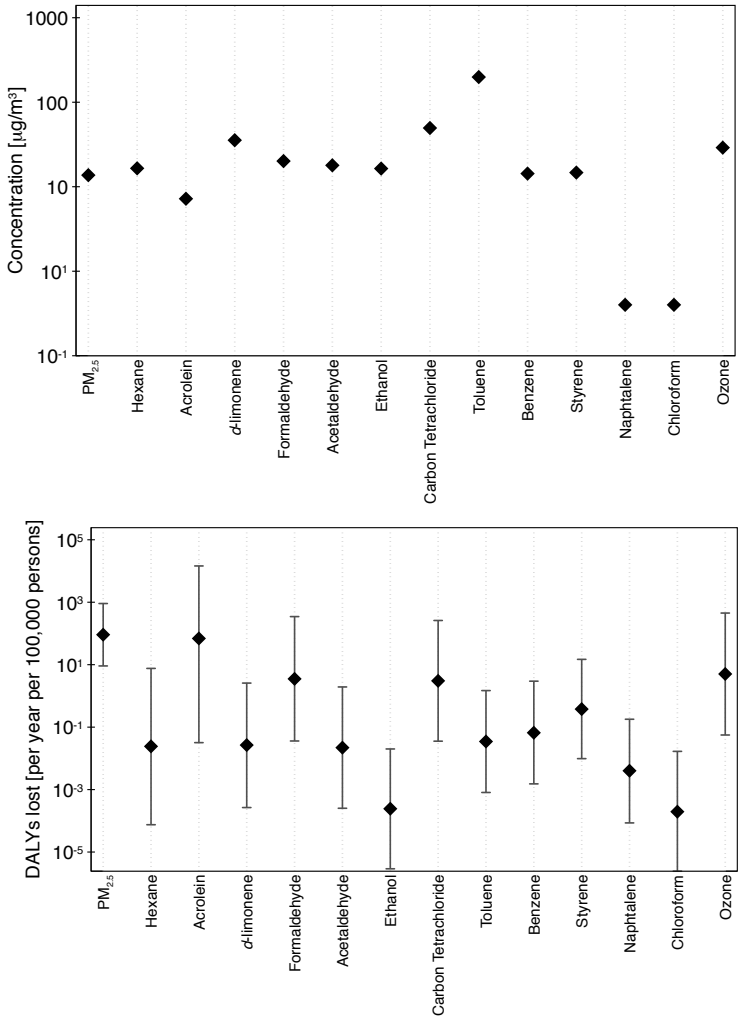

Figure 15: Mean indoor concentrations (upper graph) and DALY losses associated with intake of indoor air (lower graph). Vertical uncertainty associated with DALYs lost represents the 95% confidence interval.

As shown in Figure 15, the estimated average DALYs lost due to $PM_{2.5}$ exposure in retail stores in the U.S. contributed the largest to the annual health impacts, with an estimated 91 DALYs per 100,000 persons annually (95% CI: 9.11, 908). The average annual total DALYs from VOCs/ozone (33 pollutants) contributed an estimated 76 DALYs per 100,000 persons. Acrolein accounted for the vast majority of these losses,

contributing an annual average of 69 DALYs per 100,000 persons (95% CI: 0.3, 14,514). It should be noted that the acrolein concentration used in the DALY analysis is largely based on two grocery stores from one study, where the concentrations of acrolein were elevated likely due to the cooking activities happening in these stores (Chan et al., 2012). Thus the results are highly uncertain and more testing is warranted to verify these concentrations.

Of the uncertainties of DALYs lost assessment stated in the literature review section (Section 3.1), two uncertainties necessitate further discussion to investigate if the suggested contaminants of concern are influenced by these uncertainties. The first uncertainty is related to the threshold of acrolein. Applying a threshold below which no health effects occur would reduce the DALYs estimated for acrolein. This threshold was not considered in the results because little information is known about the threshold level for acrolein. If the exposure limit identified by OEHHA (Office for Environmental Health Hazard Assessment) (0.687 $\mu g/m^3$: 8-hours average) is considered as the threshold for acrolein, the DALYs estimated for acrolein would decrease by 9%.

The second uncertainty stem from using ambient epidemiological data in identifying $PM_{2.5}$ health risks. As stated in the results (Section 5.1.1), the median contribution of indoor sources to indoor $PM_{2.5}$ concentrations was approximately 50%. Indoor sources such as cooking, foot traffic, and cleaning are different in chemical compositions than particles generated outdoors from combustion and other sources. The toxicity of these indoor sources is not fully understood. Since outdoor sources also contribute significantly to indoor $PM_{2.5}$ concentrations, it is reasonable to assume that the ambient epidemiological data applies to some extent to exposure to $PM_{2.5}$ happening indoors. In conclusion, the order of magnitude of the DALYs estimated for both $PM_{2.5}$ and acrolein should be satisfactorily dependable to consider them as the contaminants of

concern in retail buildings, with the caveat that further epidemiological studies are needed to establish the toxicity of indoor-generated particles and more studies are needed to determine the threshold and indoor concentrations of acrolein in retail stores.

Comparison of the DALYs lost calculated for this study to DALYs lost reported for residences by Logue et al. (2012) provides further evidence that $PM_{2.5}$ and acrolein are two common contaminants of concern. In addition to $PM_{2.5}$ and acrolein, Logue et al. (2012) identified formaldehyde as a pollutant that accounts for annual DALYs comparable to acrolein. In these residences, the average concentration reported for formaldehyde was 69 $\mu g/m^3$. In retail buildings, the average formaldehyde reported was 20.1 $\mu g/m^3$ (Hotchi et al. 2006; Wu et al. 2011; Chan et al. 2012; and Siegel et al., 2013). At this concentration, formaldehyde accounts for 3.5 DALYs annually per 100,000 persons. Therefore, DALYs from exposure to formaldehyde in retail buildings can be considered small even though the average concentration reported for formaldehyde in retail stores is 2.7 times the OEHHA chronic reference exposure for formaldehyde (9 $\mu g/m^3$).

Despite the large uncertainties associated with the toxicology data (uncertainty bounds larger than one order of magnitude), the findings suggest that $PM_{2.5}$ and acrolein are the two primary COCs for which controlling methods should be prioritized in retail stores, specifically retail stores with cooking activities. Because exposures in grocery stores differ than those for non-grocery stores, the following results differentiate between these two types of retail stores. Also, besides the established COCs ($PM_{2.5}$ and acrolein), formaldehyde and acetaldehyde (identified as important pollutants because they exceeded their conservative established limits in some retail stores, see Paper 0 for Appendix A for details) were also analyzed for further assurance that the selected control strategy did not

increase the concentration of these VOCs beyond a level where they will be considered as COCs.

5.2.2 Assessment of exposure control scenarios

Two alternative exposure control scenarios were evaluated using a time-averaged mass balance multi-contaminant model. This model was applied on a well-mixed single-zone area-normalized retail store (methodology described in Section 4.2.1), and best estimates for distributions of inputs across the retail sector were modeled in Monte Carlo simulations for multiple combinations of cities (six major cities were chosen to cover different climates as well as different outdoor air quality, Austin, Philadelphia, Minneapolis, Seattle, Los Angeles, Phoenix), seasons (winter and summer), store types (grocery and non-grocery), and time of the day (store-open and store-closed). Distribution inputs of the model and comparison of the parameters used and the measurements performed as part of the fieldwork (i.e., pollutant concentrations, ventilation parameters; Section 4.1 and Siegel et al., 2013) are detailed in the Method Section of Paper 3 (Appendix D).

The first control scenario calculated $PM_{2.5}$, acrolein, formaldehyde, and acetaldehyde concentrations based on three different ventilation strategies. These are the VRP (ventilation rate procedure), DCV-based CO_2 (demand controlled ventilation), and IAQP-based $PM_{2.5}$ (indoor air quality procedure), explained in Section 4.2.1. Additionally, two modified-VRP strategies, the VRP-C (differs from the VRP by reducing the ventilation rate at night to zero) and the VRP-NG (differs from the VRP by reducing the ventilation rate requirement in non-grocery stores by half, i.e., equal to that required for a grocery store) were also considered. The second control scenario supplements the first control scenario with increase of $PM_{2.5}$ filtration level.

Pollutant concentrations were then used in Monte Carlo simulations to quantify the DALY central estimates by sampling from uncertainty distributions of DALY factors reported in epidemiological or extrapolated from animal data studies (as described in section 4.2.2). Air exchange rates obtained from different ventilation strategies were used to calculate the energy consumption. The filtration energy cost was deduced from Section 5.1.2 (Specific aim 2.b).

This section is divided into three sub-sections. The first section explores the effects of different ventilation scenarios on the identified contaminants of concern (i.e., $PM_{2.5}$ and acrolein); formaldehyde and acetaldehyde were also analyzed to ensure that the selected control strategy did not increase the concentration of these pollutants beyond a level where they will be considered as COCs. The second section repeats the work of the first section but replaces the low $PM_{2.5}$ efficiency filter with a high $PM_{2.5}$ efficiency filter. The third section presents the DALYs lost and the energy results for the different considered scenarios and determines the optimal ventilation-filtration combination strategy that led to a balance between the DALYs lost and energy consumption.

Effect of ventilation scenarios on pollutants concentrations: The case of low $PM_{2.5}$ efficiency filter

Figure 16 displays distributions of $PM_{2.5}$ indoor concentrations for six major U.S. cities (Austin, Philadelphia, Minneapolis, Seattle, Los Angeles, Phoenix), two seasons (summer and winter), and are differentiated for store-open and store-closed periods, grocery and non-grocery retail types, and four different ventilation strategies.

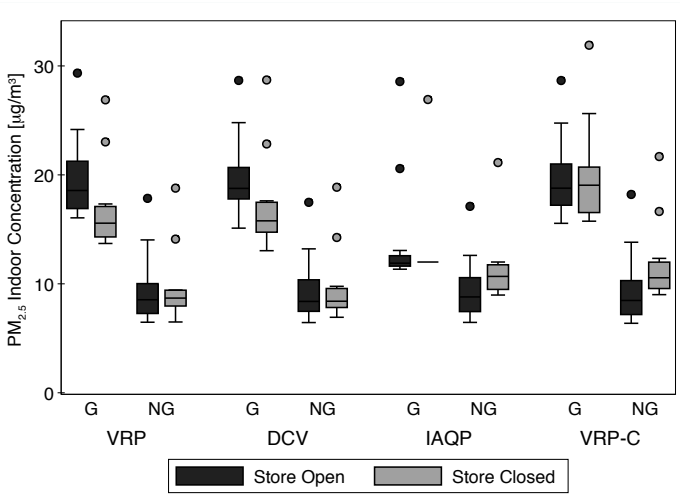

Figure 16: Box and whisker plots of modeled $PM_{2.5}$ indoor concentrations for six U.S. cities (Austin, Philadelphia, Minneapolis, Seattle, Los Angeles, Phoenix), two seasons (summer and winter) over store schedule, retail type (G: grocery, and NG: non-grocery), and different ventilation strategies.

$PM_{2.5}$ concentrations in grocery stores were higher than those reported for non-grocery stores for any ventilation scenario, mainly due to higher indoor emission sources. These results are consistent with those reported in Section 5.1.1 (Specific aim 1.b). Across all cities, median $PM_{2.5}$ indoor concentrations were comparable when applying the VRP or the DCV strategy. The VRP-C yielded higher concentrations when compared to the VRP for store-closed period because of lower ventilation rates (VRP-C assumes zero ventilation rate when the store is closed). Comparing IAQP with the other strategies revealed that IAQP resulted in lower median $PM_{2.5}$ concentration for grocery stores and higher median value for non-grocery stores. For all ventilation strategies, stores located in at least one city had indoor $PM_{2.5}$ concentrations higher than 12 $\mu g/m^3$ ($PM_{2.5}$ concentration of interest defined in ASHRAE Standard 62.1-2010). This finding is less expected when following the IAQP because the ventilation rates were calculated by setting $PM_{2.5}$ concentration equal to 12 $\mu g/m^3$. To explore this issue, Figure 17 shows

$PM_{2.5}$ concentrations and air exchange rate for two cities (Austin and Los Angeles), differentiated by store type and season.

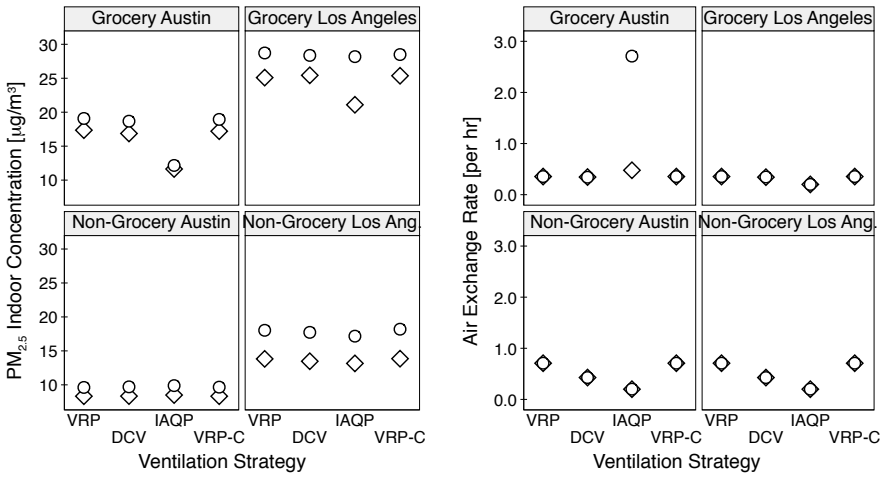

Figure 17: $PM_{2.5}$ indoor concentrations (left graph) and air exchange rate (right graph) for store-open period for two cities (Austin and Los Angeles), differentiated by store type and season. Hollow circles represent the summer season and hollow diamonds represent the winter season.

Results in Figure 17 show that in Austin, it was favorable to increase the air exchange rate in grocery stores to dilute indoor-generated $PM_{2.5}$ because the $PM_{2.5}$ indoor-to-outdoor ratios in these stores was larger than unity. Considering the IAQP, $PM_{2.5}$ concentration in grocery stores is Austin was maintained at 12 $\mu g/m^3$ with average air exchange rate of 1.6 per hour (350% increase than the VRP). In Los Angeles, the outdoor concentration was higher than the indoor concentration ($PM_{2.5}$ IO ratios lower than unity); this required the air exchange rate to be reduced to the minimum allowed to minimize admission of airborne pollution indoors. According to ASHRAE standard 62.1-2010, the minimum air exchange rate in an occupied zone cannot be zero; rather, it should be equal to the air exchange rate identified by a subjective assessment of IAQ identified based on occupants' survey responses collected for a similar type of building.

The minimum air exchange rate that matches the satisfaction of 80% of occupants in retail buildings corresponds to 0.2 per hour (Dutton et al., 2013). This minimum air exchange rate was not sufficient to keep $PM_{2.5}$ indoor concentration in grocery stores below 12 $\mu g/m^3$: the combination of high infiltration rate along with high outdoor concentration in these stores was likely the cause.

For non-grocery stores, following the IAQP, both cities (Austin and Los Angeles) were assigned the minimum air exchange of 0.2 per hour. In Austin, $PM_{2.5}$ indoor and outdoor concentrations were low (lower than 12 $\mu g/m^3$) and no extra dilution was needed. In Los Angeles, again, $PM_{2.5}$ indoor concentrations were greater than the desired concentration because of the elevated outdoor concentration and the high infiltration rate.

To explore the effects of these ventilation rates on acrolein, formaldehyde, and acetaldehyde concentrations, Monte Carlo simulations were repeated to calculate the concentrations of these pollutants, given the different ventilation strategies. Figure 18 shows the average indoor concentrations of formaldehyde, acetaldehyde, and acrolein during store-open period for six U.S. cities over retail type and different ventilation strategies.

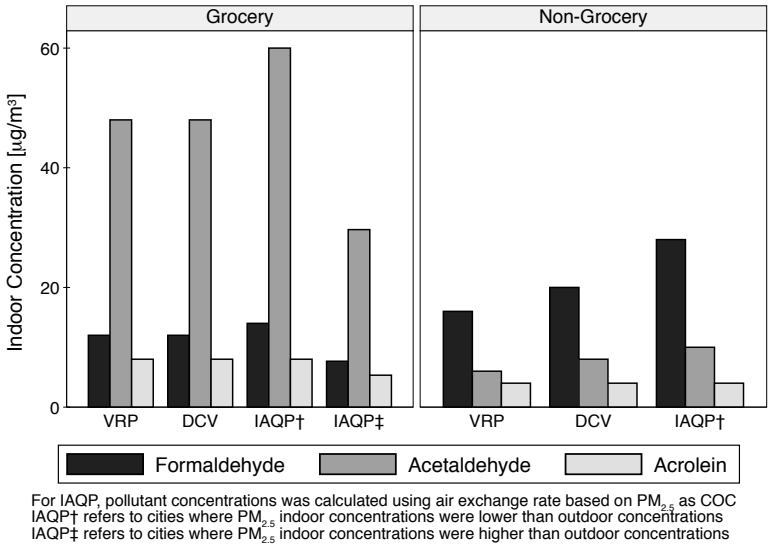

Figure 18: Bar charts of formaldehyde, acetaldehyde, and acrolein indoor concentrations for six U.S. cities over retail type and different ventilation strategies.

As shown in Figure 18, the IAQP applied to grocery stores with $PM_{2.5}$ indoor-to-outdoor ratios less than unity yielded the highest pollutant concentrations because the recommendation for these stores is to keep the air exchange rate to a minimum. Since formaldehyde, acetaldehyde, and acrolein are pollutants mainly generated indoors, lowering the air exchange increased these pollutants (formaldehyde and acrolein concentrations were 22% and acetaldehyde was 27% higher that those reported using the VRP). By analogy, applying the IAQP on grocery stores with $PM_{2.5}$ indoor-to-outdoor ratios larger than unity (i.e., increasing air exchange rate above that required by the VRP or the DCV) yielded the lowest formaldehyde, acetaldehyde, and acrolein concentrations (formaldehyde and acrolein concentrations were 31% and acetaldehyde was 38% lower than those reported for the VRP). For non-grocery stores, indoor concentrations were low; thus, air exchange rates were kept to a minimum, causing an increase of the concentrations of pollutants other than $PM_{2.5}$.

The question remains of how much the increase or decrease in pollutants concentrations translates into gain or loss in exposure benefits. Before answering this question, the next section explores the change in pollutants concentrations when using a high $PM_{2.5}$ efficiency filter.

Effect of ventilation scenarios on pollutants concentrations: The case of high $PM_{2.5}$ efficiency filter

This section investigates the impact of using a high $PM_{2.5}$ filter efficiency on pollutant concentrations: in the calculation, MERV 8 was switched to MERV 13. Figure 19 is a repeat of Figure 16 but with a high efficiency filter. Note that the ventilation rate determined by the IAQP-based $PM_{2.5}$ is the ventilation rate calculated to reduce $PM_{2.5}$ below 12 $\mu g/m^3$ with a high efficiency filter in place.

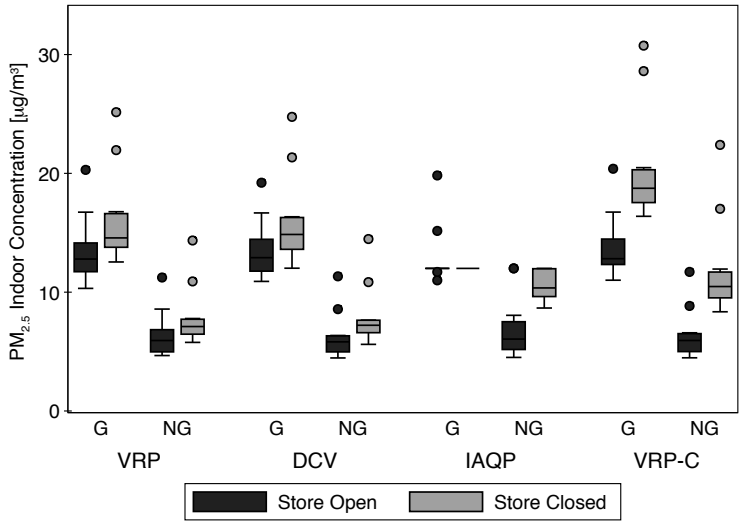

Figure 19: Box and whisker plots of modeled $PM_{2.5}$ indoor concentrations for six U.S. cities (Austin, Philadelphia, Minneapolis, Seattle, Los Angeles, Phoenix), two seasons (summer and winter) over store schedule, retail type (G: grocery, and NG: non-grocery), and different ventilation strategies.

As expected, using a high $PM_{2.5}$ efficiency filter (efficiency = 80%) decreased $PM_{2.5}$ indoor concentrations for all ventilation strategies; nonetheless, $PM_{2.5}$ indoor concentrations were higher than 12 $\mu g/m^3$ in at least one city for any of the ventilation strategies. For IAQP, the calculated air exchange rate was 0.2 per hour (minimum allowed) for all cities and for both retail types. This air exchange rate was sufficient to reduce elevated $PM_{2.5}$ concentrations in stores with $PM_{2.5}$ IO ratios larger than unity. For the stores with high outdoor concentrations, the high infiltration rates and emission rates present at these stores diminished the effect of high filtration efficiency, resulting in $PM_{2.5}$ concentrations higher than 12 $\mu g/m^3$.

Figure 20 is a repeat of Figure 18 but with using a high efficiency filter. The only difference between both figures is that the IAQP increases formaldehyde, acetaldehyde, and acrolein concentrations for both stores with $PM_{2.5}$ IO ratios higher and lower than unity, suggesting the danger of focusing on single pollutant strategies. As mentioned above, the calculated air exchange rate from the IAQP for all stores was 0.2 per hour, resulting in a decrease of dilution of these pollutants (i.e., higher formaldehyde, acetaldehyde, and acrolein concentrations).

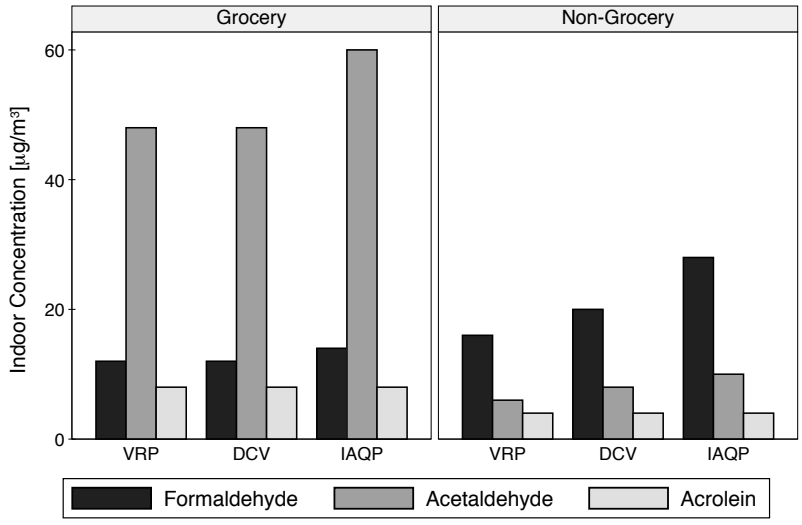

Figure 20: Bar charts for formaldehyde, acetaldehyde, and acrolein indoor concentrations for six U.S. cities over retail type and different ventilation strategies.

The next section puts these results in context of exposure and energy consumption.

Energy and exposure consequences of different ventilation and filtration scenarios

This section evaluates the control options of indoor exposures to pollutants by presenting the burden of disease (i.e., DALYs lost) attributable to each pollutant and the corresponding energy consumption. The health burden (i.e., number of DALYs lost per 100,000 persons) is only reported for employees in retail stores.

Figure 21 shows the DALYs lost from exposure to acetaldehyde, acrolein, formaldehyde, and $PM_{2.5}$ for different ventilation scenarios and filter efficiencies (top graph in Figure 21), and a zoom of DALYs attributed to exposure to $PM_{2.5}$ and acrolein (bottom graph in Figure 21). Note that the y-axis on the top figure is a log-scale. A new ventilation scenario was introduced in this figure (VRP-NG) that reduced the ventilation

rate required for non-grocery store to that required by the grocery store (as explained earlier in Figure 5).

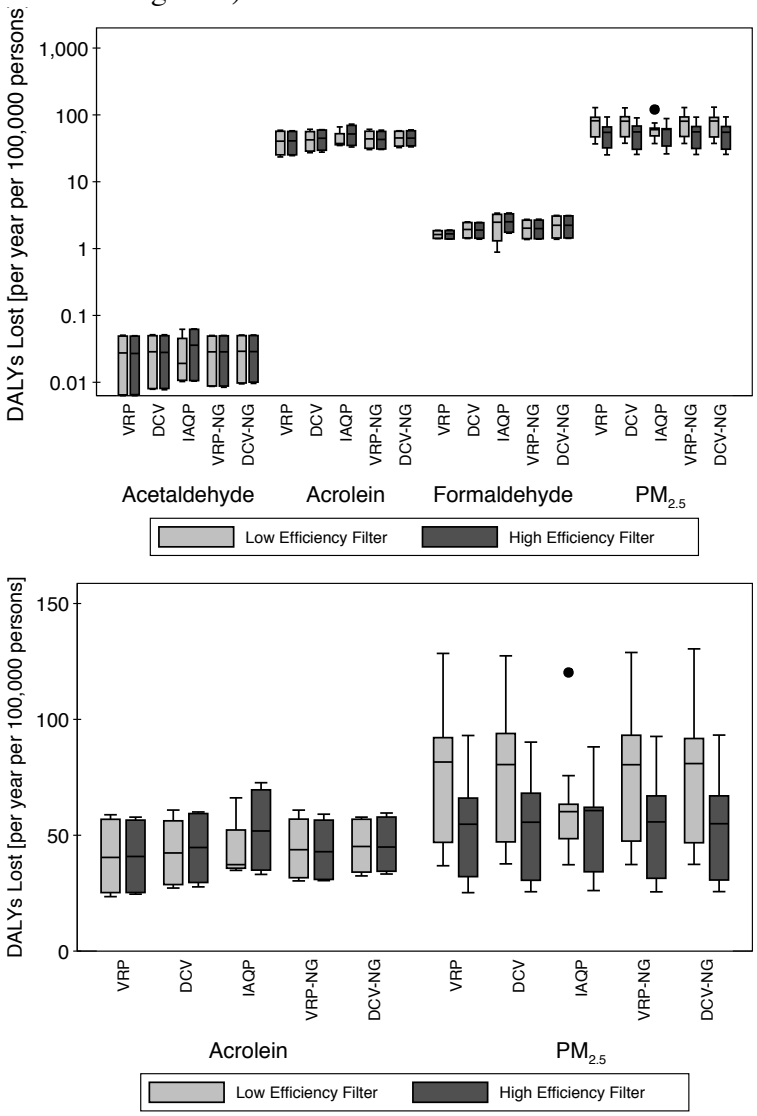

Figure 21: Box and whisker plots of modeled DALYs lost from exposure to acetaldehyde, acrolein, formaldehyde, and $PM_{2.5}$ for different ventilation scenarios and filter efficiencies (top figure). The y-axis is a log-scale. A zoom of DALYs attributed to exposure to $PM_{2.5}$ and acrolein is presented in the bottom figure.

The top figure in Figure 21 shows that the DALYs attributed to formaldehyde and acetaldehyde for all ventilation scenarios were relatively small when compared to those attributed to acrolein and $PM_{2.5}$, and the bottom figure shows that the DALYs attributed to $PM_{2.5}$ were 60% higher than those attributed to acrolein (e.g., median DALYs per 100,000 persons across all scenarios attributed to formaldehyde = 1.79; acetaldehyde = 0.019; acrolein = 37.4; $PM_{2.5}$ = 60 DALYs). As mentioned earlier, the ranking of pollutants using the DALY method should be interpreted with caution because the uncertainties of the exposure to each pollutant are fairly large, specifically for the selected VOCs.

Table 4 summarizes the total DALYs lost calculated by summing the DALYs from exposure to all pollutants. It is important to remember that the total DALYs were essentially the sum of DALYs from exposure to acrolein and $PM_{2.5}$. This table also shows the percentage change of different ventilation and filtration scenarios relative to the VRP/low $PM_{2.5}$ filter efficiency scenario.

Table 4: Summary of DALYs lost for different control scenarios and percentage change in DALYs lost relative to the VRP/low $PM_{2.5}$ filter efficiency scenario.

	DALYs Lost per year per 100,000 persons [% Change switching from VRP Low PM η]									
City	VRP Low PM η	DCV Low PM η	IAQP Low PM η	VRP-NG Low PM η	DCV-NG Low PM η	VRP High PM η	IAQP High PM η	DCV High PM η	VRP-NG High PM η	DCV-NG High PM η
Grocery										
Aus	147	152	100			124	132	127		
	(0%)	(3%)	(-32%)			(-15%)	(-10%)	(-14%)		
PA	152	150	118			125	130	129		
	(0%)	(-1%)	(-23%)			(-18%)	(-14%)	(-15%)		
LA	186	185	187			151	160	151		
	(0%)	(0.5%)	(0.4%)			(-19%)	(-14%)	(-19%)		
Mn	143	140	104			115	132	121		
	(0%)	(-2%)	(-27%)			(-20%)	(-8%)	(-16%)		
Ph	146	154	113			121	136	124		
	(0%)	(6%)	(-23%)			(-17%)	(-6%)	(-15%)		
Stl	160	160	132			127	133	132		
	(0%)	(-0.2%)	(-17%)			(-21%)	(-17%)	(-18%)		
Non-Grocery										
Aus	75	78	87	83	84	60	74	65	67	71
	(0%)	(4%)	(16%)	(11%)	(13%)	(-19%)	(-2%)	(-14%)	(-11%)	(-5%)
PA	74	81	88	82	86	59	71	64	65	68
	(0%)	(9%)	(20%)	(11%)	(16%)	(-20%)	(-3%)	(-14%)	(-12%)	(-8%)
LA	106	108	114	111	115	80	100	82	87	88
	(0%)	(2%)	(8%)	(5%)	(9%)	(-25%)	(-5%)	(-22%)	(-18%)	(-16%)
Mn	65	69	77	72	75	52	63	58	59	64
	(0%)	(6%)	(18%)	(10%)	(15%)	(-20%)	(-4%)	(-12%)	(-10%)	(-2%)
Ph	67	71	83	74	77	56	70	61	63	65
	(0%)	(5%)	(23%)	(10%)	(15%)	(-17%)	(3%)	(-9%)	(-7%)	(-3%)
Stl	74	79	88	82	85	59	74	64	64	67
	(0%)	(6%)	(18%)	(10%)	(14%)	(-21%)	(-1%)	(-15%)	(-14%)	(-11%)

Data in Table 4 show that the DALYs lost resulting from the IAQP with high efficiency filter were higher than those reported for IAQP with low efficiency filter for grocery stores. This finding is counter-intuitive. The reason is that using a high efficiency filter decreased $PM_{2.5}$ indoor concentration substantially so that the ventilation rate required was the minimum ventilation rate (i.e. 0.2 per hour). Setting the ventilation rate in grocery stores to a minimum level increased the concentrations of other pollutants

generated indoors; of particular interest is acrolein because it contributed significantly to the DALYs lost.

To assess the energy consumption of different ventilation and filtration scenarios, an energy model for a typical retail store was used to extract ventilation and fan electrical consumption for the six cities considered in this work (the model is explained in Section 4.2.5). Since the goal is to discern the impact of a specific ventilation scenario (i.e. specific air exchange rate) on energy and air quality, the fan-only mode where air exchange rate can be increased beyond what is defined in ASHRAE Standard 62.1-2010 when "free" cooling is provided by the low outdoor temperatures (i.e. economizer mode) was disabled.

The energy used by fans when a low $PM_{2.5}$ efficiency filter is installed was adjusted according to the results from Section 5.1.2 to account for the additional energy used by fans in case a high $PM_{2.5}$ efficiency filter was installed. The rooftop units were assumed to be units equipped with fan speed control (i.e., these units adjust the fan speed to maintain a constant flow rate). As established in Section 5.1.2, replacing MERV 8 filter with MERV 13 filter increased the fan power draw by 11%. Note that the results of the energy model showed that the fan electrical consumption contributed significantly (13%-57%, depending on the city and air exchange rate) to the total cooling cost (including ventilation conditioning and cost of distributing the supply air through the building). The modeled percentage contribution of the fan consumption to the total cooling cost for Austin was 19%, matching the results from fieldwork established in Section 5.1.2. Table 5 summarizes the electrical consumption by city for different ventilation and filtration scenarios

Table 5: Summary of electricity consumption for different control scenarios and percentage change in electricity consumption relative to the VRP/ low $PM_{2.5}$ filter efficiency scenario.

	Electricity Consumption-Cooling kWh x 1000 [% Change switching from VRP Low PM η]									
City	VRP Low PM η	DCV Low PM η	IAQP Low PM η	VRP-NG Low PM η	DCV-NG Low PM η	VRP High PM η	IAQP High PM η	DCV High PM η	VRP-NG High PM η	DCV-NG High PM η
Grocery										
Aus	175 (0%)	173 (-1%)	399 (128%)			186 (7%)	156 (-11%)	184 (6%)		
PA	81 (0%)	81 (0%)	168 (108%)			90 (11%)	87 (7%)	90 (11%)		
LA	62 (0%)	63 (1%)	75 (20%)			72 (16%)	85 (36%)	73 (17%)		
Mn	81 (0%)	81 (0%)	88 (9%)			90 (11%)	89 (10%)	90 (11%)		
Ph	179 (0%)	178 (-1%)	238 (33%)			193 (8%)	171 (-4%)	192 (7%)		
Stl	42 (0%)	43 (2%)	60 (43%)			52 (25%)	70 (68%)	53 (27%)		
Non-Grocery										
Aus	245 (0%)	191 (-22%)	145 (-41%)	175 (-29%)	155 (-37%)	256 (5%)	156 (-36%)	202 (-17%)	186 (-24%)	167 (-32%)
PA	94 (0%)	84 (-11%)	78 (-17%)	81 (-14%)	79 (-17%)	103 (10%)	87 (-8%)	93 (-1%)	90 (-4%)	88 (-7%)
LA	33 (0%)	56 (68%)	75 (126%)	62 (89%)	70 (113%)	43 (30%)	85 (156%)	66 (99%)	72 (119%)	80 (144%
Mn	86 (0%)	82 (-5%)	80 (-7%)	81 (-6%)	80 (-7%)	95 (11%)	89 (4%)	91 (5%)	90 (5%)	89 (4%)
Ph	238 (0%)	191 (-20%)	157 (-34%)	179 (-25%)	165 (-31%)	252 (6%)	171 (-28%)	205 (-14%)	193 (-19%)	178 (-25%)
Stl	18 (0%)	3 (-94%)	60 (226%)	42 (128%)	52 (186%)	29 (58%)	70 (284%)	46 (152%)	52 (186%)	63 (244%

Before identifying the optimal control strategy, it is important to mention that (1) lowering the air exchange in Los Angeles and Seattle increased electrical consumption because outdoor air conditions at the higher required ventilation rate helped in cooling the store, and (2) on average DCV offered no advantage over VRP in grocery stores from either energy or air quality perspectives; the air exchange specified for DCV is approximately equal to that specified for VRP.

Synthesizing the data displayed in Table 4 and Table 5 shows that for grocery stores located in cities where outdoor $PM_{2.5}$ concentration is low (median $PM_{2.5}$ outdoor concentration less than 12 $\mu g/m^3$, all sampled cities except Los Angles in our case),

adopting the IAQP with low efficiency $PM_{2.5}$ filter led to the highest health benefits (on average 26% DALYs less than the VRP). This gain in health benefits translated into doubling the air exchange rate (70%, on average, increase in electrical consumption of ventilation alone, $8,800 average increase in annual cost, with the annual increase in Austin being the highest at $25,000). For grocery stores located in Los Angeles, adopting the VRP with high $PM_{2.5}$ filter efficiency led to the highest health benefits, with electrical consumption equal to $1,100 (additional energy cost of higher efficiency filtration) more than that realized by the VRP with low efficiency filter.

For non-grocery stores, it is most beneficial from health perspective to adopt the VRP with high $PM_{2.5}$ efficiency (on average 20% less DALYs than the VRP with low efficiency filter), with electrical consumption equal to $1,200 (additional energy cost of higher efficiency filtration). Adopting the reduced VRP with high $PM_{2.5}$ efficiency filter instead of the VRP with high $PM_{2.5}$ efficiency filter led to sacrificing 8% of health benefits for half the air exchange rate ($1,700 decrease in annual cost).

A comparison between the ventilation rates prescribed by the VRP (ASHRAE Standard 62.1-2010) for grocery and non-grocery stores revealed that non-grocery stores are recommended higher ventilation rates than grocery stores (approximately double). Surprisingly, the results from the IAQP (also from ASHRAE Standard 62.1-2010), contradict the VRP results: for he IAQP, grocery stores have higher emissions and higher indoor concentrations, and thus naturally they are recommended higher ventilation rates (with the caveat that outdoor air concentration are low). A revision of ASHRAE Standard 62.1 is warranted given the new information available about retail stores; ventilation rates in grocery stores should be higher.

Pollutant control ventilation strategy. For all cities and for all retail types, adopting a tailored ventilation/filtration strategy (specific to retail type and location) led

to substantial decrease in DALYs lost. A tailored control strategy should take into consideration all contaminants of concern: a single pollutant-approach to determine ventilation rates may overlook significant exposures from other pollutants generated from a different source (indoors vs. outdoors). Although the indoor air quality procedure (IAQP) specified in ASHRAE Standard 62.1-2010 requires the determination of the minimum ventilation rate to keep each contaminant of concern below its established limits, this method converges to the single pollutant-approach as it requires the engineer to choose the larger ventilation rate determined by the different COCs. Thereby ignoring the fact that some pollutants (e.g. $PM_{2.5}$) can be generated outdoors and keeping the ventilation rate to a minimum might be more advantageous from both exposure and energy perspectives. Further, the established limits for each pollutant vary widely and it might not be feasible to calculate the ventilation rate that will drive the pollutant below its limit. For example, acrolein levels indoors and outdoors were higher than the REL and depending on ventilation alone will not drive acrolein below its REL. As demonstrated in results shown in Table 4 and Table 5, following the VRP or DCV specified in ASHRAE Standard 62.1-2010 might not be the optimal solution from both energy and exposure perspectives.

Instead of relying on a single pollutant approach (IAQP-ASHRAE Standard 62.1-2010) or a prescribed ventilation number (VRP-ASHRAE Standard 62.1-2010) to determine ventilation rates, a proposed approach is the pollutant exposure control ventilation (PECV) strategy. The PECV finds an optimal ventilation rate based on weighing the exposures of different contaminants of concern, following the same DALY approach for the contaminants of concern identified and presented in this work. However, ventilation rates are not limited only to those recommended by the standards. An example of applying PECV on a grocery store located in Los Angeles is shown in Figure 22.

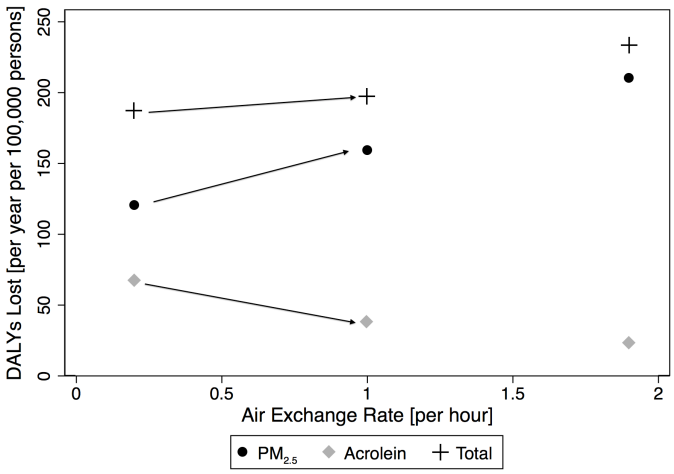

Figure 22: DALYs lost function of air exchange rate for a grocery store in Los Angeles.

Figure 22 shows that the total DALYs lost for air exchange rate between 0.2 and 1 per hour are comparable. The low air exchange rate (i.e., 0.2 per hour) increased DALYs lost from acrolein and decreased those for $PM_{2.5}$; the opposite happened when applying the higher air exchange rate (i.e., 1 per hour). Thus, the proposed strategy offers the flexibility to choose the air exchange rate that will lead to lower energy consumption. As mentioned previously, in Los Angeles, increasing the air exchange rate is beneficial to decrease the energy consumption because the conditions of the outdoor air are favorable for free cooling. Note that an additional decrease in DALYs lost can be obtained if a high efficiency filter is used, but this translates into additional cost as mentioned previously. The suggested PECV strategy depends mainly on the uncertainties associated with the estimated exposures (i.e. DALYs lost). The uncertainty bounds on exposure to some of the pollutants were higher than one order of magnitude. Besides, there were limitations associated with the use of ambient rather than indoor epidemiological studies, assuming linear relationship between exposure to acrolein and formaldehyde and health effects (i.e., eliminating the threshold effect), excluding health effects with incomplete

quantitative information (i.e., might be excluding effects from important pollutants such as ultrafine and SVOCs). However, as those available health factors and the most recent scientific knowledge are used, the order of magnitude of the results should be sufficiently reliable for prioritizing pollutant control strategies.

Chapter 6: Conclusion

The work established in this dissertation proposes a novel idea of exposure-based pollutant control strategy, presents a complete understanding of particulate matter characteristics including indoor emission rates in retail stores, introduces the energy and air quality consequences of different efficiency filters installed in commercial buildings, estimates the health burden from exposure to pollutants in retail stores, and tests the feasibility of different control strategies proposed by energy standards. Full results of these investigations are contained in Appendices A through D at the end of this dissertation. The major findings of this study can be summarized as follows:

- The highest $PM_{2.5}$ concentrations and emission rates were observed at grocery stores, due to the presence of intense cooking activities and high foot traffic. Among the stores tested in this work, the median contribution to $PM_{2.5}$ concentrations from indoor sources (vs. outdoors) was 53%.
- $PM_{2.5}$ and acrolein are the two primary COCs, contributing to 160 disability-adjusted-life-years (DALYs) per 100,000 persons annually; employees in grocery stores mainly drove this burden.
- Exposures to indoor air can only be reduced effectively if we account for all pollutant influencers: indoor and outdoor sources (or retail type and location), infiltration, ventilation, and filtration.
- Replacing MERV 8 with MERV 13/14 resulted in higher energy consumption (2-4%) during cooling mode for units equipped with and without fan speed control, energy savings during fan-only mode (8-13%) in units without fan speed control, and increased energy consumption in fan-only mode (11-18%) in the unit with fan

speed control. Energy consumption increases was offset by improvement in clean-air-delivery-rate, especially for $PM_{2.5}$ (2.9-3.8 times increase going from MERV 8 to MERV 13/14), with larger benefits achieved for the unit with fan speed control.

- The considered ventilation/filtration scenarios were able to provide substantial reductions in the health risks (19-26% decrease in DALYs), and the magnitude of the reductions depended on the ventilation/filtration scenario, the retail type, and the city.
- For cities where outdoor $PM_{2.5}$ concentration is low, adopting the IAQP with low efficiency $PM_{2.5}$ filter in grocery stores and the VRP with high $PM_{2.5}$ efficiency for non-grocery stores yielded the highest health benefits. For cities with high outdoor $PM_{2.5}$ concentration, adopting the VRP with high $PM_{2.5}$ efficiency for both store types yielded the highest health benefits. DCV was not significantly different than the VRP from both health burden and energy usage standpoint.
- A proposed strategy as alternative to the ventilation strategies recommended by the standards is the pollutant exposure control ventilation (PECV) strategy. This strategy is based on (1) weighing the exposures of different contaminants of concern found in retail buildings, (2) identifying the range of ventilation rates that lead to low DALYs lost, and (3) choosing the optimal ventilation rate that leads to energy usage savings in the climate considered.

The main goal of this work was to provide quantitative information about exposures to contaminants of concern and efficient reductions of these exposures in retail buildings. The results from this study can be used for developing indoor air quality guidelines and efficient energy resources allocation of pollutant control strategies for better health and energy usage.

Appendix A

Paper 0: Ventilation and Indoor Air Quality in Retail Stores: A Critical Review (RP-1596)

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(Submitted to *ASHRAE HVAC&R Research*)

ABSTRACT

Identifying air pollutants that pose potential adverse health exposures in retail stores will facilitate exposure mitigation. Assessing the role of ventilation in mitigating this exposure is important to understand the energy implications of maintaining acceptable indoor air quality. In this work, we summarize results from 28 papers that report ventilation rates and/or pollutant concentrations in retail stores. These results were compared to available standards as well as data collected in non-retail environments. The findings of this review are: (1) half of the stores tested met/exceeded ASHRAE Standard 62.1-2010 (or California Code of Regulations Title 24-2010) for ventilation; (2) $PM_{2.5}$, acrolein, formaldehyde, and acetaldehyde exceeded their established most conservative limits/reference exposures for a few of the stores tested in the U.S.; outside the U.S., researchers reported PM_{10}, benzene and trichloroethylene as additional pollutants found at concentrations that exceeded their limits; (3) alternative control methods would be more effective and possibly more economical than ventilation, (4) meeting or exceeding the ventilation requirements does not necessarily negate the presence of pollutants above their suggested limits; and (5) using disability-adjusted-life-years (DALYs) as a metric of disease burden, two pollutants were identified as priority hazards in retail stores: $PM_{2.5}$ and acrolein. Control strategies should focus on decreasing exposure of retail employees to these pollutants generated indoors or infiltrated from outdoors.

INTRODUCTION

Among commercial buildings, retail buildings rank second in energy consumption and greenhouse gas emissions, after office buildings (Zhang et al., 2011, Kennedy et al., 2012). Within retail stores, HVAC systems have been identified as one of the largest energy users, accounting for 28% to 48% of the total building energy use (Alhafi, 2012). The retail sector employs 15 million workers, approximately 10% of the U.S. workforce (NRF, 2010), making the indoor air quality (IAQ) of retail buildings an important occupational exposure consideration. Additionally, the average American above the age of fifteen spends 0.48 hours per day purchasing goods and groceries (ATUS, 2011), further highlighting the importance of IAQ in these environments. Ventilation is generally regarded as the most effective measure to improve IAQ when source control is not an option. However, there is a tradeoff between increasing ventilation to improve IAQ and saving fan and conditioning energy associated with ventilation. A sustainable approach to retail environment ventilation must also consider the occupant perceptions and health exposures for retail workers and customers.

Despite the importance of retail buildings, there have been no published reviews focusing on contaminants of concern in retail buildings, and whether ventilation can be used to control these contaminants below their reference or regulatory limits. Specifically this literature review addresses the following questions:

- How do ventilation rates measured in retail stores compare to standards?
- What are the dominant pollutants in different types of retail spaces?
- How do different ventilation measurement methods affect the reported results?
- What are the associations between ventilation and IAQ?
- What are the contaminants of concern from an exposure perspective?

This paper addresses these questions by summarizing ventilation rates and pollutants concentrations reported in the relevant literature. These data are also compared to the available standards and guidelines as well as data collected in non-retail environments. The reported concentrations are contextualized with information about sampling and analysis approaches where possible. Furthermore, this investigation also discusses the available data linking ventilation and measured pollutant concentrations, and finally evaluates contaminants of concern found in retail environments through a disability-adjusted-life-years (DALYs) loss assessment. The central objective of this approach is to provide baseline data for comparison of results from the ASHRAE RP-1596 project conducted between 2011 and 2013 (Siegel et al., 2013). The goal of the RP-1596 project was to characterize the ventilation, indoor air quality, and occupant perception in 14 retail stores located in Texas and Pennsylvania and this literature review serves as background to that investigation.

METHODOLOGY AND SEARCH CRITERIA

The literature review systematically searched the following online databases: the ISI Web of science, Compendex, ScienceDirect, government reports, and Google Scholar. The keywords used to guide the search included various combinations of 'energy', 'ventilation', 'infiltration', 'air exchange rate', 'outdoor air rate', 'IAQ', multiple pollutants (i.e. 'VOCs', 'SVOC', 'aldehydes', 'particulate matter', 'bacteria', 'fungi', 'radon', 'carbon monoxide', 'nitrogen dioxide', 'sulfur dioxide') with 'retail', 'mall', 'supermarket', "store", "shopping", and 'commercial building'. Other types of buildings were also included in the search for comparison purposes. Excluding those references included from other building types, papers had to report measured ventilation, indoor air quality, and/or occupant survey results for retail buildings. Records for approximately 118 papers were deemed to be the most relevant to the research questions

of interest, and of those 28 publications specifically reported measurements in retail spaces. After the articles were selected they were sorted into one of the following categories: (i) Ventilation; (ii) Particulate matter; (iii) VOCs; and (iv) Other pollutants (i.e. Ozone, SVOCs, microbiological contaminants, radon, CO, NO_X, and SO_X).

SUMMARY OF LITERATURE REVIEW

The existing studies with reported ventilation raters and/or IAQ measurements provide data for retail ventilation rates, particulate matter, VOCs, and other pollutant concentrations. Some of the studies included investigations that considered the impact of ventilation on these concentrations. The studies that reported ventilation and/or pollutant concentrations in retail stores are categorized and summarized in Table 1.

Table 1. Summary of Measurement Studies in Retail Stores

Study	N	Store Type[1]	Ventilation/ Pollutant	Sampling Method	Location
Shaw 1981	10	9 Supermarket 1 Shopping mall	Infiltration	Pressure difference	
Yu et al. 1996	10	Malls	Radon	Tsivoglou procedure (TN-WL-MS filter)	Hong Kong
Grimsrud et al. 1999	1	General merchandise	$PM_{2.5}$	Continuous particle counting (Atcor APC)	Minnesota, U.S.
Marley 2000	1	General merchandise	Radon	Continuous radon level (NITON Rad 7)	Northampton-shire, UK
Chao and Chan 2001	2	Computer	Air exchange rate	Tracer gas decay (SF_6)	Hong Kong
			43 VOCs	4-hr canister sample	
Kim et al. 2001	5	2 Department 3 Perfume	15 VOCs	2-hr sorbent tubes	Birmingham, U.S.
Li et al. 2001	9	Malls	PM_{10}	Continuous particle mass (TSI Dusttrak)	Hong Kong
			CO	Non-dispersive infrared (Thermo Electron Model 48 Analyzer)	
			Bacteria	Culturing	

Rea et al. 2001	1	Retail (unspecified)	$PM_{2.5}$	Integrated gravimetric particle mass (PEM, MSP)	Baltimore, U.S.
Sakai et al. 2002	1	Department	PM_{10}, PPAH	Continuous particle mass (TSI Dusttrak, AQM)	Tokyo, Japan
Hartman et al. 2004	5	2 Furniture 3 Electronics	SVOC	Polyurethane foam (PUF) plugs	Zurich, Switzerland
Wang et al. 2004	8	Supermarket/office	Radon	Passive detector (SSNTD of CR-39) Active detector-continuous radon level (SNC 1027)	China
Wargocki et al. 2004	1	Department	Air exchange rate	Outdoor air duct pressure differential	Germany
Liu et al. 2004	5	General merchandise	PM_{10}, $PM_{2.5}$, PM_1	Continuous particle mass (Dustmate)	Beijing, China
Tang et al. 2005	7	2 Supermarkets 4 Department 1 Book	11 VOCs	30-min sorbent tubes	Guangzhou, China
Hotchi et al. 2006	1	General merchandise	Air exchange rate	Tracer gas decay (PMCH)	San Francisco, U.S.
			34 VOCs	30-min sorbent and DNPH tubes	
Loh et al. 2006	113	17 Hardware 12 Multipurpose 16 Grocery 8 Drugstores 14 Sporting goods 11 Furniture 16 Houseware 10 Department 9 Electronics	16 VOCs	1 to 3-hr composite sorbent and DNPH tubes	Boston, U.S.
Lee and Hsu 2007	12	Photocopy	$PM_{2.5}$, ultrafine particles (<0.1 μm)	Continuous particle mass (Dusttrak), particle sizer (SMPS/CPC Model 3025A & APS)	Taiwan
			5 VOCs	2-hr sorbent tubes	
			Ozone	Ecotech ML9810B	
Bruno et al. 2008	4	2 Supermarkets 2 Pharmacies	15 VOCs	24-hr diffusive samplers	Bari, Italy
Eklund et al. 2008	4	1 Jeweler 1 Clothing rental 1 Optician	28 VOCs	8-hr canister	New Jersey, U.S.

		1 Video rental			
Tringe et al. 2008	2	Malls	Bacteria	DNA sequencing and cloning	Singapore
Caselli et al. 2009	2	Printing	23 VOCs	24-hr diffusive samplers	Bari, Italy
Maskey et al. 2011	1	Mall	Chemical analysis of particles 1–2.5 μm and 2.5–10 μm	X-ray Microanalysis (low-Z particle Electron probe)	Korea
Bennett et al. 2011	10	2 Grocery 1 Florist 1 Cabinet 1 Water filters 1 Art supplies 1 Bookstore 2 Sporting goods 1 Convenience/gas station	Air exchange rate	Tracer gas decay (SF_6) Steady-state PFT CO2 equilibrium	California, U.S.
			PM_{10}, $PM_{2.5}$, sub-3micron particles (<3 μm)	Integrated gravimetric particle mass (Harvard cascade impactors), Continuous particle counting (Met One Model 237B, TSI CPC Model 3781)	
Grimsrud et al. 2011	3	General merchandise	Air exchange rate	Tracer gas decay (CO_2)	Minnesota, Florida, Maryland, U.S.
			20 VOCs	48-hr formaldehyde and passive organic vapor badges	
Wu et al. 2011	9	2 Grocery 7 Retail (unspecified)	Air exchange rate	Tracer gas decay (SF_6)	California, U.S.
			30 VOCs	4-hr sorbent and DNPH tubes	
			SVOC (DEP)	Sorbent tubes	
Brown et al. 2012	2	1 Bookstore 1 Grocery	$PM_{2.5}$, submicron particles (<1 μm)	Continuous particle mass (TSI Dusttrak) number (TSI P-Trak Model 8525)	Atlanta, U.S.
Chan et al. 2012	5	2 Grocery 3 Furniture	Air exchange rate	Tracer gas decay (SF_6)	California, U.S.
			PM_{10}, $PM_{2.5}$	Integrated gravimetric particle mass (SKC PEM)	
			46 VOCs	1&2-hr sorbent, DNPH and PFP tubes	
			Ozone	2B Tech Model 205	

Bennett et al., 2012	10	2 Grocery 1 Florist 1 Cabinet 1 Water filters 1 Art supplies 1 Bookstore 2 Sporting goods 1 Convenience/gas station	Air exchange rate, mechanical air exchange rate	Tracer gas decay (SF6), calibrated variable speed fan with integral airflow meter	California, U.S.
Dong et al. 2013	1	General merchandise	$PM_{2.5}$	Integrated gravimetric particle mass (TH-16A)	Jinan, China

[1]As characterized by the authors.

1. Ventilation

The ventilation parameters measured in retail stores, comparisons with standards and non-retail environments, and potential implications of sampling approaches are discussed below.

1.1 Ventilation: Measurements in Retail Stores

The studies that reported ventilation rates in retail stores typically measured the total outdoor air exchange rates (Chao and Chan 2001; Wargocki et al. 2004; Hotchi et al. 2006; Lee and Hsu 2007; Bennett et al. 2011, 2012; Chan et al. 2012). In addition, two studies measured the components of total outdoor air exchange rate: mechanical ventilation rate supplied by the roof top units (RTUs) (Bennett et al. 2011, 2012), and the infiltration rate through retail entrance doors (Shaw 1981). Table 1 includes the summary on these two studies that examined ventilation rates in more details.

Total outdoor air exchange rates (AER). The required minimum ventilation rates provided in the ASHRAE standard 62.1 are determined based on space type, and occupancy density. Consequently, the standard separates minimum ventilation rate for grocery stores from retail since the building activity, and function of grocery stores that offer mostly food products have a different type of potential internal contaminant sources than the potential sources in other retail stores. Six studies have measured AER in retail

stores in the U.S., Europe and Asia. Two of these studies investigated the impact of changing AER on measured indoor pollutant concentrations and/or environmental perceptions (Wargocki et al. 2004; Hotchi et al. 2006). The number of existing studies is limited, so the current AER findings are limited to specific case studies. The average ventilation rate found in grocery stores was 1.6 ± 1.3 L/s·m^2_{floor} (0.32 ± 0.26 cfm/ft^2_{floor}) (Bennett et al. 2011; Chan et al. 2012). The average ventilation rate found in retail stores other than groceries was 1.45 ± 1.3 L/s·m^2_{floor} (0.29 ± 0.26 cfm/ft^2_{floor}) (Chao and Chan 2001; Wargocki et al. 2004; Hotchi et al. 2006; Bennett et al. 2012; Chan et al. 2012). Most of these studies used tracer gas measurements to establish time-averaged AERs, and, therefore, the relatively high standard AER deviations can indicate a relatively large temporal AER variability. A detailed description of the individual studies is provided below.

Currently, the most comprehensive study to report AER in retail stores examined typical U.S. commercial buildings in California (Bennett et al. 2012). The study included a total of 37 buildings; ten of which were retail establishments. Bennett et al. (2012) used SF_6 as the tracer gas. The average of the measured retail AERs was 1/h ± 0.7/h, which corresponds to 1 ± 0.5 L/s·m^2_{floor} (0.2 ± 0.11 cfm/ft^2_{floor}). The highest air exchange rate (2.33/h) was measured at a retail store that sold water filtration supplies. According to the authors, the infiltration rates were high since the doors were open most of the time, and contributed to the high air exchange rate observed.

In a recent pilot study that aimed to quantify the contaminant emission rates in retail stores, the outdoor air exchange rates were measured in two grocery stores and three furniture stores in northern California using SF_6 as the tracer gas (Chan et al. 2012). In the two-day-long measurement, one grocery store showed an average AER of 0.6 - 0.7/h (1.4 L/s·m^2_{floor} , 0.28 cfm/ft^2_{floor}) while the other grocery store showed a higher

average AER of 1.8/h - 2.0/h (3.5 L/s·m^2_{floor} , 0.7 cfm/ft^2_{floor}). The furniture stores were naturally ventilated and their air exchange rate ranged from 0.3 to 3.9/h (0.4-1.3 L/s·m^2_{floor} , 0.08-0.26 cfm/ft^2_{floor}). The doors in furniture stores were kept open most of the time and the entrance and loading doors in one of these stores had a large opening area.

Another study evaluated the impact of reduced ventilation rates on indoor air quality to develop a peak load shedding strategy for a general merchandise store located in California (Hotchi et al. 2006). The study measured the ventilation rate during regular operation before shutting down a certain number of RTUs. Their study used perfluoromethylcyclohexane (PMCH) as the tracer gas. The AERs were 0.71/h and 0.95/h measured in two separate experiments on two different days.

A European study examined the relationship between AERs and occupant perception of air quality in seven buildings, one of which was a department store (Wargocki et al. 2004). In their study, the ventilation measurements were conducted under constant air volume (CAV) operation of the air handling unit. AERs were determined from the pressure differential in the outdoor air supply duct under the assumption that the building was positively pressurized, so the potential infiltration rates were neglected. The total air exchange rates or floor area were not reported. The measured AER normalized by the floor area was 3.5 L/s·m^2_{floor} (0.7 cfm/ft^2_{floor}), and constituted 50% of the total airflow supplied to the building.

Chao and Chan (2001) investigated the relationship between typical AERs and VOCs concentrations in a total of 20 buildings in Hong Kong; of which two were computer shopping centers in two different shopping malls. The study used the SF_6 tracer gas decay method to measure the AER. The two computer shopping centers had the same AER of 0.9/h, which corresponds to 0.5 L/s·m^2_{floor} (0.1 cfm/ft^2_{floor}). The reported AER was

lower than the AERs in offices (1.69/h) and public buildings (2.17/h) measured by the same study.

Lee and Hsu (2007) reported the air exchange rates for 12 photocopy centers located in Taiwan, which had an average AER of 7.34/h. The study did not mention how the air exchange rate was measured.

Mechanical system ventilation rates. Only one study measured the mechanical ventilation rates in retail stores (Bennett et al. 2012). Bennett et al. (2012) reported that on average 71% of total AER was supplied mechanically. This averaged percentage for mechanically supplied air included buildings with 100% of outdoor air brought in through the HVAC system and excluding buildings with doors open most of the time and the buildings that have no mechanical ventilation. Therefore, their study indicated that even in the fully mechanically ventilated buildings, unintentional ventilation, such as infiltration, makes a significant contribution to the total AERs.

Infiltration rates through entrance doors. Retail stores are usually assumed to have a tight building envelope. However, Shaw (1981) tested air leakage in ten retail stores and found that the range of infiltration rates in these stores was greater than other building types: double the infiltration rates measured in school and quadruple the infiltration rate measured in high-rise office buildings. Although, retail ventilation systems are often designed to minimize air infiltration by pressurizing the building interior, air infiltration through door usage in entrance/exit lobbies and loading docks make a significant contribution to the whole building air leakage rate (McGowan et al. 2006). The infiltration rates through entrance doors in standalone retail and strip mall buildings have been identified as one of the significant contributors to the total energy use. For example, Bennett et al. (2012) found that buildings with the doors kept open had an average air exchange rate of 2/h, which was significantly greater than the rate of 1/h

that was found when the doors were closed. To decrease infiltration through entrance doors, ASHRAE Standard 90.1-2010 requires the use of vestibules. For a prototype well-sealed standalone retail building, Cho et al. (2010) estimated that the infiltration rate for peak hours was 3,375 m^3/h (1,986 cfm) and for off-peak hours was 442 m^3/h (260 cfm) through doors with vestibules. The infiltration rate was 1.5 times the rates observed for doors without vestibules. Cho et al. (2010) also estimated potential national weighted-average site energy saving from the use of vestibules to be 2.4% and 5.6% for standalone retail and strip mall buildings respectively, higher than any other type of commercial buildings except restaurants.

1.2 Ventilation: Comparison to Standards

Recommended outdoor ventilation rates for different building types are provided in several standards including the American Society of Heating, Refrigerating, and Air Conditioning (ASHRAE) Standard 62.1-2010, the European Standard CEN CR 1752-1998, and the California Code of Regulations Title 24-2010. ASHRAE Standard 62.1-2010 and CEN CR 1752-1998 define two required ventilation components. One ventilation component is based on the occupancy rates and the other is based on the floor area. The differentiation between the occupancy and the area requirements provide flexibility when using occupancy-based control, such as demand control ventilation with CO_2 sensors. While the California Code of Regulations Title 24-2010 recommends outdoor airflow rate no less than the larger of the ventilation rate per conditioned of floor area or 15 cfm per person. Table 2 contains a summary of the required outdoor ventilation rate for the different standards.

Table 2. Comparison of Ventilation Standards Requirements for Retail Stores and Supermarkets

Standards	People Outdoor Air Rate, $L/s \cdot m^2_{floor}$ (cfm/ft^2_{floor})	Area Outdoor Air Rate, $L/s \cdot m^2_{floor}$ (cfm/ft^2_{floor})	Combined outdoor air rate, $L/s \cdot m^2_{floor}$ (cfm/ft^2_{floor})
ASHRAE 62.1 (2010)			
Retail sales	0.571(0.11)	0.6(0.12)	1.21(0.23)
Supermarket	0.31(0.06)	0.3(0.06)	0.61(0.12)
CEN CR 1752 (1998)			
Low polluted building			
Type A[2]	2.1(0.4)	2(0.38)	4.1(0.78)
Type B[2]	1.5(0.29)	1.4(0.27)	2.9(0.55)
Type C[2]	0.9(0.17)	0.8(0.15)	1.7(0.32)
Non-low polluted building			
Type A[2]	2.1(0.4)	3(0.57)	5.1(0.97)
Type B[2]	1.5(0.29)	2.1(0.4)	3.6(0.69)
Type C[2]	0.9(0.17)	1.2(0.23)	2.1(0.4)
California Code of Regulations Title 24 (2010)	2.6[3](0.5)	1(0.2)	Max(people rate , conditioned area rate)[4]

[1] To make a comparison of total outdoor air rate requirements, these values are multiplied by the occupancy density per floor area (retail 15 people/ 100m2, supermarket 8 people/ 100m2) provided in ASHRAE Standard 62.1-2010.

[2]Type A corresponds to 15% of occupants dissatisfied; Type B corresponds to 20% of occupants dissatisfied; Type C corresponds to 30% occupants dissatisfied.

[3] From Section 1004 Occupant Load in 2010 California Building Code Title 24, Part 2, the space with a design occupant density is 60 ft2/ person. Since the space is without fixed seating, the default occupant density will be one half of the maximum occupant load that is 30 ft2 /person.

[4] California Title 24 recommends the outdoor air rate to be no less than the larger of people outdoor rate and area outdoor rate.

Table 2 shows that ASHRAE Standard 62.1-2010 recommends a combined design ventilation rate for general retail sales double of that required for grocery stores. The European Standard CEN CR 1752-1998 differentiates between low polluted buildings and non-low polluted buildings. For both cases, the European Standard requires higher outdoor ventilation rates than the ventilation rates required by ASHRAE Standard 62.1-2010. In the California Code of Regulations Title 24-2010, the recommended ventilation rate based on the floor area is between the values recommended by ASHRAE for retail non-grocery stores and grocery stores; however, when considering the ventilation rate based on the occupancy rates, the standard Title 24-2010 requires higher outdoor ventilation rate than the rates required by ASHRAE Standard 62.1-2010.

For grocery stores, the range of ventilation rates in the stores reported by Chan et al. (2012) and Bennett et al. (2011) met the minimum performance required by the ASHRAE Standard 62.1-2010 and the per floor area requirement in California Code of Regulations Title 24-2010. For the non-grocery stores, two of the eight retail stores measured by Bennett et al. (2011) and two of the three retail stores measured by Chan et al. (2012) and the store measured by Hotchi et al. (2006) met both ASHRAE Standard 62.1-2010 and the floor area minimum required in Title 24 standard. One retail store measured by Chan et al. (2012) met the ventilation rate per floor area recommended in Title 24 standard. For the retail stores measured outside the U.S., measurement reported by Wargocki et al. (2004) met both standards requirements for the U.S. and the European Standard for type C buildings (30% of occupants dissatisfied). The measurements reported by Chao and Chan (2001) in Hong Kong did not meet the requirements of both U.S. standards.

1.3 Ventilation: Comparison to Other Environments

In order to put retail environments in context, Figure 1 shows a summary of air exchange rates found in literature for various environments. As in all similar plots in this paper, the bottom of the box indicates the 25th percentile; the horizontal line indicates the median and the top of the box the 75th percentile. The whiskers indicate the data range within 1.5 times the interquartile range of the 25th and 75th percentile. If present, any filled circles are outliers.

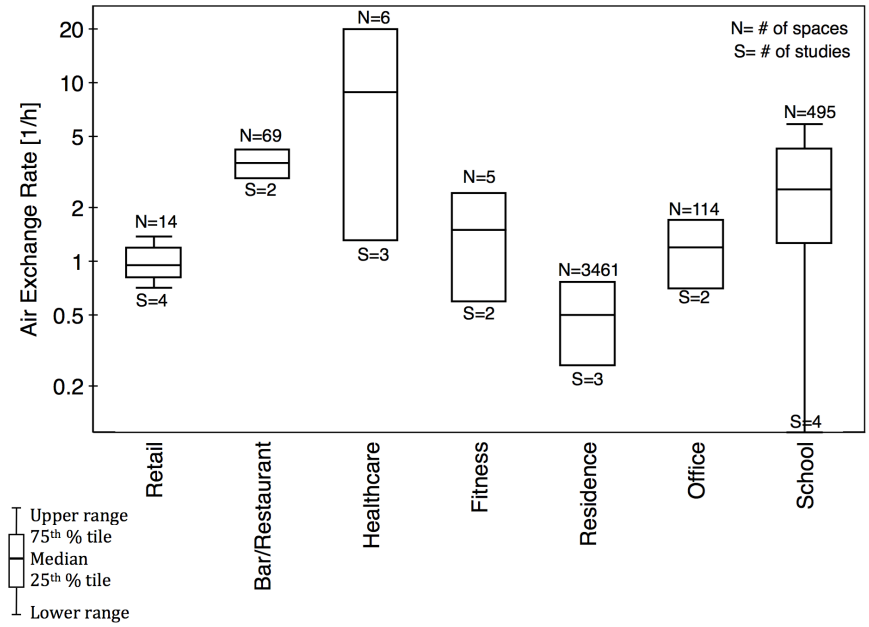

Figure 1. Summary of air exchange rate for different environments (N=number of spaces, S = number of studies). Data for retail stores are from: Chao and Chan 2001; Hotchi et al. 2006; Bennett et al. 2011; Chan et al. 2012. Data for bars/restaurants are from: Bennett et al. 2011; Bohac et al. 2012. Data for healthcare facilities are from: Qian et al. 2010; Bennett et al. 2011; Knibbs et al. 2011. Data for fitness are from: Bennett et al. 2011; Chao and Chan 2001. Data for residences are from: Murray and Burmaster, 1995; Offermann 2009; Yamamoto et al. 2010. Data for offices are from: Chao and Chan 2001; Parthasarathy et al. 2013(analyzed values from the BASE Study: Persily et al. 2005). Data for schools are from: Godwin and Batterman, 2007; Apte et al. 2012 (including results from 2 studies); Ramalho et al. 2012.

These investigations showed that the ventilation rates in retail stores are generally less than bars/restaurants and healthcare facilities and comparable to other environments (i.e. fitness centers, residences, offices, and schools). The high ventilation rates found in bars and restaurants may be driven by the use of kitchen exhaust fans. This finding is expected since the outdoor airflow required by area for restaurants in ASHRAE Standard 62.1-2010 is higher than the requirement of outdoor air flow by area required for retail and grocery stores. Also, the healthcare category shown in Figure 1 encompasses both treatment rooms for dentists as well as rooms for surgery and critical care. The requirements for surgery and critical care in ASHRAE Standard 170-2008 are significantly higher than other environments. The published studies demonstrated that the

ventilation rates within a single category varied widely. The wide range of ventilation rates is influenced by the building envelope leakage, operation of ventilation control systems, and the approach used to calculate the ventilation rate. Commercial buildings that have a relatively large envelope area and frequently used entrance doors tend to have high ventilation rates due to infiltration.

1.4 Ventilation: Implications of Sampling Approaches

The studies cited above used a variety of different tracer gas techniques to measure ventilation. Several studies evaluated sources of uncertainties when conducting tracer gas measurements. Axley (1991) found that the significant uncertainty of tracer gas measurements came from incomplete air mixing and unsteady airflow rates. Sandberg (1989) investigated the effect of airflow rates on the measurement errors for both constant concentration, and the decay methods and demonstrated a good agreement between the two tested methods. Sandberg (1989) also concluded that the measurement errors resulting from the changes in airflow rates during the experimental time period could generally be neglected when compared to the large errors due to poor air mixing in the tested environments. Poor mixing was estimated to contribute an error of 12%-18% .

Previous studies demonstrated a good correlation between the tracer gas constant injection and decay methods (Kumar et al. 1979; Sandberg 1989; Axley 1991; Heidt and Werner 1986) and between constant concentration and decay methods when the measurement was conducted for fixed airflow rates (Kumar et al. 1979). However, various analytical approaches might provide different ventilation rates and uncertainties. Sherman (1990) reported that each analytical approach provided strong results at specific environmental conditions. The study compared many different approaches including regression, integral, averaging, and transient techniques with various measurement time intervals and airflow rates. The averaging decay method presented high uncertainties

when the measurement time interval increased because the tracer gas concentrations tended to be low and variable at the end of the experiment. The regression and integral decay methods reduced the errors found in the averaging decay method, and both methods had the same level of uncertainty. The constant concentration requires preparations of the system controls for real time tracer gas injection to maintain the concentration at the same level. The uncertainties from this technique depended on the measurement time interval and the accuracy of system controls, but not on the changes in the airflow rates.

Due to high instrumentation expenses and/or labor intensity in measuring AERs with the typical tracer gas decay method using SF_6 and the fact that SF_6 is potent greenhouse gas, there is an interest in understanding CO_2-based ventilation rate measurements. One particular area of interest is the use of metabolic CO_2 as the tracer for continuous ventilation rate measurements. The CO_2 generation rate per occupant is correlated with the metabolic rate, respiration quotient and body surface area that depend on factors such as occupant activity type and physical condition. If the occupancy rates are known, the CO_2 generation rate can be calculated, and measurements of CO_2 levels can be used to calculate AER and percentage of AER supplied mechanically.

The successful application of CO_2 as the tracer gas has been demonstrated in residential and commercial buildings (Aglan 2003; Asadi et al. 2011; Grimsrud et al. 2011). In general, these studies are performed in single zone settings with easily determined occupancy rates. While the single zone assumption could be used for well-mixed retail floors, the difficulty in using CO_2 as a tracer in retail buildings comes from transient and difficult-to-account-for occupancy rates.

Grimsrud et al., 2011 measured ventilation rates in three large retail stores located in three climate regions of the U.S. (MN, FL, and MD) using the CO_2 decay method.

Hourly occupancy data for 48-hour periods, including employees, were used together with continuous CO_2 measurements taken in 6 to 12 locations in the stores to determine the hourly ventilation rates. Results agreed with SF_6 tracer-decay measurements and measurement results for the rooftop ventilation systems with an uncertainty of 10%.

Instead of using CO_2 decay method to measure the ventilation rate, Bennett et al. (2011) used the CO_2 equilibrium (steady-state) method and showed that the findings from this method are unreliable. In some of the test sites, the CO_2 concentrations never reached steady-state. In other buildings, the occupancy profiles were not consistent and in other buildings the difference between the indoor and outdoor CO_2 concentrations was small and caused large uncertainties. The authors recommended the use of the tracer decay method instead of the CO_2 steady-state method for future studies. Besides using CO_2 equilibrium to measure ventilation rates, Bennett et al. (2011) also used SF_6 decay method and PFT steady-state method. The R^2 value for the correlation between the SF_6 decay method and the PFT method was 0.76, between the PFT and CO_2 equilibrium was 0.84, and between the SF_6 and the CO_2 equilibrium was 0.74. According to the authors, the air exchange rate measured by steady state PFT was lower than the air exchange rate determined by SF_6 decay, probably because the steady state is influenced by the nighttime period, which tends to have lower ventilation rates than during the day. Another reason is related to the poor mixing in some buildings. The poor mixing lead to higher ventilation rates determined by SF_6 decay as the SF_6 sampling was done in areas in the building with significant airflow. Nonetheless, the SF_6 decay method was considered as the most accurate method to measure ventilation rates in retail stores.

2. IAQ

The following sections summarize the literature on contaminants measured in retail stores: particulate matter, VOCs, and other pollutants (i.e. ozone, SVOCs,

microbiological contaminants, radon, and carbon monoxide). For each contaminant, the average observed concentration, comparisons to standards, non-retail environments as well as the implications of collection methods is summarized below.

2.1 Particulate Matter

There is reason to believe that certain retail environments have particulate matter sources that is of indoor origin. Many groceries, general merchandise, and malls include food preparation areas or food courts. Cooking events can generate fine and ultrafine particles (e.g. Abt et al. 2000; Wallace et al. 2004). Other retail sites sell new clothing, which can be a source for textile particles (Maskey et al. 2011). Additionally, retail environments usually have high foot traffic areas, which contribute to indoor particles through movement of people, transport of outdoor dust, and resuspension of previously deposited particles (e.g. Abt et al. 2000; Lee et al. 2002). Frequent cleaning and other activities, such as the use of forklifts to move merchandise or propane-powered floor burnisher, may also contribute to elevated particle concentrations (e.g. Abt et al. 2000; Grimsrud et al., 1999).

Although short in duration, non-occupational exposures occurring during time spent in retail buildings can be significant contributors to 24-h personal exposures (Chang et al. 2000; Rea et al. 2001). Rea et al. (2001) suggested that up to 35% of particle exposure takes place in microenvironments (primarily retail buildings, restaurants, and coffee shops) where people spend only 4-13% of their time. Exposure to particulate matter has been associated with serious health effects (e.g. lung cancer, cardiovascular morbidity and mortality, ischemic heart disease mortality, non-fatal myocardial infarction, heart rate variability and systemic inflammation) in several epidemiological studies (e.g., Dockery et al. 1993; Pope et al. 2003; Pope and Dockery 2006; Weichenthal et al. 2008; Brook et al. 2010). The U.S. Environmental Protection Agency (EPA)

considers both PM_{10} and $PM_{2.5}$ as Criteria Pollutants, which can cause serious health effects. The WHO air quality guideline asserts there is no threshold below which particulate matter is not associated with adverse health effects (WHO 2005). It should be noted that PM standards/guidelines are established for outdoor concentrations. In the absence of indoor air quality standards, we assume that ambient standard/guidelines also apply to indoor concentrations. The comparisons between indoor concentrations and outdoor standards are used for reference only. The average particulate matter concentrations found in retail stores, possible indoor sources, comparisons with standards and other environments, and implications of sampling approaches are discussed below.

2.1.1 Particulate Matter: Measurements in Retail Stores

Indoor particulate matter concentrations in retail spaces have been reported in 11 articles. Of those studies, five studies reported PM_{10}, eight studies reported $PM_{2.5}$, one study reported PM_1, and one study reported particle with attached polycyclic aromatic hydrocarbons (PPAH) mass concentrations. Additionally, three literature studies reported ultrafine or a proxy for ultrafine (submicron, sub-3 μm) number concentrations. Also, one study reported chemical analysis of fine and coarse particles. Among these studies, only three (Lee and Hsu 2007; Bennett et al. 2011; and Chan et al. 2012) also reported the ventilation rates of the retail spaces. Figure 2 summarizes PM_{10}, $PM_{2.5}$, and submicron particle concentrations from these studies. In addition to the figure features described above, the dotted line represents the mean concentration and the short-dash line represents the mean concentration reported only by U.S. studies.

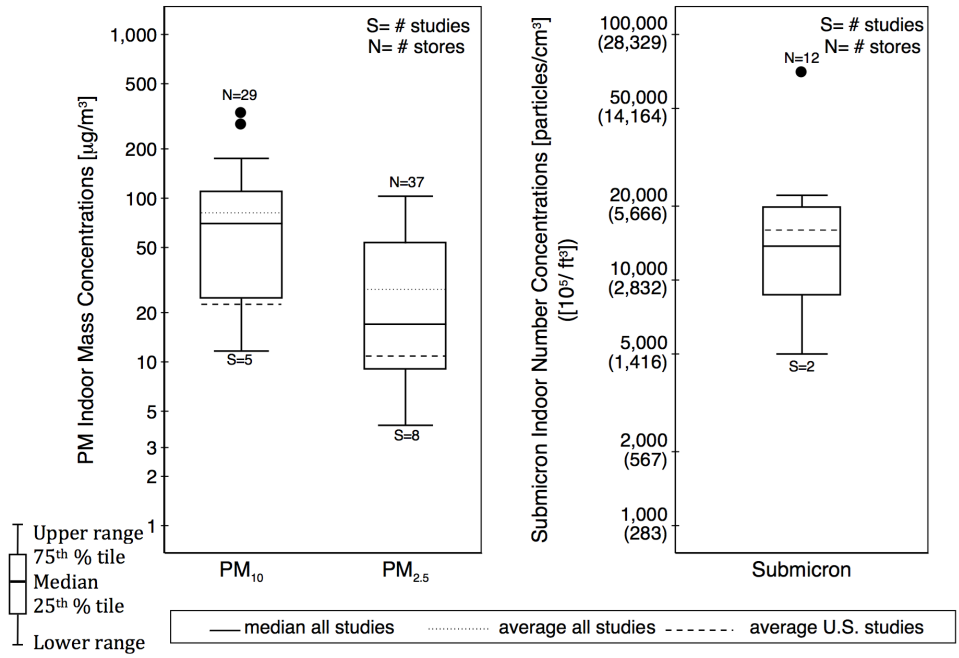

Figure 2. Summary of PM_{10}, $PM_{2.5}$ mass concentrations and submicron particle number concentrations (The horizontal line represents the median concentration, the dotted line represents the mean concentration, and the short-dash line represents the mean concentration reported only by U.S. studies. Data for PM_{10} are from : Li et al. 2000; Sakai et al. 2002; Liu et al. 2004; Bennett et al. 2011; Chan et al. 2012. Data for $PM_{2.5}$ are from : Grimsrud et al. 1999; Rea et al. 2001; Liu et al. 2004; Lee and Hsu 2007; Bennett et al. 2011; Brown et al. 2012; Chan et al. 2012; and Dong et al. 2013. Data for submicron particles are from : Bennett et al. 2011; Brown et al. 2012.

PM_{10} and $PM_{2.5}$. Median PM_{10}, shown in Figure 2, is dominated by studies conducted in Hong Kong, Taiwan, Japan and China where the outdoor air was an important contributor to indoor PM_{10} (Li et al. 2001; Lee and Hsu 2007; Dong et al. 2013). Bennett et al. (2011) and Chan et al. (2012) are the only studies that reported PM_{10} concentrations in U.S. retail stores. These two studies reported an average PM_{10} mass concentration for the 15 retail stores of 24.3 µg/m^3. Using data from Grimsrud et al., (1999); Rea et al., (2001); Bennett et al. (2011); Brown et al., (2012); and Chan et al. (2012) the average $PM_{2.5}$ mass concentration calculated for U.S. retail sites is 11.6 µg/m^3. Approximately half of all the stores tested had indoor-to-outdoor (I/O) ratio larger or equal to 1 and the rest had I/O ratio less than 1. This suggests that indoor PM_{10} and $PM_{2.5}$

can originate either indoors, outdoors, or both. The reported indoor sources include: cleaning (Grimsrud et al. 1999), cooking (Li et al. 2001; Bennett et al. 2011; Brown et al. 2012), and smoking (Li et al. 2001). Additionally, there is evidence that grocery stores exhibit higher particle concentrations than other types of stores: $PM_{2.5}$ mean concentration recorded in grocery stores measured in the U.S. was 31.7 $\mu g/m^3$ with a maximum concentration of 90 $\mu g/m^3$ (Bennett et al. 2011; Brown et al. 2012; Chan et al. 2012). The main contributor to elevated concentrations in these grocery stores is cooking events. Besides grocery stores, the data available for U.S. sites is insufficient to draw conclusions about specific indoor particle sources for different store types.

Submicron particles. Figure 2 contains a graph showing the submicron particle number concentrations reported by two studies conducted in the U.S. (Bennett et al. 2011; Brown et al. 2012). The average submicron concentration was 17,000/cm^3 ($481 \cdot 10^6/ft^3$). It is worth noting that the submicron concentration measured in these studies is a good proxy for ultrafine particles since the particle number concentration is typically dominated by small particles. Another non-U.S. study reported only the highest ultrafine number concentrations observed during photocopying of $10^8/cm^3$ (3,000 $10^9/ft^3$, Lee and Hsu 2007); thus the number reported in Lee and Hsu (2007) is not included in the figure. These investigations associated submicron particle number concentrations with in store cooking, cleaning, photocopying, and specific activities (presence of silk screening area) (Lee and Hsu 2007; Bennett et al. 2011; Brown et al. 2012).

PPAH. Sakai et al. (2002) measured particles of diameter under 1 μm with attached polycyclic aromatic hydrocarbons (PPAH), and reported an indoor average concentration of 3.5 ng/m^3 and an outdoor average concentration of 45.2 ng/m^3 in the parking lot of the store. Sakai et al. (2002) reported that the average concentration of

PPAH in the department store was low compared to levels measured by their study in restaurants, near garages, and in smoking rooms.

Chemical analysis of coarse and fine particles. Maskey et al. (2011) reviewed the chemical make-up of 7900 particles in 1–2.5 μm and 2.5–10 μm diameter bins in an underground mall in Korea. Maskey et al. (2011) reported that soil derived particles were the most common, followed in order by, primary soil-derived, carbonaceous, iron-containing, secondary soil derived, and secondary sea-salt particles. Given the proximity of the underground mall to subway terminals, their results for iron-containing particles correspond well with other studies in subway terminals (e.g., Aarnio et al. 2005). A significant baseline concentration of larger, thread-like textile particles with the chemical constituents of carbon, nitrogen, and oxygen were also observed in some of the samples.

2.1.2 Particulate Matter: Comparison to Outdoor Standards

Comparison of the measurements reported in the literature to the standards and guidelines is difficult, owing to the difference of averaging periods (1-year average or 24-hour average for regulatory/reference levels versus 8 hours or less reported in literature studies), location (ambient versus indoors), and different measurement methods. Given these limitations, comparison with the National Ambient Air Quality Standards (NAAQS) limit (24-hour PM_{10} limit =35 $\mu g/m^3$, and annual $PM_{2.5}$ limit = 12 $\mu g/m^3$) reveals that both PM_{10} and $PM_{2.5}$ mean mass concentrations measured in retail stores in the U.S. were below outdoor regulatory levels. However, the mean concentration measured in grocery stores in the U.S. for $PM_{2.5}$ mass concentration was 31.7 $\mu g/m^3$, which is 2.6 times higher than the NAAQS value.

2.1.3 Particulate Matter: Comparison to Non-Retail Environments

PM_{10} and $PM_{2.5}$. Figure 3 shows a summary of PM_{10}, and $PM_{2.5}$ mass concentrations found in literature for various environments, none of which included

(legal) smoking activities. Considering other environments, PM mass concentrations measured in retail stores in the U.S. are comparable to offices, religious buildings, healthcare facilities, and residences, and generally less than restaurants and classrooms. The wide range of values is largely the result of differences in averaging periods, ventilation rates, source strengths, outdoor concentrations, and sampling techniques used in the studies.

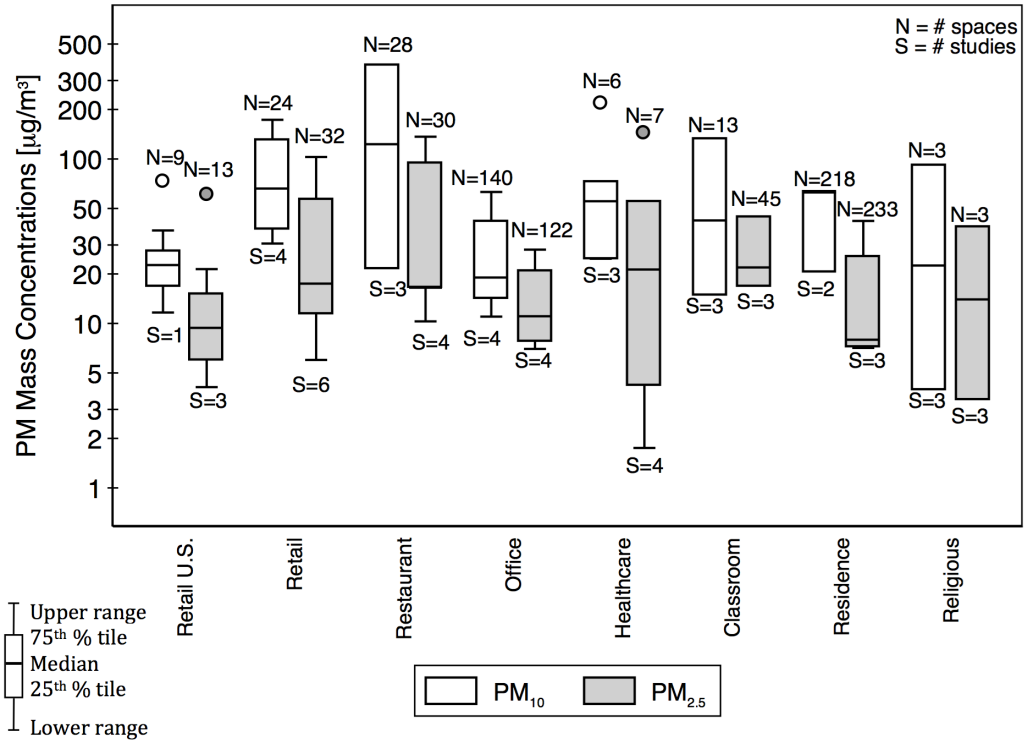

Figure 3. Summary of PM_{10}, $PM_{2.5}$ mass concentrations (N=number of spaces, S = number of studies). Data from retail stores are from: Grimsrud et al. 1999; Li et al. 2001; Rea et al. 2001; Sakai et al. 2002; Liu et al. 2004; Lee and Hsu 2007; Bennett et al. 2011; Brown et al. 2012; Chan et al. 2012; and Dong et al. 2013. Data for restaurants are from : Liu et al. 2004; Buonanno et al. 2010; Bennett et al. 2011; Brown et al. 2012. Data for offices are from : Turk et al. 1989; Burton et al. 2000; Liu et al. 2004; Bennett et al. 2011. Data for healthcare facilities are from : Tsai et al. 2000; Helmis et al. 2007; Bennett et al. 2011; Brown et al. 2012. Data for classrooms are from : Liu et al. 2004; Branis et al. 2005; Weichenthal et al. 2008. Data for residences are from : Ozkaynak et al. 1997; Long et al. 2000; Wheeler et al. 2011. Data for religious facilities are from : Bennett et al. 2011; Daher et al. 2011; Chuang et al. 2012.

Submicron particles. Submicron number concentrations reported in retail environments were lower than concentrations (average>50,000/cm^3 (>$10^9/ft^3$)) observed in restaurants or residences during cooking events (Buonanno et al. 2010; Wallace and Ott 2011; and Bennett et al. 2011), in photocopy centers (Lee and Hsu 2007; and Morawska et al. 2009), and near freeways or busy roads (Zhu et al. 2002; Kaur et al. 2005; and Zhu et al. 2008). Other environments where the submicron concentrations were lower than values reported for retail stores include residences measured overnight or with no cooking events, and classrooms (Zhu et al. 2005; Weichenthal et al. 2008; Mullen et al. 2011).

2.1.4 Particulate Matter: Implications of Sampling Approaches

Broadly, the sampling techniques used to measure particle mass concentrations can be divided into two categories: filter-based integrated gravimetric methods, and real-time methods.

For the ten studies found in the literature, four studies used a gravimetric technique, three studies used a laser photometric instrument combined with gravimetric technique for calibration, and three studies used a photometric instrument without any calibration. Since gravimetric methods directly measure the mass of particles accumulated during a sampling period, it is considered the most reliable method. However, there are many difficulties in conducting this type of measurement. Bennett et al. (2011) reported issues that arose during sampling such as the filter ripped during sampling and poor seal of the impactor. Bennett et al. (2011) also reported differences between results obtained when sampling $PM_{2.5}$ and PM_{10} directly through one impactor compared to summing PM masses through multiple stages of impactors.

For the studies that used laser photometer instruments with no calibration, their reported concentrations are less accurate than the studies that used gravimetric technique.

Laser photometers are generally calibrated at the factory against a gravimetric reference using a specific test dust. Aerosols with different physical/optical properties compared to the calibration dust can produce photometer responses differing by as much as a factor of 3 (Jiang et al., 2011). Ramachandran et al. (2000) demonstrated that the readings of TSI DustTrak (see Table 1) could exceed the true $PM_{2.5}$ values by a factor of 5 to 10. In addition to the calibration issue, the studies that employed a real-time particle counter and then converted the particle concentration to a mass concentration must assume a particle density, which increases the uncertainty.

Ultrafine particle measurements were monitored using condensation particle counters (CPCs): TSI ultrafine particle counters Model 3781 (0.006 – 3 μm; Bennett et al. 2011), Model 3025A (0.003 – 3 μm; Lee and Hsu 2007), and Model 8525 (0.02 - 1 μm; Brown et al. 2012). Given the differences in particle size range and instrument responses to environmental conditions and aerosol composition used in these studies, we would expect different results from these instruments if used in the same space.

2.2 Volatile Organic Compounds (VOCs)

Volatile organic compounds (VOCs) are a broad class of chemicals ubiquitous in the indoor environment. Concentrations of VOCs have the potential to be elevated in retail spaces as VOCS are emitted by a large range of products and activities. VOCs can be emitted by the retail merchandise (e.g. textiles, furniture containing pressed wood, etc.), produced during typical building operational procedures such as cooking or cleaning, and by building materials and displays present in the space. There is little evidence of harmful health effects caused by the majority of VOCs, at concentrations typically found non-industrial indoor environments (Molhave 2002). The average VOC concentrations found in retail stores, possible indoor sources, comparisons with standards and other environments, and implications of sampling approaches are discussed below.

2.2.1 VOCs: Measurements in Retail Stores

Indoor VOC concentrations in retail spaces have been reported in 12 journal articles. The investigators found a total of 60 different VOCs in over 160 retail spaces using a variety of collection techniques. For the purpose of this paper, the VOCs were divided into eight categories. The VOC categories were developed by combining compounds that exhibit similar chemical behaviors and/or potential exposure characteristics. Table 3 lists the individual compounds in each category.

Table 3. VOC Categories and Constituting Compounds

Name of Category	N*	Compounds
Aromatic compounds	11	Xylene and isomers, 1-ethyl-2-methylbenzene, 1-ethyl-3-methylbenzene, benzene, ethylbenzene, naphthalene, phenol, propylbenzene, styrene, toluene, benzothiazole, butylated hydroxytoluene
Halogenated compounds	8	Carbon tetrachloride, chloroform, freon 11, freon 113, methylene chloride, tetrachloroethylene, trichloroethylene, 1,4-dichlorobenzene
Terpenoids	6	Limonene, pinene, p-cymene
Alkanes	6	Cyclohexane, decane, dodecane, heptane, hexane, nonane, undecane
Alcohols	6	1-Butanol, 1-methanol-3-cyclohexene, 2-butoxyethanol, 2-ethyl-1-hexanol, ethanol, isopropyl alcohol, cyclohexanol
Carbonyls	6	2-Butanone, 4-methyl-2-pentanone, acetone, benzaldehyde, butyl ester acetic acid, decanal hexanal, nonanal, octanal, pentanal, acrolein
C1-C2 aldehydes	5	Acetaldehyde, formaldehyde
Remaining VOCs	9	1-(2-Methoxypropoxy)-2-propanol, decamethylcyclopentasiloxane, TMPD-MIB, TMPD-DIB, 1,3-butadiene, methyl tert-butyl ether, diethyl phthalate

* Number of studies investigating the category considered. Note that not all studies provided concentrations we could use for further analysis.

Eight of the papers in Table 1 (Kim et al. 2001; Tang et al. 2005; Hotchi et al. 2006; Lee and Hsu 2007; Bruno et al. 2008; Caselli et al. 2009; Wu et al. 2011; Chan et al. 2012) provided information that allowed for further analysis of the reported VOCs using this categorization. A summary of these results, converted to molar concentrations, is presented in Figure 4. Concentrations are weighted by the number of stores sampled when reported as an average of a store type.

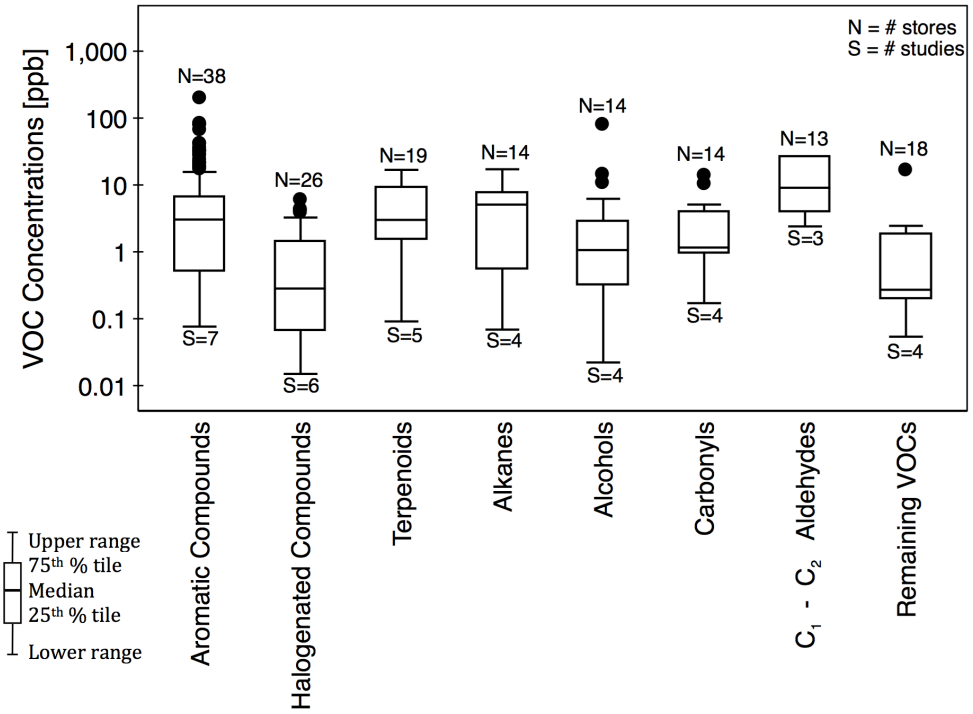

Figure 4. Summary of categorized VOC concentrations. Data are from for aromatic compounds are from Kim et al. 2001; Tang et al. 2005; Hotchi et al. 2006; Lee and Hsu 2007; Bruno et al. 2008; Caselli et al. 2009; and Wu et al. 2011. Data for halogenated compounds are from Kim et al. 2001; Tang et al. 2005; Hotchi et al. 2006; Bruno et al. 2008; Caselli et al. 2009; and Wu et al. 2011. Data for terpenoids are from Kim et al. 2001; Hotchi et al. 2006; Bruno et al. 2008; Caselli et al. 2009; and Wu et al. 2011. Data for alkanes are from Hotchi et al. 2006; Bruno et al. 2008; Caselli et al. 2009; and Wu et al. 2011. Data for alcohols are from Hotchi et al. 2006; Bruno et al. 2008; Caselli et al. 2009; and Wu et al. 2011. Data for carbonyls are from Hotchi et al. 2006; Bruno et al. 2008; Caselli et al. 2009; and Wu et al. 2011. Data for C_1-C_2 aldehydes are from Hotchi et al. 2006; Wu et al. 2011; and Chan et al. 2012. Data for remaining VOCs are from Kim et al. 2001; Hotchi et al. 2006; Wu et al. 2011; and Chan et al. 2012.

Aromatic compounds. The concentration of aromatic compounds averaged 15 ± 41 ppb (mean ± standard deviation), with a maximum concentration of 195 ppb measured for toluene in photocopy centers (Lee and Hsu 2007). The BTEXS group (benzene, toluene, ethylbenzene, xylenes, styrene) were found to have the highest concentrations in retail stores investigated in seven of the eight studies. Identified sources, particularly of toluene, include motor vehicle exhaust (Chao et al. 2001; Kim et al. 2001), newspaper ink in printing shops (Caselli et al. 2009) and photocopy centers (Lee and Hsu 2007), and

building materials in pharmacies (Bruno et al. 2008) and shopping malls (Tang et al. 2005).

Halogenated compounds. The concentrations for this category averaged 1 ± 1 ppb, with a maximum concentration of 6 ppb measured for trichloroethylene in a department store (Tang et al. 2005). Tang et al. (2005) identified halogenated compounds as contaminants of concern in a shopping mall in South China, mainly due to chlorinated cleaning agents, deodorizers, and domestic electrical appliances.

Terpenoids. The mean concentration of the terpenoids category was 5 ± 5 ppb, with a maximum of 17 ppb for limonene. Limonene, emitted by household cleaning products, was the primary pollutant in two supermarkets (Bruno et al., 2008).

C_1-C_2 Aldehydes. Three studies reported acetaldehyde and formaldehyde concentrations across a variety of store types, and all suggested formaldehyde was the primary contaminant of concern. C_1-C_2 aldehydes concentrations averaged 13 ± 10 ppb, with formaldehyde reaching a maximum of 26 ppb, and acetaldehyde concentrations as high as 15 ppb. Identified formaldehyde sources included pressed wood products (Hotchi et al. 2006; Loh et al. 2006), and carpet (Wu et al. 2011). Acetaldehyde is typically emitted during baking processes, and is naturally present in ripe fruits.

Carbonyls. The mean concentration of the carbonyls category was 3 ± 4 ppb. The highest concentration (14 ppb) was measured for acetone in the retail stores sampled by Wu et al. (2011), where it was likely emitted by medical and cosmetics products. Due to proximity to a nail salon, acetone was the primary contaminant of concern in the stores of a strip mall center (Eklund et al. 2008).

Alkanes, alcohols, and other VOCs. Compounds from the alkanes, alcohols and remaining VOCs categories were not considered pollutants of concern by any study. They were found in retail spaces at mean concentrations of 5 ± 4 ppb, 5 ± 15 ppb and 1 ±

3 ppb respectively. Specific sources were not identified. Four studies reported concentrations of non–halogenated C_6-C_{12} alkanes, which are typically emitted by fuels, paints, sealants and grease. Compounds from the alcohol category are typically generated by solvent-containing products and baking activities.

Overall individual VOC concentrations were low, with a mean under 10 ppb and a median under 2 ppb across all compounds and all studies. It remains unclear if the variability between store types explained the variation in VOC concentrations. Studies in which samples were collected in different store types can help to answer this question. Loh et al. (2006) reported higher toluene concentrations in multipurpose store, higher formaldehyde concentrations in houseware and furniture stores, and higher acetaldehyde concentrations in grocery stores. Chan et al. (2012) conducted measurement in two grocery stores and three furniture stores, and reported similar trends in formaldehyde and acetaldehyde concentrations.

In addition to individual VOC concentrations, three studies reported TVOC concentrations averaging 439 ± 313 $\mu g/m^3$. Tang et al. (2005) computed the TVOC by adding the concentrations as indicated by the peak area in total ion chromatography of all species together, while Bruno et al. (2008) and Caselli et al. (2009) summed all reported VOC concentrations. Bruno et al. (2008) observed the minimum TVOC concentration (188 $\mu g/m^3$) in a supermarket, and the maximum concentration (1,393 $\mu g/m^3$) in a pharmacy.

Figure 5 contains a comparison of categorized VOC and TVOC concentrations by store type. Considering the limitations in sample size, number of compounds per category and the presence of confounding factors across studies, there are insufficient data to draw definitive conclusions about differences between store type and VOC category concentrations.

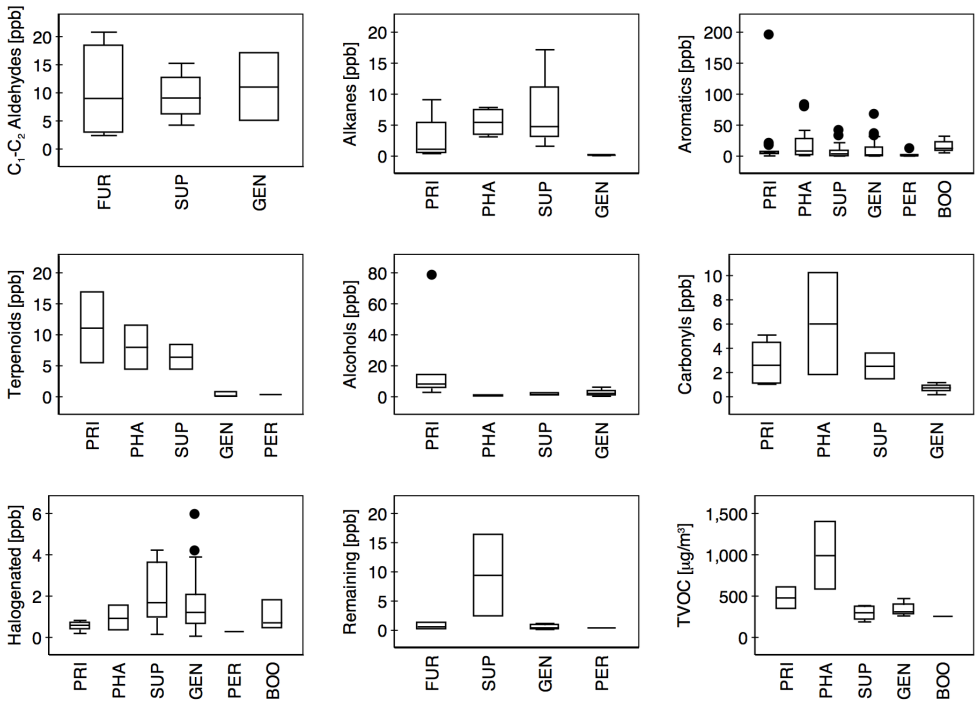

Figure 5. Comparison of categorized VOC and TVOC concentrations by store type. FUR stands for furniture store (3 stores, data from Chan et al. 2012), SUP for supermarket/grocery (6 stores, data from Tang et al. 2005; Bruno et al. 2008; and Chan et al. 2012), GEN for general merchandise/department store (7 stores, data from Kim et al. 2001; Tang et al. 2005; and Hotchi et al. 2006), PRI for printing shop/photocopying center (14 stores, data from Lee and Hsu 2007; Caselli et al. 2009), PHA for pharmacy (2 stores, data from Bruno et al. 2008), PER for perfume shop (3 stores, data from Kim et al. 2001), and BOO for bookstore (1 store, data from Tang et al. 2005).

2.2.2 VOCs: Comparison to Exposure Guidelines

Little is known regarding the health impacts from exposure to mixtures VOCs. For this reason, in this section we consider the health impacts of individual VOCs. Comparison of VOC concentrations to exposure guidelines is difficult as many organizations recommend different concentrations limits for specific compounds. We will consider here the most conservative reference limits published. Moreover, most chronic health guidelines assume either a continuous exposure, or an eight average period; while the duration of measurements conducted in the studies reviewed ranged from 30 minutes to 24 hours. Given the assumption that the samples collected in these studies were representative of typical concentrations, a comparison to heath guidelines can be used in

assessing exposure to VOCs. Six of the reviewed studies mentioned concerns about exposure of retail workers (Tang et al. 2005; Hotchi et al. 2006; Loh et al. 2006; Lee et al. 2007; Wu et al. 2011; Chan et al. 2012).

Across all studies, four compounds were reported as exceeding the most conservative exposure guidelines published. Formaldehyde concentrations (average concentrations across all the studies 20 ± 8 ppb) exceeded the California Office for Environmental Health and Hazard Assessment chronic recommended exposure limit (CA OEHHA chronic REL = 7 ppb) in most of the stores measured. In general merchandise stores (Hotchi et al. 2006; Chan et al. 2012), acetaldehyde concentrations were above the inhalation reference concentrations (RfC) set by the U.S. Environmental Protection Agency at 5 ppb. It should also be noted that the RfC is defined as "an estimate with uncertainty spanning perhaps an order of magnitude", that assumes a continuous exposure over a lifetime. Tang et al. (2005) reported benzene concentrations above the Hong-Kong IAQ objective of 5 ppb. Acrolein was only measured by Chan et al. (2012), and concentrations exceeded the U.S. EPA RfC of 0.009 ppb.

Out of the 160 compounds detected across all studies, several are considered potentially carcinogenic by the U.S. EPA. For instance carbon tetrachloride, chloroform, and tetrachloroethylene were detected at several sites, but at concentrations lower than their U.S. EPA RfCs. However, trichloroethylene concentrations averaged 1.1 ± 1.8 ppb, above its RfC of 0.4 ppb. This result is dominated by the concentrations found in a shopping mall in South China (Tang et al., 2005).

2.2.3 VOCs: Comparison to Non-Retail Environments

Two of the reviewed studies investigated multiple micro-environments. These studies allow for direct comparisons between VOC concentrations and eliminate potential confounding factors such as sampling techniques. Unfortunately, the findings are

inconsistent, especially for VOCs generated outdoors, for which the location of the space is the decisive factor. For instance Loh et al. (2006) reported higher average concentrations of aromatic compounds in retail stores than in transportation microenvironments, while Kim et al. (2001) reached the opposite conclusion. Residences and offices are two significant indoor environments, both in terms of occupancy and published investigations. The relationship of indoor, outdoor, and personal air (RIOPA) study was conducted in 100 residences from 1999 to 2001 in three American cities (Weisel et al. 2005). Logue et al. (2011) reviewed 77 published studies that reported measurements of chemical pollutants in residences from 2001-2008 (newer homes than in the RIOPA study) in the United States and in countries with similar lifestyles. The building assessment survey and evaluation (BASE) study involved 100 office buildings across the U.S. (Apte and Erdmann 2002). The mean categorized VOC concentrations in retail stores presented in Figure 4 were generally similar, or slightly lower, compared to the results reported in the BASE and the RIOPA studies. However, some of the contaminants found in retail stores were much lower than the mean concentration reported by Logue et al. (2011). For instance, formaldehyde averaged 20 ± 8 ppb in retail stores, 21 ± 11 ppb in residences-RIOPA, 56 ± 25 ppb in residences-Logue et al. (2011), and 13 ± 7 ppb in offices. The concentrations in retail spaces are also in general agreement with averaged concentrations found in residences in North America in the 1990s, as reported by Hodgson and Levin (2003). Notable exceptions are acetone and toluene, which are on average 10 times higher in retail environments when compared to residences.

2.2.4 VOC: Implications of Sampling Approaches

While all samples of C_1-C_2 aldehydes were obtained using 2,4-dinitrophenylhydrazine (DNPH) cartridges, the sampling approach for all other VOCs

differed amongst studies (cf. Table 1). VOCs were measured using stainless steel canisters (Summa canisters) in two studies, passive organic badges in three studies, and sorbent tubes in seven studies. The sorbents used by these seven studies also differed: four used tubes packed with Tenax TA, one with Tenax GR, and two with multiple sorbents.

Each sampling approach is associated with characteristic issues that can affect the identification and quantification of VOCs. For example, Summa canisters have been reported to show low recovery of alcohols (Ochiai et al. 2002), as well as aldehydes and terpenes (Batterman et al. 1998). Compounds with fewer than four carbon atoms and relatively high vapor pressures such as acetone, ethanol and isopropanol, are not readily sorbed by Tenax (Harper et al. 2000). In addition to the use of different collection methods, variations in the duration of sampling from 30 minutes to 24 hours might confound comparisons. Loh et al. (2006) also mentioned the potential differences in concentrations arising from mobile/personal versus stationary sampling, as the space might not be perfectly mixed.

Few field studies have investigated the differences in the concentrations obtained by various sampling techniques used simultaneously, and quantified these biases for the specific compounds collected. Depending on the compound sampled, organic vapor monitors have been reported to underestimate (Stock et al. 1999) or overestimate (Pratt et al. 2005) concentrations when compared to canisters. Dobos et al. (2000) reported that styrene concentrations measured by organic vapor monitors were 31% higher than measurements from sorbent tubes.

2.3 Other Pollutants

Ozone. Ozone is a contaminant that is typically generated though photochemical reactions between nitrogen oxides and VOCs in the troposphere. Many epidemiological

studies have documented the adverse health effects of ozone on the respiratory and cardiovascular systems, particularly the association between short-term exposure to ozone and daily mortality (Ito et al. 2005; Parodi et al. 2005; Bell et al. 2004; Levy et al. 2005; Gryparis et al. 2004; Zhang et al. 2006). Only two studies reported ozone concentrations in retail spaces. Lee and Hsu (2007) reported a maximum indoor ozone concentration of 70 ppb, measured when photocopying activities occurred, nearing the current U.S. EPA National Ambient Air Quality Standard (NAAQS) for ozone of 75 ppb. Indoor ozone concentrations before photocopying operations averaged about 4 ppb. Chan et al. (2012) presented the temporal profile of indoor and outdoor ozone concentrations for three stores over an afternoon. Average indoor concentrations ranged 10-25 ppb, while average outdoor concentrations range 29-37 ppb, both far below ozone health standards. The authors noted a positive relationship between ozone indoor-to-outdoor ratios and air exchange rates, moderated by potential losses to filters in mechanically ventilated spaces, further confirming the importance of outdoor air as a source of ozone.

Semivolatile organic compounds (SVOCs). SVOCs may be particularly important in the retail environment. Many stores have vinyl flooring, merchandise made of soft and hard plastics, foams and fabrics, all of which may contain plasticizers and flame-retardants. Also, cooking events, which occur in some retail environments, can release PAHs and pyrene (Weschler and Nazaroff 2008). Several SVOCs have been associated with adverse health effects and many SVOCs are endocrine-disrupting chemicals. Studies examining SVOC metabolites in blood and urine have provided direct evidence of widespread human exposure (Vonderheide et al. 2008; Heudorf et al. 2007; Weschler and Nazaroff 2008). Despite their health impact, SVOCs are understudied in indoor environments and there are relatively few standardized methods for their analysis. Only two studies have reported SVOC concentrations in retail environments (Hartmann

et al. 2004; and Wu et al. 2011). Hartmann et al. (2004) used polyurethane foam (PUF) plugs to collect air samples and reported the concentrations of eight organophosphates in two furniture and three electronic stores in Zurich, Switzerland. Tributyl phosphate (TBP), tris (2-chloroethyl) phosphate (TCEP) and triphenyl phosphate (TPP) were found in all of sites. Tris(2-chloro-isopropyl)phosphate (TCPP), a flame retardant, was found in furniture stores, but not in electronics stores, and Tris(1,3-dichloroisopropyl) phosphate (TDCP) was not found in any of the locations studied. Wu et al. (2011) used multi-bed sorbent tubes with a primary bed of Tenax-TA sorbent backed with a section of Carbosieve III to capture air samples and measure diethyl phthalate (DEP), as well as several VOC concentrations in 37 small- and medium-sized commercial buildings distributed across different sizes, ages, uses, and regions of California. DEP, a plasticizer commonly found in cosmetics and food packaging, was found in all seven of the retail sites with mean concentration equal to 0.49 $\mu g/m^3$. This concentration is lower than the mean concentration found in residences (0.67 $\mu g/m^3$) by Rudel et al. (2003, 2010), Otake et al. (2004), and Fromme et al. (2004).

Fungi and Bacteria. It is important to understand microbiological pollutants in the retail environment because of the high person density and potentially favorable conditions for which these pollutants can proliferate. There is some evidence that highly occupied environments that have to bring in large amounts of fresh air can increase the levels of indoor microbiological pollutants due to outdoor sources (Wu et al. 2005). The impacts of biological aerosol exposure on human health are considerable (e.g., Monto 2002; D'Amato et al. 2005). Only two studies investigated the presence of bacteria in shopping malls (Li et al. 2001; Tringe et al. 2008). Li et al. (2001) collected airborne bacteria and used culturing to estimate the amount of colony forming bacteria per cubic meter of air in nine shopping malls. Generally, they found an increased amount of

bacteria on weekends, with increased occupant density, as assessed by carbon dioxide measurements. Tringe et al. (2008) characterized bacteria present on HVAC filters in two malls in Singapore through genomic DNA sequencing and large scale cloning processes. Tringe et al. (2008) also compared bacteria found on HVAC filter dust with water from an adjacent river, floor dust samples (inside and outside), soil samples near the mall, and human nasal swabs. The microbiome captured on the HVAC filter dust was significantly less diverse compared to the soil and water samples. Each of the two malls shared some abundant microorganisms, but overall phylotypes were diverse, suggesting the malls have different microbiomes. The DNA functional analysis showed genes participating in cell mobility and secretion were over-represented, compared to previously reported genes in other soil and oceanic studies. Currently, there are no threshold values suggested for any microbiological measurement in the indoor environment.

Radon. Radon comes from the natural (radioactive) decay of geological materials (e.g. uranium in soil, and rocks). According to EPA, radon is responsible of many thousands of deaths each year in the U.S. Thus, indoor radon is an important pollutant of concern, especially in underground stores or in locations with known high radon concentrations. Three studies have reported radon concentrations in retail spaces (Yu et al. 1997; Marley 2000; Wang et al. 2004). Yu et al. (1997) measured 10 underground shopping centers located in Hong Kong and found an average concentration of 29.2 Bq/m^3, comparable to the average of dwellings of 30.1 Bq/m^3 and lower than the concentration measured in underground railway stations of 41.6 Bq/m^3, sampled by the same author in Hong Kong. The average reported for the shopping malls is well below EPA recommended action limit for radon of 148 Bq/m^3. Marley (2000) tested radon in a general merchandise store located in Northamptonshire (UK), an area classified as a radon affected area. The affected area of the store was the basement. The average radon

concentration reported before switching the AC On was 326 Bq/m^3, and the radon average after six hours from switching-on the AC was 22 Bq/m^3. The reported AER was 5 air exchange rates per hour. The study indicated that high radon concentrations can be mitigated using an appropriate ventilation strategy. Wang et al. (2004) measured 8 underground supermarket and office structures in China and reported a total average of 33.7 Bq/m^3. The reported average in supermarket/office was lower than the radon concentration found in underground parking lot, and in covered and uncovered tunnels measured by the same study.

Carbon Monoxide (CO). The U.S. EPA considers CO as one of the Criteria Pollutants that can cause serious health effects, and reports that CO poisoning is responsible for five hundred deaths annually in the U.S. Some indoor activities such as cooking using gas stoves and outdoor activities such as stock loading and nearby traffic can elevate indoor CO levels inside retail buildings. Li et al. (2001) measured CO in nine shopping malls, and found an average between 890 and 5200 μg/m^3. According to the authors, heavy traffic near the shopping malls, cooking inside food courts, and in some malls the operation of the fuel-powered ice-resurfacing resulted in elevated indoor concentration of CO. Inside these malls, all the measured CO concentrations were below the Hong Kong Indoor Air Quality Objective (HKIAQO) and the NAAQS regulatoy limit of 10,000 μg/m^3 (8-hour average).

Other inorganic pollutants (NOx, SOx). The literature search did not reveal any studies that measured NOx, or SOx in retail environments.

DISCUSSION

The following discussion focuses on the impact of ventilation on contaminants found in retail stores and the possible contaminants of concern for which control strategy should be prioritized.

Impact of Ventilation Rate on Contaminant Concentrations

Among all the reviewed studies, only four studies measured the impact of changing ventilation rates on pollutants concentrations or occupant's perception (VOCs: Hotchi et al. 2006; Grimsrud et al. 2011, Radon: Marley 2000, Occupant perception: Wargocki et al. 2004). In addition, Apte et al. (2011) modeled various ventilation scenarios (including scenarios that satisfy the ASHRAE 62.1 IAQ procedure) in California big box retail stores and estimated VOC concentrations and energy consequences.

Hotchi et al. (2006) reduced the ventilation rate by 30% (0.83/h to 0.57/h) in a general merchandise store in the U.S. and observed subsequent increases in VOCs concentrations ranging from 15% to 170%. In three general merchandise stores, Grimsrud et al. (2011) suggested that the ventilation rates could be lowered and still maintain an acceptable indoor air quality in accordance with ASHRAE standards in regards to TVOC and formaldehyde. Marley et al. (2000) suggested that an appropriate ventilation strategy is an effective solution to decrease radon concentrations in a general merchandise store located in a radon-affected area. Wargocki et al. (2004) found that the percent of panelists dissatisfied with indoor air quality increased from roughly 5% to 30% when the outdoor airflow rate decreased from 4 to 1 $L/s \cdot m^2_{floor}$ (0.8 to 0.2 cfm/ft^2_{floor}). To balance between energy consumption and air quality in retail stores, Apte et al. (2011) suggested alternative control strategies such as source removal, air cleaning, and local ventilation, combined with moderate ventilation rates.

The grocery stores investigated by Chan et al. (2012) exceeded both the minimum ventilation rate specified by ASHRAE Standard 62.1-2010 and California Code of Regulations Title 24-2010 and had acetaldehyde (12.7 ± 3.3 ppb) and acrolein (9.4 ± 8 ppb) concentrations above exposure guidelines (respectively 5 ppb and 0.009 ppb). For

acrolein, both indoor and outdoor concentration exceeded the reference exposure level, limiting the value of ventilation to reduce exposure. In addition, one of the retail stores investigated by Bennett et al. (2011) met both ASHRAE Standard-2010 and California Code of Regulations Title 24-2010 ventilation requirements and had formaldehyde (11.3 ± 4.7 ppb) above the most conservative exposure guidelines (7 ppb). In addition, two of the tested stores by Bennett et al. (2011) also exceeded ventilation requirements but the measured $PM_{2.5}$ (60.2 $\mu g/m^3$, 21.4 $\mu g/m^3$) were above the NAAQS outdoor regulatory limit (12 $\mu g/m^3$). For both stores, $PM_{2.5}$ outdoor concentration also exceeded the NAAQS regulatory limit, limiting the value of ventilation to reduce exposure. These findings suggest that comparing ventilation rates in the tested store to ventilation requirements does not necessarily correlate to the air quality in the store. Depending on the pollutant and its source, alternative control methods such as filtration might be more effective and possibly more economical than ventilation.

Contaminants of Concern

From the contaminants found in retail stores reported in the literature, there is evidence that some pollutants in some retail types are present at concentrations higher than their established limits. The established limits of some pollutants vary widely between different organizations. Thus, to evaluate contaminants of concern it is necessary to compare exposure to pollutants through a common metric of harm, such as disability-adjusted-life-years (DALYs; Murray and Lopez 1996). This will allow prioritizing mitigation strategies, including energy efficiency measures that affect indoor air quality.

Two different approaches were used to estimate DALYs from exposure to contaminants: (1) intake-incidence-DALY (IND) approach based on epidemiology-based concentration-response (C-R) functions following the work of Logue et al. (2012); and (2) intake-DALY (ID) approach based on animal toxicity literature following the work of

Huijbregts et al. (2005). The IND approach is a preferred health impact model than the ID approach since it does not require interspecies extrapolations; however, only information on C-R functions in humans are available for $PM_{2.5}$ and ozone. For the rest of pollutants, animal toxicity data was available in the literature to apply the ID approach.

$PM_{2.5}$ DALYs were calculated separately for total mortality, chronic bronchitis, and nonfatal stroke outcomes. For the rest of pollutants, the DALYs losses were calculated separately for the cancer and non-cancer outcomes taking into account the variations in susceptibilities of different age categories. For both methods, for each pollutant and each outcome, a Monte Carlo approach was used to quantify the central estimate and 95% confidence interval (CI) by sampling from uncertainty distributions of DALYs factors.

Average concentrations for thirty pollutants (e.g. $PM_{2.5}$, acrolein, formaldehyde, acetaldehyde, trichloroethylene, toluene, benzene, xylene, styrene, carbon tetrachloride, chloroform, naphthalene, ozone, etc.) were included in the analysis. $PM_{2.5}$ concentrations are taken from: Grimsrud et al. 1999; Rea et al. 2001; Bennett et al., 2011; Brown et al. 2012; Chan et al. 2012; and Dong et al. 2013. VOCs concentrations are taken from: Kim et al. 2001; Tang et al. 2005; Hotchi et al. 2006; Loh et al., 2006; Lee and Hsu 2007; Bruno et al. 2008; Caselli et al. 2009; Bennett et al., 2011; Chan et al. 2012. Ozone concentrations are taken from Chan et al., 2012. These pollutants were chosen for their high potential health impacts at the concentrations found indoors, as well as the availability of damage factors to perform the calculations. Health impacts were determined using the damage per intake factors reported in Pope et al. (2003) and Huijbregts et al. (2005). For each pollutant, the average DALYs were estimated for customers and employees, assuming customers above the age of fifteen spend on average

0.48 hours of their daily time shopping, and employees work eight hours per week-day in the retail site.

The estimated average DALYs lost due to $PM_{2.5}$ exposure in retail stores in the U.S. contributed the largest to the annual health impacts, with an estimated 76 DALYs per 100,000 persons annually (95% CI: 7.9, 728.5). The average annual total DALYs from VOCs/ozone (29-pollutants) contributed an estimated 70 DALYs per 100,000 persons. Acrolein accounted for the vast majority of these losses, contributing an annual average of 60 DALYs per 100,000 persons (95% CI: 0.23, 15,609). It should be noted that the acrolein concentration used in the DALY analysis is largely based on two grocery stores from one study (Chan et al., 2012), and thus the results are highly uncertain and more testing is warranted to verify these concentrations. Both $PM_{2.5}$ and acrolein were mostly elevated in grocery stores with cooking activities, thus the DALYs lost presented herein may not apply to exposures in other store types. The total health burden was mainly attributed to the employees in retail stores. A figure showing the estimated number of DALYs lost due to indoor inhalation intake of $PM_{2.5}$, ozone, and selected VOCs and the corresponding pollutant concentrations is shown in the Supporting Information.

Comparing the findings from this investigation to DALYs reported by Logue et al. (2012) in residences provides further evidence that $PM_{2.5}$ and acrolein are two common contaminants of concern. In addition to $PM_{2.5}$ and acrolein, Logue et al. (2012) identified formaldehyde as a pollutant that accounts for annual DALYs comparable to acrolein. In these residences, the average concentration reported for formaldehyde was 69 $\mu g/m^3$. In retail buildings, the average formaldehyde reported by the three studies (Hotchi et al. 2006; Wu et al. 2011; and Chan et al. 2012) was 24.6 $\mu g/m^3$. It should be noted that these three studies took place in California, and therefore may not be representative of

formaldehyde concentrations in retail stores in other states or countries. At this concentration formaldehyde accounts for 4 DALYs annually per 100,000 persons. DALYs from exposure to formaldehyde in retail buildings can be considered small even though the average concentration reported for formaldehyde in retail stores is 2.7 times the OEHHA chronic reference exposure for formaldehyde (9 $\mu g/m^3$).

Despite the large uncertainties associated with the toxicology data and the DALY approach, our analysis support the finding that mitigation strategies in retail stores, specifically the ones with cooking activities, should focus on decreasing exposure of employees to $PM_{2.5}$ and acrolein, with the caveat that more acrolein concentration data is needed to confirm this finding.

CONCLUSIONS

A review of ventilation measurements in retail stores found that half of the stores tested met/exceeded Standard 62.1-2010 or California Code of Regulations Title 24-2010; nonetheless, these ventilation rates were not sufficient to keep all pollutants below their most conservative limits, but this might not be true if we consider other authoritative limits. The work presented here reduced the number of pollutants found in retail environments (>70 pollutants) to a set of seven important pollutants that are found to exceed their established limits/reference exposures for a few of the stores tested ($PM_{2.5}$, acrolein, formaldehyde, acetaldehyde, trichloroethylene, benzene, and PM_{10}) and two priority pollutants ($PM_{2.5}$, acrolein) that contributed the largest to DALYs lost. Only a limited number of studies measured acrolein in retail stores, which suggests that further attention on this pollutant is warranted. Because of the lack of health guidelines for ultrafine particles and SVOCs, the concentrations of these pollutant categories could not be judged whether it is safe or unsafe, but it would also be prudent to also generate additional measurements of these compounds. Few pollutants, such as $PM_{2.5}$ and acrolein,

had elevated concentrations both indoors and outdoors above their established limits. Alternative ventilation strategies (i.e., nighttime flushes) for stores with high indoor concentrations and air cleaning approaches (i.e. particle filtration) might offer a more effective and less energy-intensive method to improve the air quality in retail stores. Generally, for the important pollutants identified above, there remains a need to perform additional testing in retail stores in an effort to determine sources of pollutant and potentially link them to activities in the store, as well as determine the importance of outdoor sources.

Appendix B

Paper 1: Particle Characterization in Retail Environments: Concentrations, Sources, and Removal Mechanisms

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(Submitted to *Indoor Air*)

ABSTRACT

Particles in retail environments can have consequences for the occupational exposures of retail workers and customers, as well as the energy costs associated with ventilation and filtration. Little is known about particle characteristics in retail environments. We measured indoor and outdoor mass concentrations of PM_{10} and $PM_{2.5}$, number concentrations of submicron particles (0.02–1 µm), size-resolved 0.3–10 µm particles, as well as ventilation rates in 14 retail stores during 24 site visits in Pennsylvania and Texas. Overall, the results were generally suggestive of relatively clean environments when compared to investigations of other building types, and ambient/occupational regulatory limits. PM_{10} and $PM_{2.5}$ concentrations (mean±sd) were 20±14 and 11±10 µg/m^3, respectively, with indoor-to-outdoor ratios of 1.0±0.7 and 0.88±1.0. Mean submicron particle concentrations were 7,220±7,500 particles/cm^3 with an indoor-to-outdoor ratio of 1.18±1.30. The median contribution to PM_{10} and $PM_{2.5}$ concentrations from indoor sources (vs. outdoors) was 83% and 53%, respectively. There were no significant correlations between measured ventilation rates and particle concentrations of any size. When examining options to lower PM2.5 concentrations below regulatory

limits, the required changes to ventilation and filtration efficiency were site specific, and depended on the indoor and outdoor concentration, emission rate, and infiltration level.

Key words: Particle Concentrations; Indoor-Outdoor Relationship; Indoor Source Strengths; Ventilation; Filtration; Retail.

PRACTICAL IMPLICATIONS

Little is known about particle concentrations, contribution of indoor sources, and emission rates in retail environments. Knowledge of these particle characteristics informs health scientists with input parameters to include in exposure modeling. The predicted concentration change in response to different ventilation rates and filtration efficiencies may be used for guidance to develop control strategies to lower particulate matter concentrations in retail environments.

INTRODUCTION

The U.S. EPA considers both PM_{10} and $PM_{2.5}$ as criteria pollutants due to the strong association between exposure to ambient PM and serious health effects, including lung cancer, cardiovascular morbidity and mortality, heart rate variability and systemic inflammation (e.g., Dockery et al., 1993; Pope et al., 2003; Weichenthal et al., 2008; Brook et al., 2010). Recent attention has focused on the health effects of ultrafine particles (Crüts et al., 2008; Knol et al., 2009), which are not yet typically regulated. Specifically in some of the tested retail buildings, research shows that $PM_{2.5}$ mass concentration was higher than the 24-hour average NAAQS (National Ambient Air Quality Standards) limit of 35 $\mu g/m^3$ (Li et al., 2001; Lee & Hsu, 2007; Bennett et al., 2011; and Dong et al., 2013). Exposure to high PM concentrations may have adverse

consequences for the occupational health of retail workers (approximately 10% of U.S. workforce; NRF, 2010) and intermittent exposure of 107 million shoppers daily (ATUS, 2011). Although short in duration, exposures of shoppers occurring during time spent in retail buildings can contribute significantly to an individual's 24-hour personal exposures (Chang et al., 2000; Rea et al., 2001).

Concentrations of particles indoors depend upon the fraction of outdoor particles that penetrate through the building shell (i.e., infiltration) or are brought inside through the HVAC system (i.e., mechanical ventilation), the generation of particles by indoor sources, and the loss mechanisms such as filtration and the deposition into indoor surfaces. The ability to link PM indoor concentrations to personal exposure in retail stores depends upon understanding the indoor fate of ambient and indoor-generated particles. There are useful data in the literature about particle concentrations found in retail environments (Li et al., 2001; Rea et al., 2001; Sakai et al., 2002; Liu et al., 2003; Lee & Hsu, 2007; Maskey et al., 2011; Bennett et al., 2011; Brown et al., 2012; Chan et al., 2012; Dong at al., 2013). Identified indoor sources of particles are cleaning (Grimsrud et al., 1999), cooking (Li et al., 2001; Bennett et al., 2011; Brown et al., 2012), smoking (Li et al., 2001), photocopying (Lee & Hsu, 2007), and other specific activities (e.g., presence of silk screening area; Bennett et al., 2011). However, there is no broad study to date that has evaluated the relationship between particle concentrations, ventilation rates, and filtration efficiencies in different types of retail stores. In addition, little is known about indoor emission rates in different types of retail stores, which is important to

predict how indoor concentrations change with changing filtration efficiencies and ventilation rates.

The objectives of this study are to characterize indoor particle concentrations and indoor-to-outdoor ratios, to examine the relationship of indoor concentrations with air exchange rates, to model emission rate distributions by retail type, to calculate the contribution of indoor and outdoor sources to indoor particle concentrations, and to predict the effectiveness of control mechanisms in different types of retail stores in Pennsylvania and Texas. The results from this study provide indoor concentrations and emission rates by retail type that can be used as input parameters for exposure analysis studies as well as guidance on the effect of ventilation rates and filtration efficiencies on particle concentrations.

METHODS

Sample description

A total of 14 retail buildings were monitored between April 2011 and May 2012. The building sample was divided equally between two locations with different climates (State College, Pennsylvania and Austin, Texas), with the attempt of choosing the same store brand found in both locations, but this was largely dictated by recruiting success. The sample included six stores visited multiple times over a year to assess any seasonal changes, and two intervention sites tested at different ventilation rates for two consecutive weeks. Each visit was treated as an independent observation for a total of 24 separate observations.

Table 1 describes the sample in more detail. The first column of Table 1 is a unique site identifier where the first letter is the store type (H = home improvement, M = general merchandise, F = furniture, E = electronics, O = office supplies, G = mid-sized grocery with floor area >2000 m^2, and S = small grocery with floor area <2000 m^2), the second letter is a brand identifier, and the third character indicates whether the site was located in Pennsylvania (P) or Texas (T). In the result section, for those stores that were visited more than once, a number is added to the site identifier to differentiate between visits to the same site. The maximum number of register transactions, often used with a multiplier as a proxy for store occupancy, was obtained from the store records for each given visit (Column 5 of Table 1). The maximum transaction number per hour was observed over four to five weekdays that correspond to the testing days. The number of transactions serves to compare occupancy between same store visited multiple times and between same store tested in two different locations (Pennsylvania, and Texas). The last column summarizes potential outdoor sources based on the store location.

Table 1. Characteristics of the retail sites visited.

Site	Months tested	Floor area [m^2]	Volume [m^3]	Max Transactions [#/h]	Potential Outdoor Sources
HaP	May Nov	11,700	93,400	207-380	Heavy traffic, near highway
HaT	Jun	12,400	91,800	265	Heavy traffic, near railway, in a strip-mall
MbP	Sep Jan May Jul	16,600	99,500	474-605	Rural area, near highway
MbT	Jul Oct Feb Apr	8,740	61,200	285-521	Heavy traffic, in a strip-mall
MiP[1]	Apr	11,300	66,800	400	Heavy traffic, in a strip-mall
MiT[1]	Apr	12,100	55,200	560	Heavy traffic, next to a highway
EgP	Feb Jun	2,700	20,300	141-157	Moderate traffic, next to bus stops
OhT	May	2,280	20,700	76	Heavy traffic
FfP	Mar	1,970	8,170	38	Moderate traffic, in a strip-mall
FfT	Oct Feb	3,250	19,800	48-55	Heavy traffic, near highway, in a strip-mall
GeP	Jul	3,100	25,300	370	Moderate traffic, next to a railway
GeT	Sep Jan	3,250	14,900	260-467	Heavy traffic, in a strip-mall
ScP	Aug	836	3,340	151	Moderate traffic, next to bars and restaurants
SdT	Aug	1,180	5,390	157[2]	Heavy traffic, entrance next to a bus stop

[1] Indicates intervention sites tested for two consecutive weeks, one week at the normal ventilation rate and another week at the maximum ventilation rate allowed by the store.
[2] Indicates people count rather than number of transactions.

Aerosol sampling and quality assurance

On site experiments typically took five workdays (started on Monday and ended on Friday) and data collection activities included fixed and mobile sampling. During the fixed sampling, the instruments were placed indoors in a location of convenience that did not interfere with customers and employees. For outdoor measurements, the instruments were located on the roof. For mobile sampling, portable particle instruments were carried in baskets or shopping carts and moved around in the store for four hours in order to collect more representative spatial data within the retail space. Mobile sampling generally took place at the middle of the testing week and usually started between 9:00 and 10:00 am and ended between 12:00 and 1:00 pm. An attempt to minimize resuspension from the shopping cart was done by parking the cart in different places of the store.

Real-time PM_{10} and $PM_{2.5}$ measurements were collected simultaneously indoors and outdoors using photometric instruments (TSI Sidepak, TSI Dusttrak, and Met One Aerocet). At five sites (HaP1, HaP2, MbP2, MbT1, and SdT), outdoor $PM_{2.5}$ concentrations measurements were obtained from the closest monitoring station from either the Pennsylvania Department of Environmental Protection (PA DEP) or from the Texas Commission on Environmental Quality (TCEQ) because of instrument failure or because adverse weather conditions prevented outdoor measurements. Caution should be taken when assessing the indoor-to-outdoor ratios from the sites where offsite outdoor particle measurements were used. For example, Site SdT was located in a very dense area and we identified a bus stop next the entrance of the store. For this site, we estimate that the offsite outdoor particle measurements might underestimate outdoor concentrations, and thus overestimate indoor-to-outdoor ratios.

Integrated PM_{10} and $PM_{2.5}$ measurements were collocated with real-time particle instruments to obtain calibration factors that corrected for the difference between the physical/optical properties of aerosols tested and the dust used in the calibration. For each test, seven filters (37-mm diameter Emfab filters; Pall PK100) were prepared. Four filters were used to monitor PM_{10} and $PM_{2.5}$, both inside and outside the building. The three additional filters served as a field blank and a laboratory blank and, in some sites, the seventh filter was used to collect a replicate sample. Prior to sampling, filters were conditioned in a controlled environment (relative humidity 52-55% and temperature 21-22 °C) for at least 48 hours. After conditioning, the filters are weighed using a 5-digit microbalance (Mettler Toledo AB135). The filters are then placed in personal

environmental monitors (SKC $PM_{2.5}$ PEMs) and transported to the test sites in a field test box that maintained the relative humidity at 52-55%. The PEMs were attached to air pumps with calibrated flow rate of 10 L/min. The indoor and outdoor samples were collected simultaneously for a minimum of 48 hours, producing a sample volume of at least 28.8 m^3. At the end of sampling, the PEMs including the filters were transported back to the laboratory in the field test box and conditioned for at least 48 hours prior to weighing. The filters were weighed as soon as the conditioning phase was over. When weighing the filters, the analyst used a smooth, non-serrated forceps that was cleaned with alcohol and lint-free wipes and allowed to air-dry before handling filters. To minimize the static charge, filters were placed inside the balance on a disposable plastic anti-static tray. The weight for each filter was calculated as the mean of three weighing repetitions. The sample mass, the average flow rate, and the sampling duration were used to determine the average gravimetric mass concentration. The main uncertainty in the gravimetric measurements stems from the accuracy of the microbalance (5-digit) used to weigh the filters, and the difference between the type of balance used in Pennsylvania and Texas. At one site (MiT), a real-time gravimetric instrument (Thermo Scientific TEOM) was used to continuously measure real-time mass concentrations of PM_{10} and $PM_{2.5}$. To validate the gravimetric measurements, the Thermo Scientific TEOM was collocated with $PM_{2.5}$ PEMs in Site MiT. Results showed that the calculated calibration factors from both methods deviated less than 10%.

A submicron particle counter (TSI P-Trak) was used to measure indoor and outdoor particle number concentrations of submicron particles (>20 nm). The TSI P-Trak emits

isopropanol and therefore submicron particle measurements delayed until after the mobile sampling period to avoid interference with VOC measurements conducted simultaneously with particle measurements. Generally, the measurement period started after the mobile sampling period and lasted for one hour with measurements alternated between indoors and outdoors; with the exception of Sites HaP1, HaP2, and MbP1 where only indoor measurements were performed. Using particle counter instruments (TSI Aerotrak), average number concentrations were measured simultaneously indoors and outdoors for six different size fractions: 0.3 to 0.5 μm, 0.5 to 1 μm, 1 to 2 μm, 2 to 3 μm, 3 to 10 μm, and >10 μm. At selected sites, TSI Aerotrak instruments were collocated at the permanent sample location. The calibration factors obtained from collocation of multiple instruments are detailed in the Supporting Information.

For real-time measurements, the arithmetic mean concentrations and standard deviations were calculated and reported for the particle sizes considered in this work. Shapiro-Wilk test was used to test normality or lognormality of variables (using the log transformed variable in the latter case). Spearman Rank Correlation non-parametric test was used to assess the relationship between the non-normal variables and the air exchange rate. P-values were deemed significant if $p<0.05$. All statistical analyses were performed using Stata version 12.1 (StataCorp, College Station, USA

HVAC characterization and quality assurance

All stores were conditioned and mechanically ventilated with rooftop air conditioning units (RTUs) that ranged in size from 3 to 55 tons. Total building ventilation rates (ACH) were measured using a tracer decay method. SF_6 was released into different sections of

the store to obtain a uniform concentration. After waiting one hour (depending on the site), three field technicians simultaneously collected air samples every four to six minutes for a total of four hours with 500 mL plastic syringes at 15-18 sampling locations. In addition to the sample bags, QA/QC bags, including trip blanks and trip standard bags, were prepared and mixed in with the sample bags prior to analysis. All samples were analyzed using a Lagus Autotrac SF_6 analyzer generally within 48 hours of collection. Detailed methodology of SF_6 sampling can be found in Siegel et al., 2013.

System airflow rates were measured with an Energy Conservatory TrueFlow metering plate and DG-700 digital manometer. For the measurement, the Trueflow plates were installed in place of the system filter (removed for measurement). To account for differences in pressure drop across the Trueflow plates and the system filter, the static pressure was measured for both cases and used to adjust the supply flow, following the calculation procedure in the instrument manual.

Mechanical ventilation fractions were calculated on a mass balance of calibrated CO_2 concentrations applied in the mixed stream of the RTU, following the work of Persily (1997).

Emission rate modeling approach

The indoor emission rates normalized by the area of the store were calculated for PM_{10} and $PM_{2.5}$ by applying the time-averaged (during the four hours of mobile sampling period) mass-balance provided by Riley et al. (2002) and adding an indoor source emission term, as shown in Equation 1.

$$\frac{E}{A} = \frac{V}{A}(C_{in}\left(\frac{Q_r\eta}{V} + \beta + \frac{Q_e}{V}\right) - C_{out}\left(\frac{Q_m(1-\eta)}{V} + \frac{pQ_i}{V}\right)) \quad (1)$$

In Equation 1, E is the indoor emission rate [μg/h], V is the volume of the store [m^3], A is the area of the store [m^2], C_{out} and C_{in} are the outdoor and indoor concentrations [$\mu g/m^3$], Q_m is the mechanical outdoor airflow rate [m^3/h], Q_r is the recirculation airflow rate [m^3/h], Q_e is the exhaust airflow rate [m^3/h], η is the filter efficiency, β is the particle deposition rate onto indoor surfaces [1/h], p is the penetration factor of particles through leaks in building envelopes and major openings, and Q_i is the infiltration airflow rate [m^3/h]. The major assumption in applying this model is assuming a well-mixed space. To assess the degree of mixing, measured air exchange results suggested that the mean ages of air in most of the tested stores were fairly-to-well mixed (mean age of air ranged from 0.8 to 2.5 hours); exceptions are the home improvement stores, the office supply store, and the Texas furniture store. Caution should be taken when looking at the emission rates in these stores. In addition, we assumed no resuspension or coagulation of particles, and no phase change processes (gas-to-particle transformations and vice versa).

Concentrations and ventilation parameters were measured as detailed in the Methods section; filtration efficiency, deposition rate, and penetration factor were estimated. For filter efficiency, three different ASHRAE Standard 52.2 minimum efficiency reporting values (MERV) were considered: MERV 6, MERV 7, and MERV 8. Filtration efficiency [%] for integrated PM_{10} and $PM_{2.5}$ concentrations was calculated by combining manufacturer reported efficiencies with indoor and outdoor particle distributions for each geometric mean of the different particle sizes measured by TSI Aerotrak according to the

fraction of recirculated and mechanical outdoor airflow rates. For particle deposition loss rate coefficient, only deposition into indoor surfaces is considered because deposition on heat exchangers were determined in previous literature studies to be minimal (e.g. Siegel et al., 2003) and deposition on ducts is considered negligible since HVAC distribution systems in retail stores have minimal ducting. In this work, three different deposition rates reported by Lai and Nazaroff (2000) corresponding to three different friction velocities were considered. The considered values encompass most of the values reported in the literature for residential environments. To account for the very high ceilings encountered in most retail spaces, two different S/V (S/V=1/m, S/V=2/m) were considered. Following the same procedure for efficiency, deposition values for different particle sizes were integrated to determine deposition for PM_{10} and $PM_{2.5}$. Particle penetration factor was assumed to be equal to unity because particle infiltration in retail stores is dominated by large openings such as entrance, exit, and loading doors.

Table 2 summarizes the minimum, maximum, median, 10^{th}, and 90^{th} percentile of the total air exchange rate, the mechanical ventilation rate, the total supply rate, the filter efficiency of outside and recirculated air, and the deposition rate across the 24 sites. From this data, the recirculation airflow rate is calculated by subtracting the mechanical ventilation airflow rate from the total supply airflow rate. The infiltration airflow rate is calculated by subtracting the mechanical ventilation airflow rate from the total air exchange rate. The ventilation exhaust is calculated by summing the ventilation air and infiltration entering the building.

Table 2. Summary of measured ventilation and supply rates, and estimated efficiency and deposition parameters across the 24 sites.

Parameter	Min	10th	50th	90th	Max
Total Air Exchange Rate [1/h]	0.2	0.25	0.51	1.14	1.5
Mechanical Ventilation Rate [1/h]	0.13	0.16	0.19	0.37	0.45
Total Supply Rate [1/h]	1.16	1.17	1.45	2.8	3.4
Filter Efficiency $PM_{2.5}$ [%]	17	17	24	32	37
Deposition Rate [1/h]	0.14	0.17	0.26	0.36	0.38

Uncertainty in emission rate calculations stems from uncertainty of measurements of particle concentrations and ventilation parameters, and uncertainty of estimating unknown parameters (i.e., filtration, penetration, and deposition). To account for these uncertainties, emission rate distributions were modeled by sampling from uncertainty distributions of the influencing parameters. In the parametric analysis, 13,122 samples by site were drawn from values that correspond to the mean, mean±sd for each measured indoor, outdoor concentration, particle size distribution, infiltration, ventilation, and recirculation rates, and three different estimated MERV filter efficiencies, three different deposition rates, and two S/V ratios. Additional simulation runs were completed to test the importance of varying efficiency and deposition values on $PM_{2.5}$ and PM_{10} emission rates as described in the Supporting Information.

RESULTS

Indoor concentrations, indoor-to-outdoor ratios, and indoor sources

Bar charts of indoor and indoor-to-outdoor (IO) ratio for $PM_{2.5}$ and submicron concentrations by site are shown in Figures 1 and 2. PM_{10} results are detailed in the

Supporting Information and show similar trends to the $PM_{2.5}$ results. Sites are grouped by store types between dashed lines. Within the same type of stores, dotted lines are used to separate different brands of stores. The maximum concentration measured at each site is displayed in parentheses.

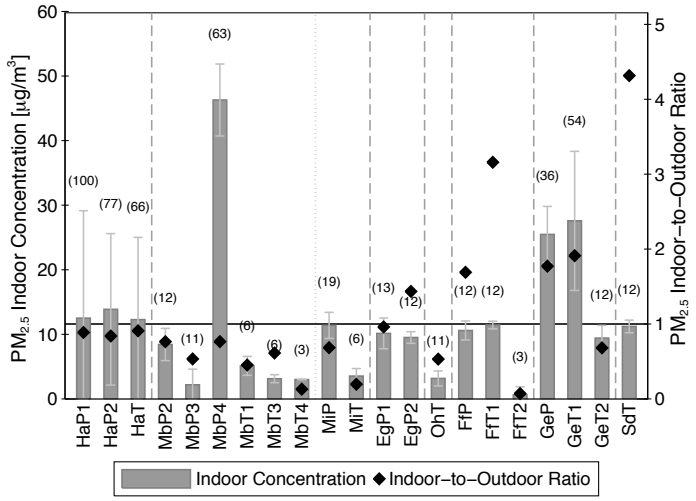

Figure 1. $PM_{2.5}$ indoor mass concentration and indoor-to-outdoor ratio by site. Maximum concentration is presented in parentheses. Horizontal black line represents $PM_{2.5}$ IO ratio equal to 1.0 (right axis).

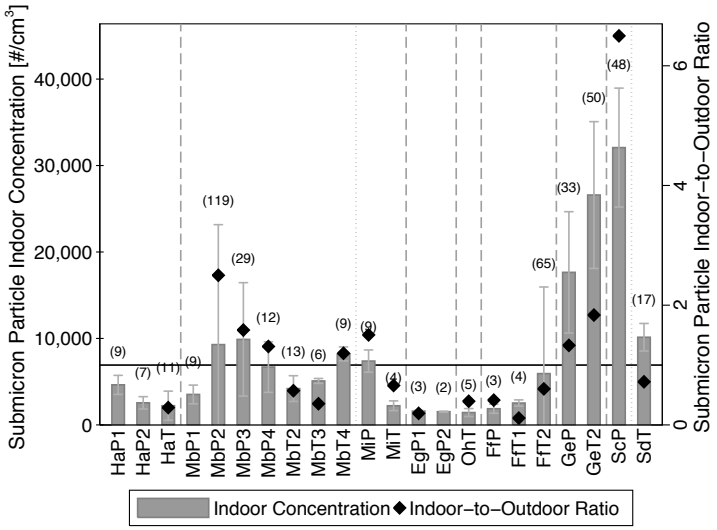

Figure 2. Submicron indoor number concentration (PN_1) and indoor-to-outdoor ratio by site. Maximum concentration divided by 1000 is presented in parentheses. Horizontal black line represents PN_1 IO ratio equal to 1.0 (right axis).

PM_{10} and $PM_{2.5}$ concentrations (mean±sd) were 20±14 and 11±10 μg/m^3, respectively. PM_{10} concentrations ranged from 4.5 μg/m^3 at Site FfT2 to 59.5 μg/m^3 at Site GeT1. $PM_{2.5}$ mass concentrations ranged from 0.6 μg/m^3 at Site FfT2 to 46.3 μg/m^3 at Site MbP4. Across all stores, the average PM_{10} and $PM_{2.5}$ concentrations reported in this study were lower than the average PM_{10} (24.3 μg/m^3; Bennett et al., 2011; Chan et al., 2012) and slightly lower than the average $PM_{2.5}$ (11.6 μg/m^3; Grimsrud et al., 1999; Rea et al., 2001; Bennett et al., 2011; Brown et al., 2012; Chan et al., 2012) reported for retail stores measured in the U.S. The average $PM_{2.5}$ reported for grocery stores measured in the U.S. was 31.7 μg/m^3, which is higher than the results of this investigation (17.5 μg/m^3). Dissimilarities may be due to differences in outdoor concentrations, occupant density, ventilation rate, the amount and frequency of cooking activities and associated use of exhaust fans, as well as differences in collection/calibration techniques. For other environments, generally PM concentrations reported for retail stores (this study and previous studies) were lower than concentrations reported for restaurants (Buonanno et al., 2010; Bennett et al., 2011; Brown et al., 2012), residences (Clayton et al., 1993; Wallace et al., 1994), and classrooms (Branis et al., 2005) and higher than concentration reported in offices (Burton et al., 2000).

Mean submicron particle number (PN_1) concentrations were 7,220±7,500 particles/cm^3. PN_1 ranged from 1,429 particles/cm^3 at Site OhT to 32,074 particles/cm^3 at Site ScP. Mean submicron particle number concentrations in the current study was lower than the average value reported in the literature for retail stores tested in the U.S. (17,000 particles/cm^3, Bennett et al., 2011; Brown et al., 2012). This may be explained by the

short duration of the measurements performed in this study (one hour) and the difference in the size range measured. For example, Bennett et al. (2011) used a TSI CPC ultrafine particle counter that detects a wider range of particle sizes than the TSI P-Trak used in this study. Generally, reported concentrations in retail stores were much lower than other environments. For example, average reported particle concentration in retail stores (this study and previous studies) was at least four times lower than concentrations observed in restaurants or residences during cooking events (Buonanno et al., 2010; Wallace & Ott, 2011; Bennett et al., 2011), in photocopy centers (Lee & Hsu, 2007; Morawska et al., 2009), and near freeways or busy roads (Zhu et al., 2002; Kaur et al., 2005; Zhu et al., 2008). Other environments where the submicron concentrations were comparable to concentrations found in retail stores include residences measured overnight or with no cooking events and classrooms (Zhu et al., 2005; Weichenthal et al., 2008; Mullen et al., 2011).

PM_{10} and $PM_{2.5}$ indoor-to-outdoor (IO) ratios were 1.0±0.7 and 0.88±1.0, respectively. The mean submicron particle IO ratio was 1.18±1.30. Sites with IO ratio larger than 1.0 (e.g., grocery stores for any particle size) suggest the presence of a strong indoor source. However, explicit conclusions are difficult to make when IO ratios are less or equal to 1.0 as this may indicate one of or combinations of the following factors: low indoor emission rates, effective filtration, and/or high outdoor concentrations.

Across all sites, there was no obvious association of either PM_{10} or $PM_{2.5}$ indoor concentrations with store type. Generally, submicron particle number concentrations for all sites were comparable except for grocery stores with cooking activities (i.e. frying,

baking, grill/burner use), which had relatively higher submicron particle number concentrations. These cooking activities also increased $PM_{2.5}$ concentrations, as observed at the tested grocery stores. Cleaning activities using cleaning agents also appeared to increase submicron particle concentrations at the sites where cleaning was observed. Specifically, submicron particle concentrations were measured before and after a cleaning event at Site MiT while maintaining a constant ventilation rate. The resulting indoor concentration after cleaning was 3.5 times higher than before cleaning occurred and resulted in IO ratio larger than 1. Cleaning and cooking sources were also found to increase particle concentrations in residential and hospitality environments (e.g., Buonanno et al., 2010; Wallace and Ott, 2011; Bennett et al., 2011). Outdoor sources, associated with nearby agriculture burning, were significant at Site MbP4 (Outdoor $PM_{2.5}$ = 60 $\mu g/m^3$), and likely contributed to the high measured indoor $PM_{2.5}$ concentration. Among the four visits to Site MbP, high outdoor sources were only observed at Site MbP4; for other visits (Sites MbP1, MbP2, MbP3) both indoor and outdoor concentrations were much lower than those observed at Site MbP4. Besides the difference in outdoor concentrations, the same store visited twice (different season) might have a different indoor emission rate that causes lower or higher indoor concentration. For example, this was observed at the grocery store brand e measured in Texas (Sites GeT1 and GeT2). Nonetheless, some stores had comparable indoor concentrations among different visits: we measured approximately the same indoor and outdoor concentrations at the different home improvement stores visits, these stores likely had comparable

indoor emission rates. Indoor emission rates results are presented later in the indoor emission rates section.

Home improvement stores had indoor particle sources independent of consumer activities; these sources were potentially caused by diminished ventilation rates, or the use of forklifts at night. There was very limited evidence for secondary particle formation from precursor emissions such as volatile organic compounds, but such emissions may be important in some parts of some stores. For home improvement stores, the average mass concentration for both PM_{10} and $PM_{2.5}$ measured during business hours (7:00 AM till 10:00 PM) were 30% less than the 24-hour average concentration. During this time, outdoor concentrations were lower than indoors. For all other sites with high indoor concentrations, these concentrations were generally associated with high activity levels for customers and/or employees or high outdoor concentrations. An example of the temporal profile of indoor and outdoor number concentrations for different particle sizes between 0.3 and >10 μm for Site SdT is shown in Figure 3 and 4, respectively. PN concentrations in all sizes dropped when the store was closed. Additionally, few peaks were observed during the day, which are likely explained by the intermittent nature of particle sources. During the day, outdoor concentrations were generally lower than indoor concentrations.

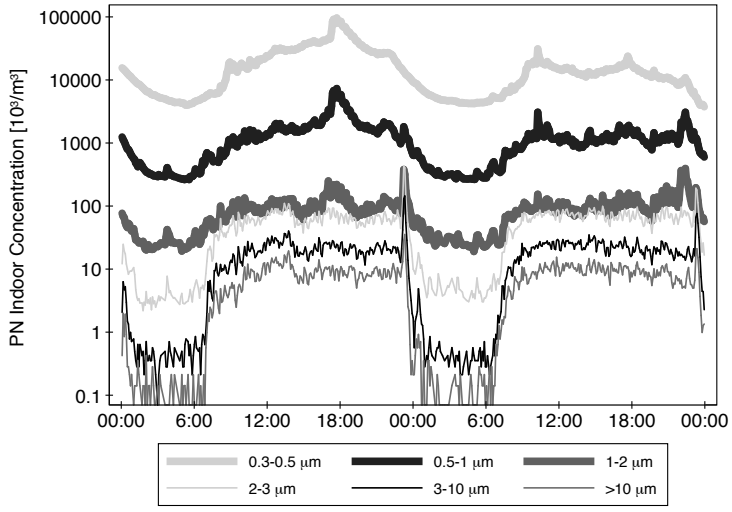

Figure 3. PN indoor concentration by time at Site SdT.

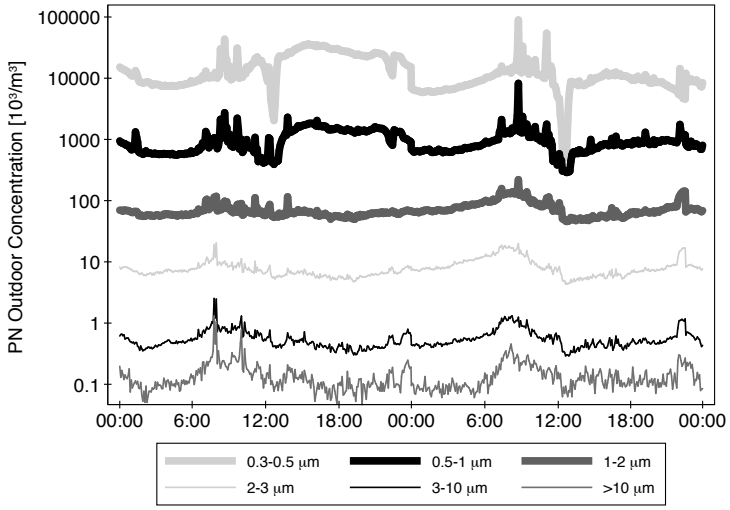

Figure 4. PN outdoor concentration by time at Site SdT.

Figure 3 also shows that the particle number concentration of the smallest size bin (i.e., 0.3 to 0.5 μm) dominated the other particle sizes. This was true for all the measured sites and in conformity with findings of Chan et al. (2012). Figure 5 shows the percentage contribution of the various size fractions to the total indoor number concentrations. As shown in Figure 5, the particle size distribution for the measured size fractions is essentially constant for the test sites. The smallest size bin (i.e., 0.3 to 0.5 μm) accounted

for approximately 92% of particles measured. The second smallest size bin (i.e., 0.5 to 1 μm) contributed 7% of the total number concentrations, and the remaining 1% came from the remaining larger size bins. The graph showing the contribution of different size fractions to the total outdoor number concentrations by site is detailed in the Supporting Information and shows similar trends as the indoor results.

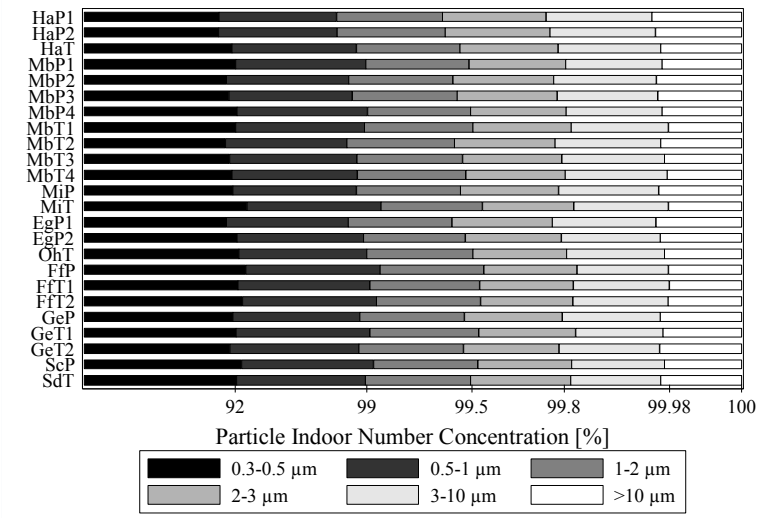

Figure 5. Contribution of indoor number concentrations by site displayed as percentage of the total number concentrations on a log scale.

Comparison to regulatory limits

OSHA (Occupational Safety and Health Administration) has established a permissible exposure limit (PEL) for respirable (PM_{10}) and total particles that workers can be exposed to for an 8-hour work shift (PEL for respirable particles is equal to 5 mg/m^3 while PEL for total particles is equal to 15 mg/m^3). OSHA has currently no occupational standard specifically or $PM_{2.5}$. We compared the exposure of employees to PM_{10} with the

occupational OSHA PEL, and we found that all stores met the occupational PM_{10} OSHA PEL.

According to the NAAQS (National Ambient Air Quality Standards), for an area to attain the 24-hour $PM_{2.5}$ primary and secondary standard, the 98th percentile 24-hour value for each year, averaged over 3 years, must not exceed 35 $\mu g/m^3$. For an area to attain the 24-hour PM_{10} primary and secondary standard, 150 $\mu g/m^3$ must not be exceeded more than once per year on average over 3 years. For the purpose of providing a reference to the data sampled in this study, we compared the short-term exposures inside retail buildings to the daily NAAQS. One of the 21 tests had indoor $PM_{2.5}$ concentrations higher than the NAAQS limit, and all stores had indoor PM_{10} concentrations lower than the NAAQS limit. It should be noted that the NAAQS was specifically designed to protect human health, including sensitive receptors, from exposure to ambient particulate matter. Thus, comparing the indoor concentration measured in this study to the NAAQS limit might be misleading especially that indoor particulate matter emitted from indoor sources have different compositions that that emitted from outdoor sources. Also, in a later section of this paper we demonstrate the importance of indoor sources contribution versus outdoor sources to both PM_{10} and $PM_{2.5}$. Further comparison limitations arise from the difference of averaging period between the measured data and the NAAQS limit: the PM sampled concentrations in this study represent relatively short monitoring periods compared to a year, causing PM concentrations to be unlikely representative of the 98th percentile (case of $PM_{2.5}$) or of the maximum 24 hour value in a three year period (case of PM_{10}).

However, in the absence of any regulatory limit for indoor exposure to $PM_{2.5}$, it is reasonable to compare the findings of this study to the NAAQS limit.

Indoor emission rates

Modeled distributions of indoor emissions of PM_{10} and $PM_{2.5}$ are calculated using Equation 1 for a total of N=13,122 runs and are plotted by site in Figure 6 and 7, respectively.

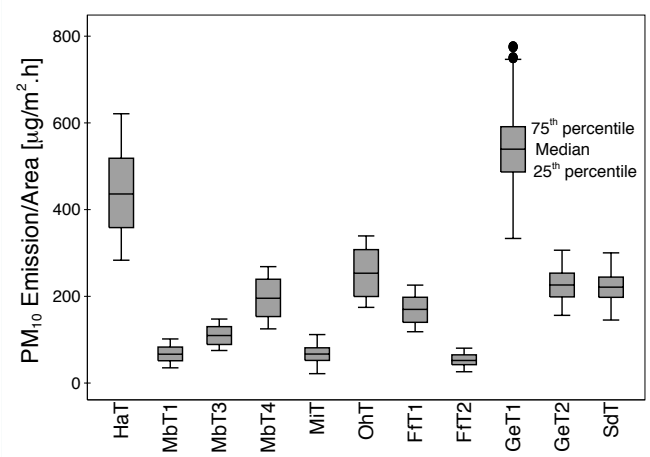

Figure 6. PM_{10} indoor emission source rates normalized by area.

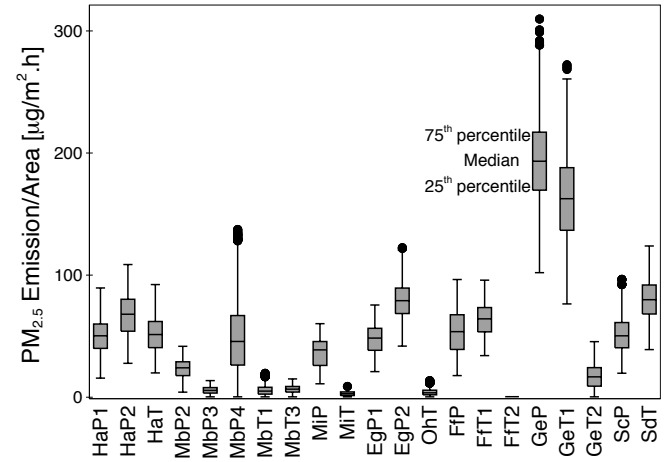

Figure 7. $PM_{2.5}$ indoor emission source rates normalized by area.

Median PM_{10} indoor emission rates ranged from 52 $\mu g/m^2.h$ at Site FfT2 to 539 $\mu g/m^2.h$ at Site GeT1. Median $PM_{2.5}$ indoor emission rates range from 0.42 $\mu g/m^2.h$ at Site FfT2 to 193 $\mu g/m^2.h$ at Site GeP. The highest emission rate for both particle sizes was observed at a grocery store, likely due to the presence of intense cooking activities and heavy foot traffic. The low indoor emission values observed at some sites indicate either that the indoor concentrations were low or they were dominated by outdoor sources.

For stores measured more than once, indoor emission rates serve to explain the difference in the measured indoor concentration, given the measured ventilation rates and outdoor concentrations. An example is the grocery store Brand e measured twice in Texas. The outdoor concentration and the total ventilation rate were comparable in both visits. The median indoor $PM_{2.5}$ emission rate calculated for the first visit (Site GeT1) was 162 $\mu g/m^2.h$ and for the second visit (Site GeT2) was 14 $\mu g/m^2.h$. Because the number of transactions (a proxy for human activity) was a factor of 1.8 higher at the second visit (Site GeT2), the difference of emission rates between these visits may be better explained by differences in the amount of cooking activities. This analysis suggests the complexity of particle sources and sinks in these environments.

Contribution of indoor sources to indoor concentrations

Using the modeled median emission rate, the contribution of indoor sources (versus the contribution of outdoor sources) was estimated using Equation 1 by isolating the portion of the indoor concentration generated from indoor sources. Figure 8 shows bar charts of the contribution of indoor sources to indoor $PM_{2.5}$ and PM_{10} concentrations. Median contribution of indoor sources to $PM_{2.5}$ was 53%. Median contribution of indoor sources

to PM_{10} was higher than that for $PM_{2.5}$ for the majority of the measured sites (median: 83%). This is expected because the efficiency of removal of outdoor particles through HVAC filters is higher for PM_{10} than $PM_{2.5}$.

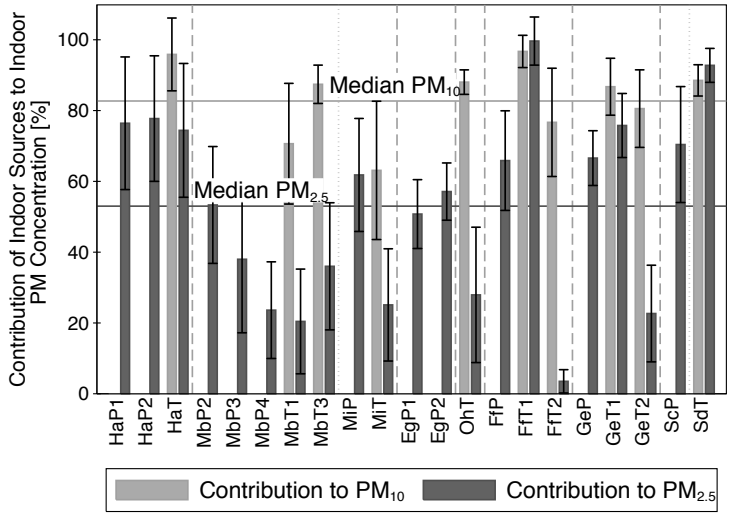

Figure 8. Contribution of indoor sources to indoor PM_{10} and $PM_{2.5}$ particle concentration by site.

As shown in Figure 8, the contribution of indoor sources to indoor concentrations can be significant in some stores; likely causing a limited direct relationship between indoor and outdoor concentrations. For Sites GeP and GeT1, indoor sources, such as cooking activities, contributed significantly to the elevated concentration. These findings show the importance of indoor sources and are consistent with findings reported in the literature. Wu et al. (2012) reported contribution of indoor sources to indoor particle concentrations in small and medium retail and grocery stores located in California. This study found that indoor sources in most of the stores contributed between 67-100% to PM_{10} and between 33-67% to $PM_{2.5}$. For other environments, Abt et al. (2000) assessed the contribution of indoor sources to indoor concentration in four homes in Boston also using the regression

method. This study reported that the indoor sources for indoor particles from 2 to 10 μm contributed to 57-80% of particle concentrations. Outdoor sources can also be significant. Site MbP4 had the largest $PM_{2.5}$ concentration among all the tested sites; here, outdoor sources contributed to 76% of the indoor concentration. The high outdoor concentrations at this test were associated with nearby agricultural burning.

Effect of ventilation and filtration on $PM_{2.5}$ concentrations

Across all sites, the mean and standard deviation of the ventilation rate (ACH) were 0.63 ± 0.37 1/h (ACH for stores located in Pennsylvania: 0.66 ± 0.41 1/h; ACH for stores located in Texas: 0.60 ± 0.33 1/h) and ranged from 0.2 to 1.5 1/h. Shapiro-Wilk test showed that PM concentrations and indoor-to-outdoor ratio variables across all sites did not follow the normal distribution. Spearman Rank Correlation test performed across all sites shows that particle concentrations and I/O ratios measured for all particle sizes did not exhibit a statistically significant relationship with air exchange rates (p values of all the tests was higher than 0.05). Several factors may have contributed to this finding. These mainly include variability in indoor and outdoor sources and removal mechanisms: the small sample size and the fact that the air exchange rate were generally low likely played a role as well. To account for some of these differences, the impact of changing the ventilation rate within the same store on PM concentrations was investigated experimentally in two of the measured general merchandise stores (MiP and MiT). Among the different particle sizes, $PM_{2.5}$ was specifically chosen as the control parameter due to its short- and long-term adverse exposure consequences. Figure 9 contains a scatter plot of indoor $PM_{2.5}$ concentration as a function of ACH. Arrows on the plot

shows the impact of change in $PM_{2.5}$ concentration when going from normal to elevated ACH at these sites.

Increasing the ventilation rate by 54% in the intervention Site MiP resulted in reduction in $PM_{2.5}$ mass concentrations of 90%. At Site MiT, $PM_{2.5}$ mass concentration exhibited the opposite relation: 94% increase in $PM_{2.5}$ concentration with 53% increase in ventilation rate. Indoor emission rates and outdoor concentrations were different between both weeks for both stores. This suggests that determining the minimum ventilation rate to lower indoor PM concentrations should be based on the knowledge of indoor emission rates and outdoor concentrations.

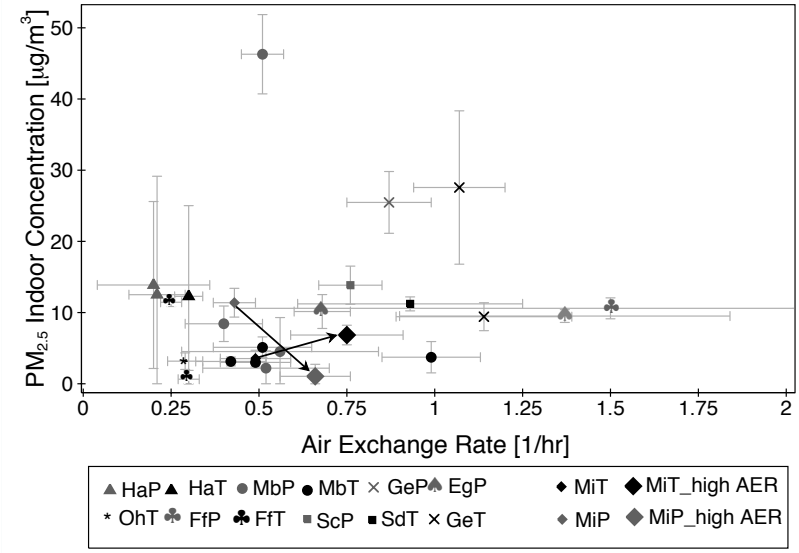

Figure 9. $PM_{2.5}$ indoor mass concentration by air exchange rate (ACH). Arrows visualize the impact on $PM_{2.5}$ concentration when going from normal ACH to high ACH for two Sites: MiP and MiT.

Besides the experimental investigation, indoor $PM_{2.5}$ concentration was modeled for different scenarios of ventilation rates and filtration efficiencies to assess possible control strategies for the sites that exceeded the $PM_{2.5}$ concentration of interest defined in

ASHRAE Standard 62.1-2010 (Appendix B; ASHRAE, 2010). Note that this concentration corresponds to the NAAQS ambient regulatory limit of 12 $\mu g/m^3$ over a one-year average. These sites were grouped according to their indoor-to-outdoor ratio owing to differences in the sources of particles. For sites with IO ratio greater than unity, the magnitude of increasing ventilation rates was estimated, and for sites with IO ratio lower than unity, the impact of decreasing ventilation rates was projected. In addition, the impacts of doubling filtration efficiency with and without changing ventilation rates, and setting the efficiency to maximum with and without changing infiltration rates were modeled. Modeling was conducted by rearranging Equation 1 and assuming the median emission rate from the analysis above, constant outdoor concentrations, and a base case filtration of 35%.

Table 3 summarizes the impact of varying air exchange rate on indoor $PM_{2.5}$ concentration. Indoor $PM_{2.5}$ concentration was set equal to the regulatory limit, and the corresponding ventilation rate was calculated (column 5). If the resulting ventilation rate was negative, this means that changing the ventilation rate with everything else being equal was not sufficient to lower $PM_{2.5}$ below the regulatory limit. For these sites, mechanical ventilation rate was set equal to zero (i.e., ACH was set equal to the infiltration rate) and a "No" is placed in column 7.

Table 3. Air exchange rate measured (existing ACH), air exchange rate according to Standard 62.1-2010 (St-62.1 ACH), minimum air exchange rate estimated (new ACH), air exchange rate multiplier (f), and impact of applying the new ACH on $PM_{2.5}$ concentrations across the sites where $PM_{2.5}$ was higher than $PM_{2.5}$ concentration of interest defined in ASHRAE Standard 62.1-2010 (12 µg/m^3).

Sites where C_{in}>12[1] µg/m^3	Description	Control Strategy	Existing ACH[2]	St-62.1 ACH	New ACH[3]	f= New/Existing ACH	With New ACH, was C_{in}≤12[1] µg/m^3?
HaP1	Sites where C_{out}>C_{in}	Decrease ACH	0.21	0.53	0.05	0.25	Yes
HaP2			0.2	0.53	0.04	0.22	No
HaT			0.3	0.57	0.10	0.34	Yes
MbP4			0.51	0.7	0.32	0.62	No
GeP	Sites where C_{out}<C_{in}	Increase ACH	0.87	0.27	7.66	8.80	Yes
GeT1			1.07	0.48	13.37	12.49	Yes
ScP			0.76	0.54	1.21	1.59	Yes

[1] $PM_{2.5}$ concentration of interest defined in ASHRAE Standard 62.1-2010.
[2] Existing ACH is the measured ACH.
[3] New ACH is the modeled minimum ventilation rate that can achieve the lowest possible indoor $PM_{2.5}$ concentration.

Sites HaP1, HaP2, HaT and MbP4 had outdoor concentrations higher than indoor concentrations and the indoor concentrations were above the regulatory limit. For these sites, it was favorable to keep the mechanical ventilation rate to a minimum: the multiplier f (New ACH/ACH) of the ventilation rate is less than 1.0. However, for Sites HaP2 and MbP4, setting the mechanical outdoor ventilation rate to zero was not sufficient to drive indoor $PM_{2.5}$ below the regulatory limit due to the high infiltration rates observed at these sites. Furthermore, the consequence of lowering ventilation for other pollutants may be an obvious issue at these sites. For the remaining sites (GeP, GeT1, and SdT) an increase in ventilation rate (up to 12 times in Site GeP) was favorable to lower indoor $PM_{2.5}$ mass concentrations below the reference level. This increase would require large air exchange rates that are infeasible to attain.

Table 4 summarizes the impact of varying the filtration efficiency with or without variation of ACH and infiltration rates for the sites where the indoor concentration

exceeded that limit. Table 4 shows that doubling the filter efficiency (i.e., η=70%) and maintaining the same ventilation rate was not sufficient to drive $PM_{2.5}$ below regulatory limits for Sites MbP4, GeP, and GeT1 (column 3). Setting the filter efficiency to 100% lowered $PM_{2.5}$ concentrations but was still not enough to drive $PM_{2.5}$ in Sites MbP4, GeP, and GeT1 below the regulatory limit (column 4). In these stores, high infiltration rates accounted for a large fraction of the air entering the retail building and both grocery stores also had a high emission rate. Both of these terms were not subjected to filtration. Eliminating the infiltration rates (i.e., with positive pressurization) and using a highly efficient filter would drive the concentrations in all sites below the regulatory limit (column 5). If the ventilation rate is set as the new ventilation rate (column 5 in Table 3) and efficiency is doubled, then the concentrations would be less than the regulatory limit for all sites except Site MbP4 (column 6), where outdoor sources were significant (Outdoor $PM_{2.5}$ = 60 μg/m^3).

Table 4. Impact of doubling the efficiency, setting the efficiency to maximum, setting the efficiency to maximum with assuming zero infiltration rates, and doubling the efficiency with applying the minimum air exchange rate estimated (new ACH) on $PM_{2.5}$ concentrations across the sites where $PM_{2.5}$ was higher than $PM_{2.5}$ concentration of interest defined in ASHRAE Standard 62.1-2010

Sites where C_{in}>12[1] μg/m^3	Control Strategy / Description	After applying the control strategy, was C_{in}≤12[1] μg/m^3?			
		η=70%[2] & Existing ACH[3]	η=100%[2] & Existing ACH[3]	η=100%[2] & infiltration=0 & Existing ACH[3]	η=70%[2] & New ACH[4]
HaP1	Sites where C_{out}>C_{in}	Yes	Yes	Yes	Yes
HaP2		Yes	Yes	Yes	Yes
HaT		Yes	Yes	Yes	Yes
MbP4		No	No	Yes	No
GeP	Sites where C_{out}<C_{in}	No	No	Yes	Yes
GeT1		No	No	Yes	Yes
ScP		Yes	Yes	Yes	Yes

[1] $PM_{2.5}$ concentration of interest defined in ASHRAE Standard 62.1-2010..

[2] η is the filtration efficiency (both recirculation and outdoor air gets filtered).
[3] Existing ACH is the measured ACH.
[4] New ACH is the modeled minimum ventilation rate that can achieve the lowest possible indoor $PM_{2.5}$ concentration.

In the previous calculations, emission rates and outdoor concentrations were assumed to be constant; this assumption is only valid for the duration of the mobile sampling period. From the observed diurnal variation, emission rates varied during the day and night owing to episodic indoor activities related to consumer and employee activity. The complexity of these interconnected factors suggests a requirement of understanding indoor sources and infiltration rates and balancing these inputs with cost of ventilation rates and/or cost of having a higher efficiency filter.

CONCLUSIONS

Retail stores are important sites of exposure to particles and indoor particle sources are an important contributor to this exposure. Observed particle concentrations in retail environments were generally lower to other building types, although concentrations exceeded ambient regulatory limits in several stores. Stores with high concentrations of particles were generally associated with cooking ($PM_{2.5}$ and PN_1), recent cleaning (PN_1), heavy foot traffic (PM_{10} and $PM_{2.5}$), as well as high outdoor concentrations (all sizes). The direction and magnitude of changing ventilation rate to lower $PM_{2.5}$ indoor concentrations depends on the indoor-to-outdoor ratio. Infiltration levels and high emission rates present at some sites diminished the effect of high filtration efficiency. Depending solely on ventilation rates to lower indoor concentration in the sites where indoor to outdoor concentration ratios were higher than unity might not be economically attractive. Eliminating the infiltration rates (i.e., with positive pressurization) and using a

highly efficient filter would drive the concentrations in most retail stores, including stores with elevated outdoor concentrations, below the $PM_{2.5}$ concentration of interest of 12 $\mu g/m^3$.

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Paper 1 Supporting Information

A. Summary of calibration factors for PM_{10} and $PM_{2.5}$ by instrument used

B. Summary of Aerotrak collocation data and calibration factors

C. Summary of instruments used for particle sampling and HVAC characterization

D. Bar graph of PM_{10} indoor and indoor-to-outdoor ratio by site

E. Emission rate sensitivity analysis for unknown parameters

A. Summary of calibration factors for PM_{10} and $PM_{2.5}$ by instrument used

A.1 *TSI Dusttrak:* We calculated the calibration factors for the TSI Dusttrak from collocated measurements made with the Thermo Scientific TEOM at Site MiT. At concentrations below 8 $\mu g/m^3$, we assumed the variance of the calibration factors to be a linear function of the concentration. We determined the variance by fitting a line to the standard deviations of the calibration factors as a function of concentration. At concentrations above 8 $\mu g/m^3$, we assumed the variance of the calibration factors to be equal to the instrument resolution, 1 $\mu g/m^3$. We calculated the overall uncertainty of the calibration factors using standard error propagation to include the variance of the calibration factors, and the uncertainty in the measurements of the Thermo Scientific TEOM. Table 1 summarizes the calibration factors, the uncertainty of the calibration, and the calibration equation as a function of the TSI Dusttrak measured concentration.

Table 1 TSI Dusttrak calibration equation as a function of the concentration

Instrument	PM Size	Calibration Equation
TSI Dusttrak	PM_{10}	If concentration ≤ 8 μg/m^3 Uncertainty in calibration (UC) = $\sqrt{(-0.16\times \text{concentration}\left[\mu\text{g/m}^3\right]+1.39)^2+2^2}$ $\sqrt{(-0.16 \text{ x concentration } [\mu \text{ g/m3}] + 1.39)^2 + 2^2}$Cal. concentration [μg/m^3] =concentration [μg/m^3]×(2.23±UC) If concentration > 8 μg/m^3 Cal. concentration [μg/m^3] =(concentration [μg/m^3] × 2.23)±2.23[μg/m^3]

A.2 TSI Sidepak: For the TSI Sidepak, we determined the calibration factors to be dependent on the measured particle concentration. For concentrations less than 3 μg/m^3, we calculated the calibration factor from collocated measurements made with the Thermo Scientific TEOM. We performed the collocation at low particle concentrations at the UT test house. The manufacturer reported a detection limit for the TSI Sidepak of 1 μg/m^3, however data collected during the collocation indicated that the minimum detection limit was approximately 3 μg/m^3. We calculated this value from the average concentration measured by the TEOM when the TSI Sidepak recorded a concentration equal to zero. For particle concentrations greater than 10 μg/m^3, we used the calibration factor reported by Jiang et al. (2011) for urban outdoor areas. We assumed the transition from the calibration factors for low concentrations (≥3 μg/m^3) to the calibration factors for high concentrations (≥10 μg/m^3) to be linear. We assumed the uncertainty for the calibration factors to be equal to the calculated minimum detection limit of 3 μg/m^3 for concentrations greater than 3 μg/m^3. For concentrations below 3 μg/m^3, we calculated the overall uncertainty of the calibration factor using the standard error propagation to

include the variance of the calibration factors, and the uncertainty of the Thermo Scientific TEOM itself. Table 2 summarizes the calibration factors, the uncertainty of the calibration, and the calibration equation as a function of the TSI Sidepak measured concentration.

Table 2 TSI Sidepak calibration equation as a function of the concentration

Instrument	PM Size	Calibration Equation
TSI Sidepak	$PM_{2.5}$	If concentration < 3 $\mu g/m^3$ Cal. concentration [$\mu g/m^3$] =(concentration [$\mu g/m^3$] × 3)±3.2 [$\mu g/m^3$] If 3 $\mu g/m^3 \leq$ concentration $\leq$ 10 $\mu g/m^3$ calibration factor(CF)=-0.31x concentration+3.94 Cal. concentration [$\mu g/m^3$] =(concentration [$\mu g/m^3$] × CF)±3 [$\mu g/m^3$] If concentration $\geq$ 10 $\mu g/m^3$ Cal. concentration [$\mu g/m^3$] =(concentration [$\mu g/m^3$] × 0.8)±3 [$\mu g/m^3$]

A.3 Met One Aerocet: We calculated the calibration factors for both PM sizes reported by the Met One Aerocet from collocated measurements made with gravimetric time-integrated filters at Sites HaT, MbT1, GeT2, and MbT3. The gravimetric data were collected using SKC personal environmental monitors (PEMs) with Pall Emfab 37mm filters. The reported calibration factors are generally in agreement with reported literature values. We assumed the uncertainty of the calibration to be one standard deviation of the calculated calibration factors. Table 3 summarizes the calibration factors, the uncertainty in the calibration factors, and the calibration equation as a function of the concentration measured by the Met One Aerocet.

Table 3 Met One Aerocet calibration equation as a function of the concentration.

Instrument	PM Size	Calibration Equation
Met One Aerocet	PM_{10}	Cal. concentration [$\mu g/m^3$] =(concentration [$\mu g/m^3$] × (1.63±0.76)
	$PM_{2.5}$	Cal. concentration [$\mu g/m^3$] =(concentration [$\mu g/m^3$] × (5.53±1.73)

B. Summary of Aerotrak collocation data and calibration factors

We employed at least two TSI Aerotraks for all UT tests. We used the TSI Aerotraks to simultaneously measure indoor and outdoor particle concentrations. Over the course of this study, UT used five different TSI Aerotraks. Since we used one TSI Aerotrak (TSI Aerotrak Number 2) at all UT sites, we selected it as the reference monitor. At five of the test sites, we collocated TSI Aerotrak number 2 with the other TSI Aerotrak monitors used in this study. We evaluated the variance between TSI Aerotrak monitors using the relative percent deviation (RPD) with TSI Aerotrak number 2 as the reference. We calculated the RPD using 15-minute averages for each bin size using Equation 1.

$$RPD(i,b) = \frac{\left|C_i - C_r\right|_b}{0.5 \times (C_i + C_r)_b} \times 100 \quad (1)$$

Where:
C = the number concentration measured by TSI Aerotrak [$\#/cm^3$],
i = the index for the time-series measurements of the TSI Aerotrak monitors,
b = the bin size, and
r = the index for the time-series measurements of the reference TSI Aerotrak monitor.

Figure 1 contains a box plot of RPD for all monitors used relative to monitor 2 for the six bin sizes. The bottom of the box indicates the 25th percentile the horizontal line indicates the median and the top of the box the 75th percentile. The whiskers indicate the data range within 1.5 times the interquartile range of the 25th and 75th percentile. Filled circles are

outliers. Generally, the largest variance was observed for bin 6 (>10μm) ranging from 35% to 83%.

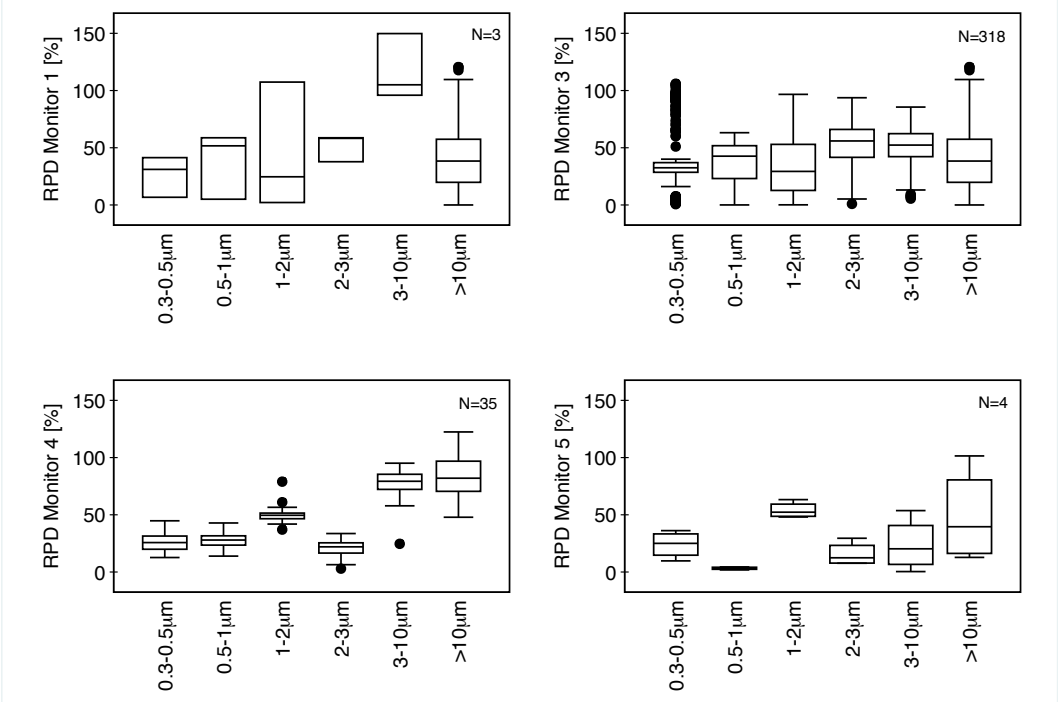

Fig. 1 Relative variance deviation between the TSI Aerotrak monitors used and the reference monitor.

We calculated the calibration factor for each instrument using 15-minute averages for each bin size using Equation 2, as shown below.

$$CF(i,b) = \overline{\left(\frac{C_{i,b}}{C_{r,b}}\right)} \tag{2}$$

Where:
C = the number concentration measured by TSI Aerotrak [#/cm^3],
i = the index for the time-series measurements of the TSI Aerotrak monitors,
b = the bin size, and
r = the index for the time-series measurements of the reference TSI Aerotrak monitor.

We calculated the uncertainty in the calibration factors using the variance observed between the collocated TSI Aerotrak and the measurement variance observed for the reference TSI Aerotrak. We assumed the uncertainty associated with the collocation to be one standard deviation of the calculated calibration factors. We calculated the variance of reference monitor (TSI Aerotrak Number 2) using data collected by this monitor at night at Test 24 where the particle distribution was assumed to be constant. The variance observed for Aerotrak Number 2 is presented in Table 4.

Table 4 Variance of Aerotrak Number 2 at night at Test 24 when the particle distribution was approximately constant.

Bin Size	Variation [%]
0.3-0.5µm	2.4
0.5-1µm	5.3
1-2µm	27
2-3µm	52
3-10µm	67
>10µm	100

Table 5 shows the calibration factors and standard deviation for each instrument and bin size. The data in Table 4 along with the variance in calibration factors were calculated using the standard error propagation to determine the overall uncertainty for the TSI Aerotrak calibration factors shown in Table 5.

Table 5 Calibration factor (CF) and standard deviation for each TSI monitor used at the UT sites

Monitor UT	CF±Standard Deviation 0.3-0.5 µm	0.5-1 µm	1-2 µm	2-3 µm	3-10 µm	>10 µm	Num. Hours of Collocation
1	1.11±0.4	1.1±0.58	0.87±0.56	1.28±0.93	2.54±4.19	2.87±3.94	0.75
3	1.59±0.57	1.48±0.28	0.74±0.26	1.78±0.97	1.73±1.2	1.54±1.64	79.5
4	1.32±0.13	1.34±0.11	0.6±0.17	1.22±0.64	2.28±1.56	2.52±2.59	8.75
5	1.28±0.15	1.01±0.06	0.58±0.16	1.13±0.61	1.3±0.93	1.82±2.02	1

We used only one TSI Aerotrak at the PSU test sites. For each size fraction, we estimated the calibration factor for this instrument as the average of all the calibration factors obtained for the TSI Aerotrak monitors used at the UT test sites for the same size bin. Similarly, for each size fraction we assumed the standard deviation for the calibration factor to be the average of the standard deviations observed for the UT monitors. Table 6 shows the calibration factors and the standard deviations for the TSI Aerotrak used at the PSU sites.

Table 6 Calibration factor (CF) and standard deviation for the TSI Aerotrak monitor used at the PSU sites

Monitor UT	CF±Standard Deviation 0.3-0.5 µm	0.5-1 µm	1-2 µm	2-3 µm	3-10 µm	>10 µm	Num. Hours of Collocation
1	1.32± 0.31	1.23±0.26	0.7± 0.29	1.35± 0.79	1.96± 1.97	2.19±2.55	-

We determined the calibrated response for each TSI Aerotrak per bin using Equation 3:

$$\text{Calibrated } C_{i,b} = C_{i,b} \times \left(CF_b \pm \text{standard deviation}_b\right) \quad (3)$$

Where:
C = the number concentration measured by TSI Aerotrak [#/cm^3],
i = the index for the time-series measurements of the TSI Aerotrak monitors, and
b = the bin size used.

C. Summary of instrument used for particle sampling and HVAC characterization

The instruments used for each type of measurement, along with the manufacturer reported resolution and uncertainty are summarized in Table 7.

Table 7 Summary of instruments used for particle sampling and HVAC characterization

Measurement	Time Resolution	Instrument	Uncertainty
$PM_{2.5}$ &PM_{10} particle mass concentrations	Integrated >48 hrs	SKC Personal Environmental Monitor PEM 2.5 µm 761- 203B & PEM 10 µm 761-200B	
	Continuous 5 min	TSI Sidepak Personal Aerosol Monitor AM510	

	Continuous 5 min	TSI Dustrak Aerosol Monitor 8520	
	Continuous 2 min	Met One Aerocet-531 Mass Particle Counter/ Dust Monitor	Accuracy = ± 10%, to calibration aerosol
	Continuous 1 min	TEOM-1405D Dichotomous Continuous Ambient Particulate Monitoring	Accuracy= ±0.75% Precision = ±2.0µg/m^3 (one-hour average), ±1.0µg/m^3 (24-hour average)
Submicron (0.02 to 1 µm) particle number concentrations	Continuous 30 sec	TSI P-TRAK submicron Particle Counter 8525	
Particle size distribution six channels between 0.3 µm and >10 µm	Continuous 5 min	TSI Aerotrak 8220	Counting efficiencies= 50%±10% at 0.3µm 100% by 0.45µm 50%±20% at all calibration cut sizes, Zero count≤1 particle counted in 5 minutes, coincidence loss= ±5% at 2,000,000 particles/ft^3
		TSI Aerotrak 9306	
Differential pressure	Continuous 1 min	Energy Conservatory DG-500 and DG-700	±1%
CO_2 concentration	Continuous 5 min	Telaire Model 7001	±50 ppm
	Continuous 5 min	TSI Q-Trak Indoor Air Quality Monitor 7565-X	The larger of ±3% of reading or ±50 ppm
Pumps		Buck Libra-plus LP-12 pumps	±5% flow
		Sensidyne Gilian AirCon-2 sampling pump 801012 -100	
		Buck LinEair 40	
Volumetric flow		Gilian Gilibrator	±1% reading
RTU flow		Trueflow air handler flow meter	±1% reading when used with Energy Conservatory DG-500 and DG-700
SF_6 concentration		Lagus Autotrac SF_6 analyzer Model 101	Manufacturer accuracy: ±5%; precision: ±1%; and calibration gas accuracy: ±2%

D. Bar graph of PM_{10} indoor and indoor-to-outdoor ratio by site

Bar chart of indoor PM_{10} mass concentration by site is shown in Figure 2. The bar height represents the arithmetic mean observed during the mobile sampling event, the uncertainty displayed on the graph is the standard deviation of the data collected during

the mobile sampling event. The maximum observed concentration is listed at the top of the standard deviation line.

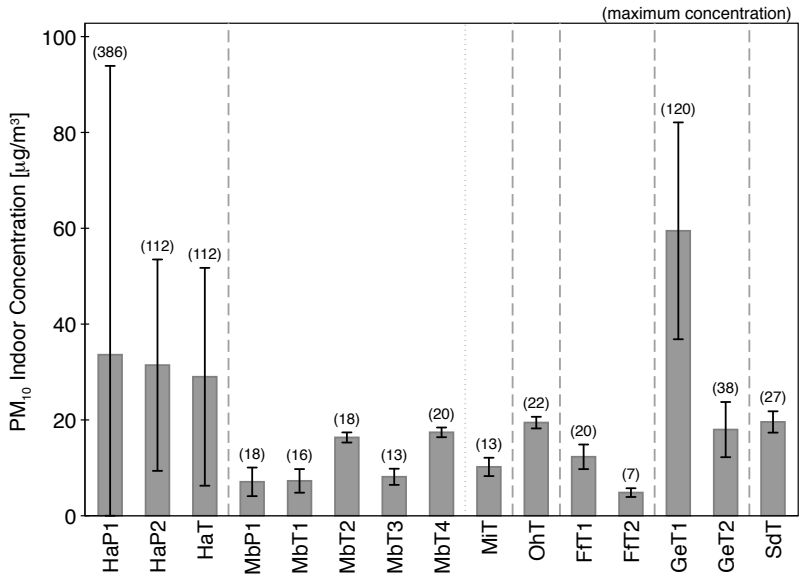

Fig. 2 PM_{10} indoor mass concentration by site, and PM_{10} maximum measured concentration is presented in parentheses.

Figure 3 contains bar chart of the indoor-to-outdoor ratio of PM_{10} by site. The uncertainty displayed on both graphs is the standard deviation of the data collected during the mobile sampling period. The 4-hour average outdoor concentration is listed at the top of the standard deviation line.

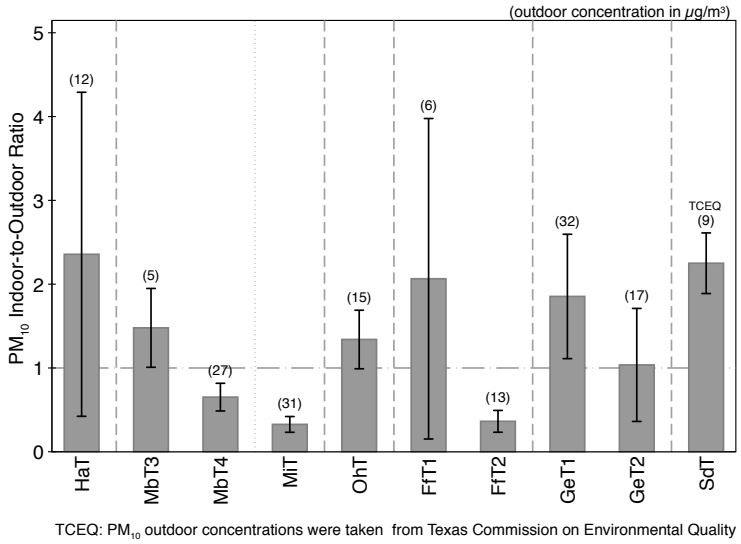

Fig. 3 PM_{10} indoor-to-outdoor ratio by site, and PM_{10} measured outdoor concentration is presented in parentheses.

C. Emission rate sensitivity analysis for unknown parameters

Additional tests were run to test the importance of varying efficiency and deposition values on $PM_{2.5}$ and PM_{10} emission rates. For $PM_{2.5}$, results indicate that changing the deposition value by a factor of 2 (range: 0.12 to 0.18 1/h) does not impact the emission rate. Note that the high deposition value (0.36 1/h) that constituted one of the deposition values considered in the runs to calculate emission rates corresponds to the deposition value reported by the PTEAM study (Özkaynak et al., 1997). However, doubling the filter efficiency (range: 11% to 23%) with measured indoor concentration being the same indicated that the median emission rate across sites was approximately 1.6 times higher. This finding suggests the importance of accurately predicting the efficiency value for $PM_{2.5}$. For PM_{10}, both the deposition and the efficiency values did not vary more than 5% and therefore did not significantly impact the indoor emission rate. The reason is that deposition and efficiency values do not vary considerably for large particle sizes.

Appendix C

Paper 2: The Relationship between Filter Pressure Drop, Indoor Air Quality, and Energy Consumption in Rooftop HVAC Units

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(Submitted to Building and Environment)

ABSTRACT

HVAC filters are commonly used to decrease exposure to particulate matter, yet little is known about the energy impacts and air quality consequences of high efficiency filters installed in commercial buildings. To explore these effects, system airflow, filter and coil pressure drop, fan pressure rise, and power draw were measured, and cooling capacity and compressor power were modeled for at least four filter pressure drops in 15 rooftop units equipped with and without fan speed control. Energy implications and clean-air-delivery-rate were estimated for a large dataset of filters divided into four efficiency (MERV) categories. Field measurements conducted on units without fan speed control showed that increased filter pressure drop decreased flow, cooling capacity, and power. For a unit with fan speed control, the same increase in pressure drop resulted in the same magnitude change of fan power but in the opposite direction, and other parameters were unchanged. Replacing MERV 8 with MERV 13/14 resulted in higher energy consumption (2-4%) during cooling mode for both unit types, energy savings during fan-only mode (8-13%) in units without fan speed control, and increased energy consumption in fan-only mode (11-18%) in the unit with fan speed control. Energy consumption

increases were offset by improvement in clean-air-delivery-rate, especially for $PM_{2.5}$ (2.9-3.8 times increase going from MERV 8 to MERV 13/14), with larger benefits achieved for the unit with fan speed control. A comprehensive understanding of the impact of filtration is essential to selecting the appropriate efficiency of filters that ensures low-energy use and healthy buildings.

Keywords

Filtration; Packaged terminal air conditioner (PTAC); HVAC energy use; Clean air delivery rate; Commercial buildings.

1. INTRODUCTION

Particles are an important pollutant of concern in commercial buildings due to their significant health effects. To decrease particles of outdoor and indoor origin, the use of high-efficiency heating ventilation and air-conditioning (HVAC) filters is often recommended as an alternative to supplying additional ventilation because such filters (1) can lower particle matter (PM) concentrations in a less energy intensive way and (2) are effective even when outdoor concentrations of PM are high. However, high-efficiency HVAC filters generally have a higher-pressure drop and are presumed to have large energy penalties. This is particularly important because fan energy necessary to move air throughout commercial buildings equipped with rooftop units accounts for at least 7% of total site energy consumption (261 trillion BTUs annually; EIA, 2003; DOE, 2012). As energy conservation becomes essential, building designers and operators need to understand the operational power requirements and the underlying benefits in improving indoor air quality through the use of high efficiency filters.

Quantifying the relationship between filter pressure drop, indoor air quality, and energy cost of air filters has been the focus of a number of studies. Field, lab, and computer simulation work have mostly focused on small systems (<30 kW) with no fan speed control (Palani et al., 1992; Rodriguez et al., 1996; Parker et al., 1997; Yang et al., 2006; Stephens et al., 2010[a&b]; Noh & Hwang, 2010; Proctor et al., 2011; Nassif et al.; 2012). Only one recent field study tested small systems equipped with and without fan speed control in residences (Walker et al., 2013). Three small-scale studies of large commercial systems investigated the effect of reducing the pressure drop across filters on energy consumption (Chimack & Sellers, 2000; Lam et al., 2006; Nassif, 2012). Of those studies, however, only Lam et al. (2006) performed field measurements. They conducted a field study on a 40-floor building with a centralized HVAC system; however, the results from this study are difficult to apply to most of buildings in U.S. that are smaller, because the filter pressure drop on the studied 40-floor building is negligible when compared to the high static pressure drop of the entire supply ducting system of this building.

Other studies have focused on determining an approximate cost of filter operation (life cost analysis) by applying a filter model (Fisk et al., 2002; Arnold et al., 2005; Bekö et al., 2008; Mayer et al., 2008; Sun & Woodman, 2009; Sun, 2010; Montgomery et al., 2012). In these studies, the filter models used assumed the airflow rate through the filter to be constant over the life of the filter, and therefore the impact on power for systems without fan speed control was not captured. In addition, the actual system operating point of the fan and duct curves and the resulting system efficiency can significantly impact the

fan power consumption. Also, the fan efficiency reported in the literature varies widely, with values that range from 22-49% for residential and light-commercial buildings (e.g., Stephens et al., 2010[a]), and 30-80% for modeled commercial systems (e.g. Chimack & Sellers, 2000; Fisk et al., 2002; Yang et al., 2006; Bekö et al., 2008).

There is a clear lack of measured data and analysis of energy and indoor air quality consequences of filters used in big commercial systems (rated cooling capacity>30 kW) equipped with or without fan speed control. The purpose of this paper is to (1) quantify the relationship between filter pressure drop and fan pressure, airflow rate, power, and efficiency through fieldwork conducted on rooftop units equipped with and without speed control; (2) estimate the impact of filter pressure drop on cooling capacity and compressor power through modeling the vapor compression cycle; and (3) compare the energy and air quality performance of different filtration efficiencies in these systems.

2. METHODOLOGY

2.1 Data collection/simulation: Impact of pressure drop on unit performance

The sample of buildings for which field data were collected includes 14 rooftop units (RTUs) equipped with no fan speed control and one unit equipped with fan speed control. These units were installed in big box retail stores in Austin, Texas that were tested as part of ASHRAE RP-1596 (Siegel et al., 2013). Cooling capacity ratings on these units ranged from 30 to 84 kW. For each sampled system, system airflow, filter and coil pressure drop, fan pressure rise, and power draw were measured for at least four different filter pressure drops (induced by blocking airflow at the filter). Duct leakage impacts were excluded

because HVAC distribution systems in retail stores typically have almost no ducts and instead deliver conditioned air directly to the space.

System airflow rates were measured with an Energy Conservatory TrueFlow metering plate and DG-700 digital manometer (with uncertainty of ±7% of measured value) for the baseline pressure drop. Change in supply static pressure for different filter pressure drops were measured and, following the calculation procedure in the instrument manual, corrections were made based on changes in the supply plenum static pressure. Also, for baseline and each elevated pressured drop, filter and coil pressure drop and fan pressure rise were measured using a DG-700 manometer (±1% measurement uncertainty). An Onset HOBO Energy Logger was used to record the power draw of the air handler fan measured by a Continental Control Systems (CCS) Wattnode AC true power meter for approximately 30 minutes at 10-second intervals (±3% measurement uncertainty). The Energy Logger box was connected to pressure taps, voltage taps, and 0 to 20 amp CCS current transducers. For the unit equipped with fan speed control, fan speed was adjusted manually using the variable frequency drive installed on the unit to maintain approximately the same airflow rate delivered by the fan for each measured filter pressure drop. Using the pressure rise across the fan, the airflow rate, and fan power, the fan efficiency was calculated for different pressure drops.

The impact of filter pressure drop on cooling capacity, sensible heat ratio, and compressor power were modeled using two vapor compression models: the ACHP vapor compression system model (Braun, 2013) and Secondary HVAC Toolkit software (Nassif, 2013). Two additional parameters, outdoor temperature and relative humidity,

were also measured during field work and were used as inputs for the model. Other inputs include geometry of coils, refrigerant type (obtained from manufacturer specifications), and fan power and airflow rate (obtained from the collected dataset). Comparison of the modeling results of the two models showed very similar results, and in this paper the results from the ACHP vapor compression system model are reported.

To capture and analyze the effect of different filter pressure drops on system runtime and the total power draw, the collected and simulated data were used to calculate the change in length of the cooling cycle (i.e., duty cycle) and the energy efficiency ratio (EER). The change in duty cycle is used to see how much the system will run longer to achieve the same cooling capacity when the system airflow is decreased relative to the baseline case; the total power draw is then multiplied by the additional length cycle. Also, the energy efficiency ratio (EER) was calculated to assess the change in the net cooling capacity (difference between modeled cooling capacity and measured evaporator-side fan power) divided by the total power input (measured fan power, modeled compressor and condenser power) relative to the base case.

2.2 Energy and indoor air quality comparison of different efficiency filters

The established relationships between (1) filter pressure drop and (2) system parameters (including: airflow, fan speed, fan power, cooling capacity, compressor power, and duty cycle) study results were presented in the context of comparing the energy and particle reduction performance of different filters. Data from sixty filters obtained from 15 different manufacturers and an additional 15 filters provided by Rivers and Murphy (1999) was investigated. For each filter, filtration efficiency data was

obtained. Then, these filters were classified with an ASHRAE Standard 52.2-2012 Minimum Efficiency Reporting Value (MERV) and the sample consisted of MERV 8 (n=16), MERV 11 (n=21), MERV 13 (n=24), MERV 14 (n=14) filters. It should be noted that Rivers and Murphy conducted filter traverse tests in 1996 and thus did not necessarily do full test according to Standard 52.2-2012. Accordingly, the MERV values that would results from a Standard 52.2 test may be slightly different. Besides conducting tests at clean conditions, Rivers and Murphy (1999) provided an extra set of ASHRAE tests for laboratory-fouled filters that were loaded with a standard test dust. In all cases, fractional efficiency (η) for three particle size bins (E_1: 0.3 μm -1 μm; E_2: 1 μm - 3 μm; E_3: 3 μm - 10 μm) was reported by filter manufacturers or in the case of Rivers and Murphy it was calculated by averaging efficiencies of particle sizes within each corresponding size range group. Filter efficiencies were typically measured at a face velocity of 2.54 ms^{-1}, which is generally higher than the velocities considered in this work and typically found in rooftop HVAC units. Based on findings from Rivers and Murphy (1999) and Hanley et al. (1994) on the relationship between velocities and filter efficiency for particle sizes larger than 0.3 μm, the impact of velocity on filter efficiency was determined to be minimal and the filtration efficiency at 2.54 ms^{-1} was used with no further adjustment.

Filter efficiency for integrated particle sizes $PM_{2.5}$ and PM_{10} was calculated using Equation 1; where the efficiency for each particle size (η_i) was combined with indoor ($N_{in,i}$) and outdoor ($N_{out,i}$) particle size distribution, particle geometric mean (GM_i), and outdoor air fraction (OA); these parameters were all measured in ASHRAE RP-1596

(Siegel et al., 2013). In the present work, particle density was assumed to be 1 gcm^{-3} for all particle sizes.

$$\eta = OA \times \sum(\eta_i \times N_{out,i} \times GM_i{}^3) / \sum N_{out,i} \times GM_i{}^3 + (1 - OA) \times \sum(\eta_i \times N_{ind,i} \times GM_i{}^3) / \sum N_{ind,i} \times GM_i{}^3 \quad (1)$$

Using the test reports, a second-degree polynomial fit was established for each filter between pressure drop and the face velocity of the system (p=constant$_1 \times V^2$+ constant$_2 \times V$; where p is the pressure in Pa and V is the face velocity in ms^{-1}). Three different face velocities were chosen for this analysis: 0.6, 0.9, and 1.2 ms^{-1}. Velocities of 0.9 and 1.2 ms^{-1} constitute the range of design face velocities adopted by rooftop unit manufacturers for different unit sizes, and 0.6 ms^{-1} was the lowest face velocity observed during fieldwork. A one-way analysis of variance (ANOVA) was conducted to check if there was a significant difference between the parameters analyzed and the three face velocities of the RTU. The analysis was repeated for the different cases for units equipped with and without fan speed control. For all parameters and both type of units, the difference was not significant ($p>0.05$). Therefore, it is reasonable to assume that the conclusions established for RTU face velocity of 0.9 ms^{-1} (including the effect of fouling) still hold for RTU velocity of 0.6 ms^{-1} and 1.2 ms^{-1}. Results only show the findings from face velocities of 0.9 ms^{-1}; tables detailing the impact of replacing MERV 8 with MERV 11, 13, 14 on the studied parameters for the other two velocities (0.6 and 1.2 ms^{-1}) are provided in the Supplementary data.

The energy performance of filters was calculated for cooling mode by incorporating

the fan power and the compressor power; and for fan-only mode by considering only the fan power to depict economizer mode and ventilation modes. The percentage contribution of fan power to the total power was calculated from the collected dataset for each measured unit. The indoor air quality performance of filters was calculated using the clean air delivery rate (CADR; e.g., MacIntosh et al., 2008), which takes into account the change in efficiency (as calculated above) as well as the change in airflow through the filter (as measured for multiple pressure drops).

The energy and air quality performance of MERV 8 filters was compared to that of higher efficiency filters (MERV 11, MERV 13, and MERV 14). The MERV 8 filter was chosen as the basis of comparison as it is the most commonly used efficiency in rooftop units in commercial buildings. Different MERV 8 filters (i.e., filter pleating, depth, and area) have different resistances and do not perform the same way on all particle size ranges. To capture the performance of different filters with the same MERV 8 category, five base cases were chosen to cover a wide range of pressure drop and efficiencies encountered in the marketplace for filters with a MERV of 8. Figure 1 summarizes the pressure drop and integrated $PM_{2.5}$ and PM_{10} particle efficiencies for these five filters.

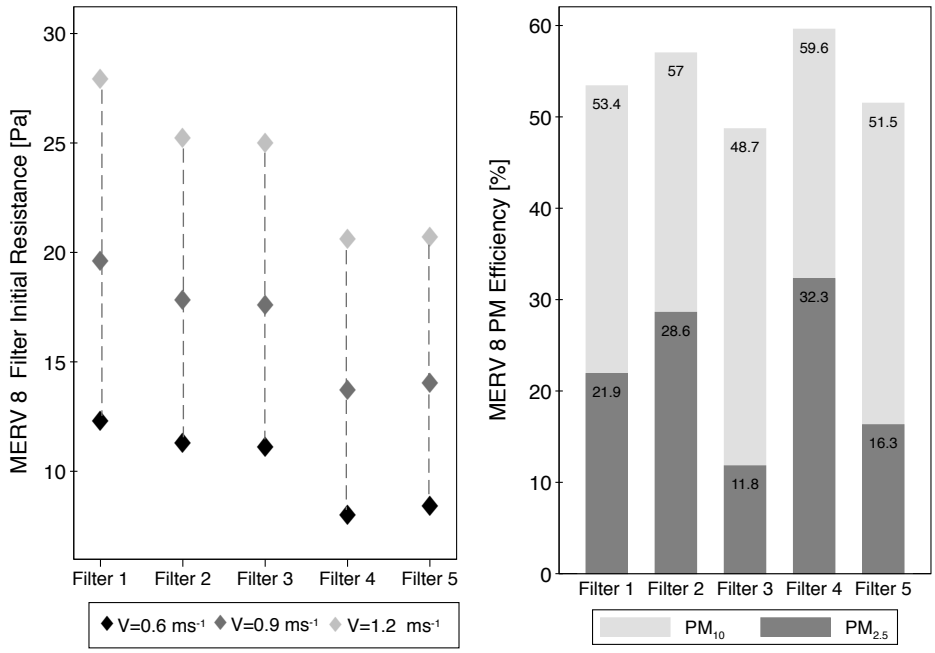

Figure 1. Initial resistances at face velocities of 0.6, 0.9, and 1.2 ms^{-1} (left), and $PM_{2.5}$ and PM_{10} efficiencies (right) for five MERV 8 base case filters.

3. RESULTS AND DISCUSSION

This section details a summary of sampled rooftop units characteristics, field measurements and modeling results of the impact of increased pressure drop on units without and with fan speed control (Section 3.1), analysis of the impact of replacing a MERV 8 filter with a higher MERV filter on energy and IAQ for clean filter conditions (Section 3.2), and for fouled conditions (Section 3.3).

Table 1 summarizes the capacity, airflow, and face velocities for the sampled units measured in this study. In total, there were 15 rooftop units; the first unit shown in Table 1 was the only unit equipped with fan speed control. Across all units, the rated cooling capacity ranged between 30 and 84 kW; the airflow normalized by the nominal cooling

capacity ranged between 78 to 187 $m^3kW^{-1}hr^{-1}$; and the face velocity ranged between 0.6 and 1.2 ms^{-1}.

Table 1. Summary of units tested with and without fan speed control.

Nominal Capacity [kW]	Number of Units Tested	Flow/Nominal Capacity [$m^3kW^{-1}hr^{-1}$]	Face Velocity [ms^{-1}]
84.4	1[1]	78±5	0.6±0.04
70.3	7	104±7 - 183±13	0.93±0.07 - 1.1± 0.08
61.5	3	82±6 - 115±8	0.68±0.04 - 0.95±0.07
52.7	1	123±8	0.87±0.06
35.2	2	137±10 - 187±13	0.92±0.06 - 1.2±0.09
29.9	1	138±10	1.2±0.08

[1] Unit equipped with fan speed control.

3.1 Impact of increased pressure drop on rooftop unit performance

The two sections below detail the field measurements and modeling results of the impact of increased pressure drop on units without and with fan speed control.

3.1.1 Units without fan speed control

For systems with no fan speed control, increasing the pressure drop across filters led to system curve changes in a way that induced flow decrease and fan static pressure increase, as shown in Figure 2. Figure 2 displays measured fan pressure, power, efficiency, and theoretically derived system curves (assuming p=constant×Q^2, where p is the fan pressure rise and Q is the fan airflow rate) for a 70.3 kW nominal capacity unit. The same trend is observed for residential and light-commercial systems in the absence of any fan speed control (Stephens et al., 2010[a]).

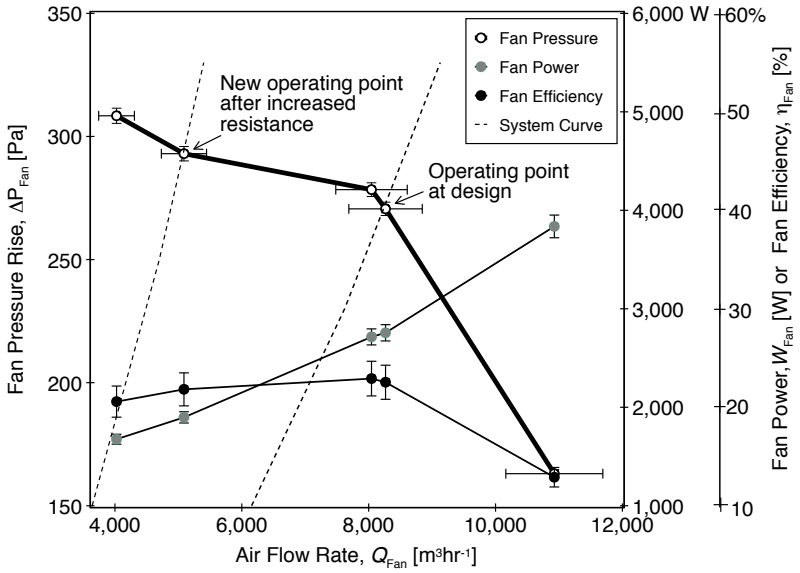

Figure 2. Measured fan and two theoretical system curves (assuming p = constant × Q^2) for a 70.3 kW nominal capacity unit.

As a result of the flow decrease, the fan power, cooling capacity, and compressor power also decreased. Figure 3 displays an example of measured filter and coil pressure drop versus flow and measured power versus flow for three different size units. This figure shows that the relationship of filter pressure drop/fan power was approximately linear with the change in airflow rate. The linear relationship between these parameters was used later in the paper to project these findings on different MERV filters.

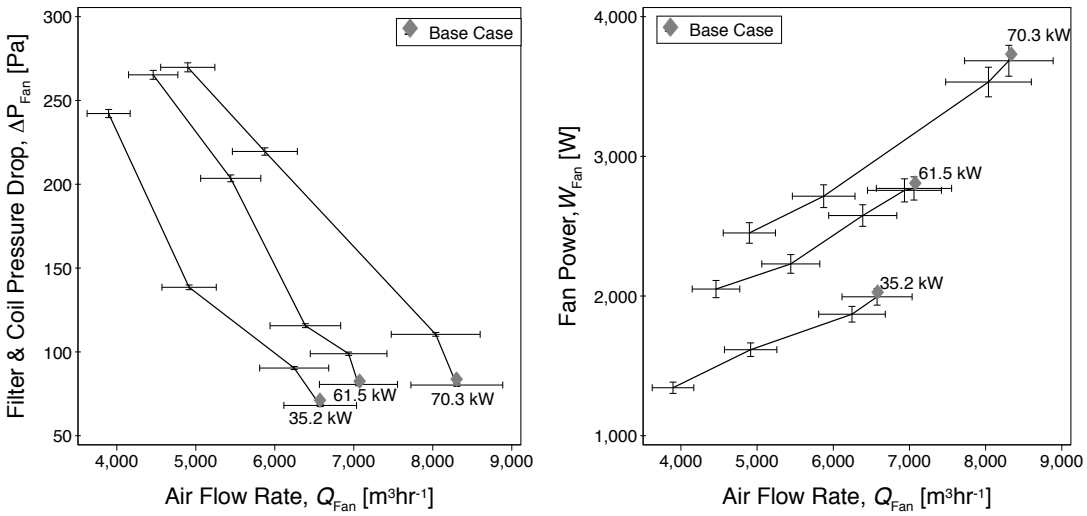

Figure 3. Measured filter and coil pressure drop (left) and measured fan power (right) versus fan airflow rate for three units tested without fan speed control (70.3, 61.5, and 35.2 kW). The initial base case denotes the normal operating condition of the RTU unit tested (i.e. filters installed and no blockage is performed) and is marked as a diamond in gray on the graph.

Figure 4 displays an example of modeled cooling capacity/compressor power (using the ACHP vapor compression system model) versus flow. The example shows that the relationship of cooling capacity/compressor power and flow is essentially non-linear over the whole range of flow. However, it could be approximated as a linear function for the range of flow considered in this work, which is the area between the vertical dashed lines.

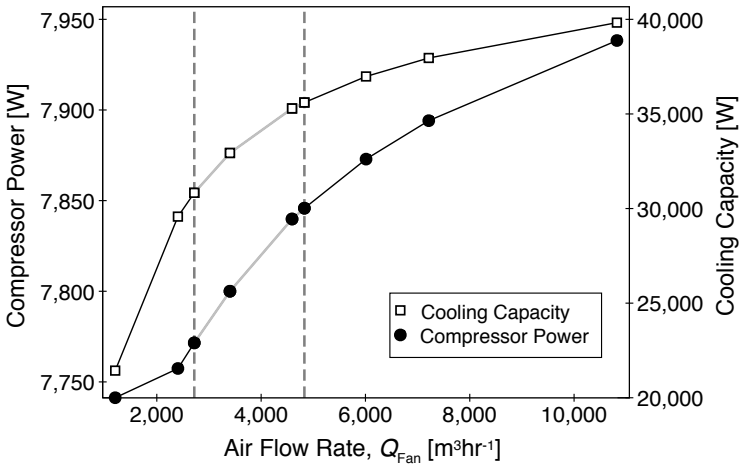

Figure 4. Modeled compressor power and cooling capacity versus fan airflow rate for a 35.2 kW nominal capacity unit with no fan speed control. The area between the dashed vertical lines represents the range of flow considered for this unit.

To show the results on all the units tested, the distribution of the impact of increasing the pressure drop across filters is shown in Figure 5. Figure 5 contains the range of flow, power, and EER change measured for different filter pressure drops normalized to baseline conditions (baseline conditions represents the normal operating condition of the RTU unit tested). The bottom of the box indicates the 25th percentile; the horizontal line indicates the median, and the top of the box represents the 75th percentile. Whiskers indicate the data ranging within 1.5 times the interquartile range of the 25th and 75th percentile. Open circles are outliers.

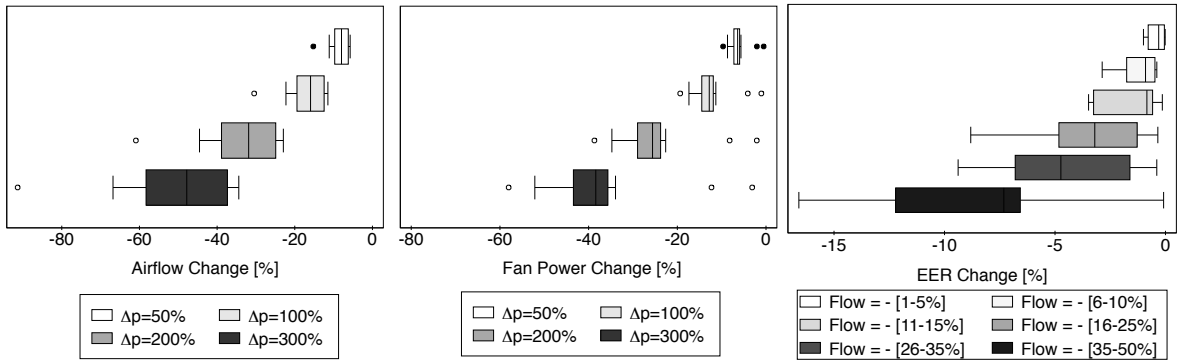

Figure 5. Median and range of fan airflow (left), fan power (middle), and EER (right) change subject to different filter pressure drop normalized to baseline conditions across all units.

According to filters manufacturers, the recommended final filter pressure drop ranges from approximately two to five times that of the initial pressure drop. Following this recommendation, doubling the pressure drop will incur a median airflow decrease of 15.9% and median fan power decrease of 12.8%, as shown in Figure 5. For a five times increase in filter pressure drop, the flow and fan power will decrease at least 47% and 38%, respectively. The reason for this significant power decrease is the larger impact of flow decrease compared to the impact of pressure rise while the fan efficiency increases only marginally (from the fan power equation: $W = \Delta p \times Q/\eta$). Table 2 shows a comparison of the effect of filter pressure drop on airflow and fan power measured in this study with findings from measurements in residences/light commercial buildings (buildings with cooling capacity smaller than 18 kW) during cooling mode for units equipped with no fan speed control (Stephen et al., 2010[b]; Walker et al., 2013). It should be noted that two of the units tested by Stephens et al. (2010[b]) had electronically commutated motor (ECM) fans; nonetheless these units did not maintain the flow constant when subject to a higher filter pressure drop.

Table 2. Comparison of the effect of pressure drop with measurements performed in residences and light commercial buildings during cooling mode.

Pressure Drop Increase	Airflow Change [%]	Fan Power Change [%]	Study	n
1.3-1.4	-8	-4	Walker et al. (2013)	2
1.4	-10	-4	Stephens et al. (2010[b])	16
1.5	-8	-6	This study	14
1.8-2.3	-36	-19	Walker et al. (2013)	3
2	-8.6	-4.7	Stephens et al. (2010[b])	15
2	-15.9	-12.8	This study	14

Table 2 shows that the measurements conducted in residences agree with the findings of this study for 1.3 - 1.5 times increase in pressure drop across filters. However, for 1.8 - 2.3 increase in pressure drop, the findings of this study lay between the two studies referenced from the literature. Discrepancies between these studies could be due to the shape of the fan efficiency and power curves for different equipment, as well as the location of the working point.

Figure 5 also displays the median and range of the modeled energy efficiency ratio (EER). EER serve to compare the net decrease in cooling capacity (difference between modeled cooling capacity and measured evaporator-side fan power) relative to the decrease in total power (measured fan power, modeled compressor and condenser power) when subject to drop in airflow rate. Figure 5 shows that decreasing the flow as much as 15% (i.e. doubling the pressure drop) minimally impacted EER (< 1.4% decrease), and decreasing the flow 35% to 50% (i.e. at least tripling the pressure drop) caused a median EER decrease of 7.3%. The reduced refrigerant flow rate decreases the power consumption of the compressor; nonetheless, the net effect of the EER is lower because

the decrease in cooling capacity is greater than the decrease in compressor power. The minimal variance of EER in response to decreasing airflow is anticipated because the tested units are all equipped with a thermostatic expansion valve (TXV). TXVs maintain the refrigerant vapor at the exit of the evaporator at a relatively constant evaporation pressure (instead of a lower evaporation pressure). For example, Rodriguez et al. (1996) compared two air-conditioner (AC) units, one equipped with a TXV valve and one with a short tube orifice. The AC unit equipped with the orifice caused seven times more decrease in EER compared to the unit equipped with TXV for a 20% or less reduction in airflow and two times more decrease in EER for a flow decrease between 20 and 30%.

Another consequence of decreasing airflow rate is the decrease in sensible heat ratio (SHR), which can be favorable for hot and humid climates. For example, lower airflow cause a longer residence time in the cooling coil, which increases dehumidification. The reduced airflow also decreases the convective heat transfer coefficient (i.e., sensible heat transfer between coil and air), enabling a lower coil temperature and leading to higher moisture removal. Decreasing the flow as much as 15% (doubling the pressure drop) caused the SHR to decrease 2.7%, and decreasing the flow 35% to 50% caused a median decrease of SHR of 5.6%.

The primary analysis of units without fan speed control shows that the increase of the pressure drop across filters produce some energy advantages (decrease in fan power) and disadvantages (decrease in EER) in the air handling unit operation. Thus, during fan-only mode, the filter pressure drop leads to energy savings. During cooling mode, the overall energy consequence of the filter pressure drop is more complex: in addition to fan power

reductions, the cooling capacity decreases, the compressor power decreases, and at the same time the duty cycle increases to maintain the same cooling capacity. Detailed calculations on the impact of high efficiency filters on energy during cooling and fan-only mode is presented in Section 3.2.

3.1.2 Unit with fan speed control

For systems with fan speed control, increasing the pressure drop across filters yields fan curve changes that maintain the flow constant, as shown in Figure 6. Figure 6 displays measured fan pressure, power, speed, and theoretically derived system curves for a 84.4 kW nominal capacity unit.

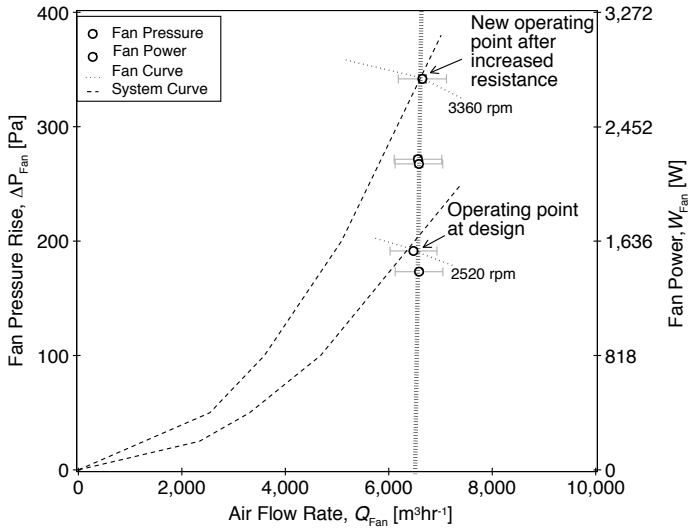

Figure 6. Measured fan operating points and two theoretical system curves (assuming p = constant × Q^2) for a 84.4 kW nominal capacity unit.

As a result of pressure drop increase, the speed and the power of the fan increase. Figure 7 shows the measured filter and coil pressure drop and the fan power versus the fan speed. As shown in Figure 7, changes in the fan speed and the fan power can be approximated as a linear function with the increase in pressure drop.

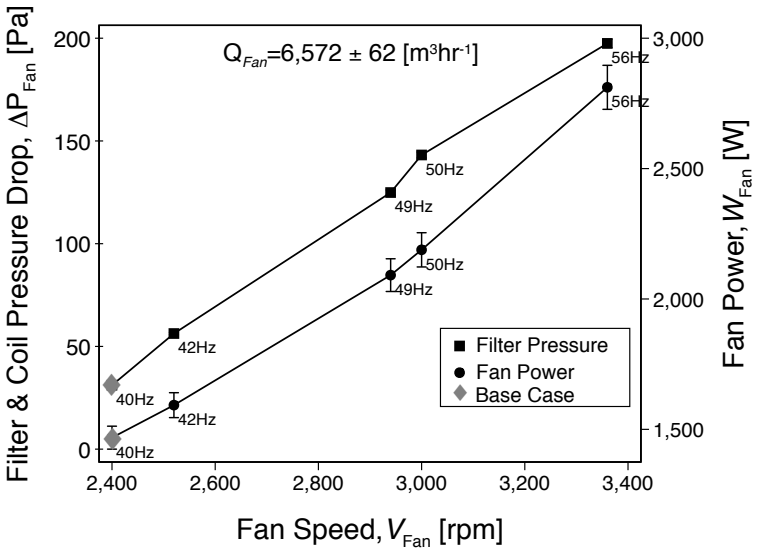

Figure 7. Filter and coil pressure drop (left) and fan power (right) versus fan speed for one unit tested with fan speed control (84.4 kW). The initial base case denotes the normal operating condition of the RTU unit tested (i.e. filters installed and no blockage is performed) and is marked as a diamond in gray on the graph.

For the tested unit, doubling the pressure across the filter caused 12.5% increase in fan power, and quintupling the pressure drop across the filter caused 62.5% increase in fan power. Walker et al. (2013) tested the effect of doubling the pressure drop (1.9-2.0 times increase in pressure drop) on one unit equipped with fan speed control installed in a residence. Their study found higher increase in fan power (22%) than the current study, a difference that is likely attributed to the difference between fan efficiency and power curves for different fans. The increase in power is caused by the increase in fan pressure and constant flow and fan efficiency (based on the fan power equation). Because the flow remained constant, the cooling capacity, compressor power, and duty cycle also remained approximately constant. The magnitude of the effect of the increased fan power translated into additional heat to the air is considered to be small and therefore it was not accounted for in these results.

Across all measured units (i.e., with and without fan speed control), the median calculated fan efficiency was 23% with a maximum calculated fan efficiency of 36%. This finding suggests considerably lower fan efficiency than has been assumed elsewhere for commercial buildings in modeling studies (e.g., Fisk et al., 2002; Bekö et al., 2008) and also that potential energy savings could be realized by improving fan efficiency.

3.2 Impact of higher efficiency filters on energy and indoor air quality

The goal of this analysis was to extend the energy results to investigate how filter efficiency influences indoor particle concentrations. Before showing the results, it is important to discuss the relationship between the $PM_{2.5}$ and PM_{10} efficiency and MERV category, as well as the relationship between filter pressure drop and MERV category.

3.2.1 Relationship of $PM_{2.5}$ and PM_{10} efficiency with MERV category

Figure 8 shows the fractional efficiency obtained from filter manufacturers and Rivers and Murphy (1999) datasets for particle size between 0.3 to 1 μm, integrated $PM_{2.5}$ and PM_{10} efficiency function of MERV filter category. PM efficiency was calculated as described in the Method Section 2.2, Equation 1. According to the results in Figure 8, replacing MERV 8 by MERV 11 may not be necessarily advantageous to decrease $PM_{2.5}$ particle concentrations, specifically for MERV< 13. However, this statement is not true for $PM_{2.5}$ efficiency if we replace MERV 8 with MERV 13 or 14. For PM_{10} particles, PM_{10} efficiency was observed to increase with MERV category for most of the filters.

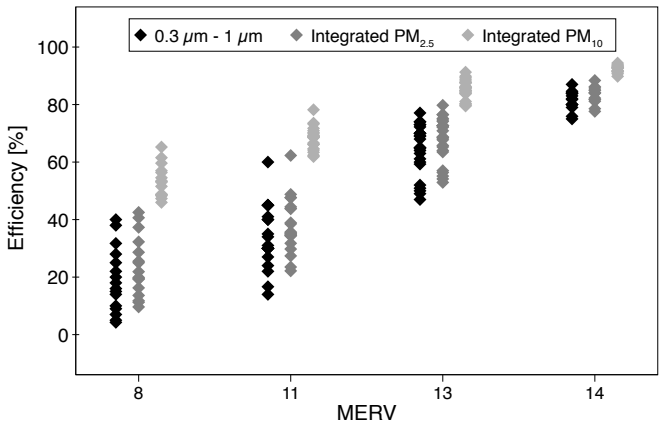

Figure 8. Fractional efficiency for particle size between 0.3 μm – 1 μm, integrated $PM_{2.5}$ and PM_{10} efficiency versus MERV filter categories

3.2.2 Relationship of filter initial pressure drop with MERV category

Another observation is the correlation between filter pressure drop and MERV at a fixed flow rate, as shown in Figure 9. For this analysis, characteristics of 90 filters (30 filters tested by Rivers and Murphy (1999), and the rest from different filter manufacturers) were compared with MERV category ranging from 7 to 14. Filter pressure drop was calculated using a second-degree polynomial fit for each filter that links pressure drop and the face velocity of the RTU, as detailed in the Method Section 2.2. As shown in the figure and demonstrated by regression analysis, there was a weak correlation between filter pressure drop and MERV filter category ($p<0.01$ and $R^2=0.2792$; n=90). These results agree with the experimental findings of Springer (2009) and Stephens et al. (2010[b]) that show a wide variety of pressure drops owing to differences in filter design. This suggests that MERV category by itself cannot be used as a metric for $PM_{2.5}$ filter efficiency and pressure drop (i.e., energy consumption), especially for MERV categories less than 12.

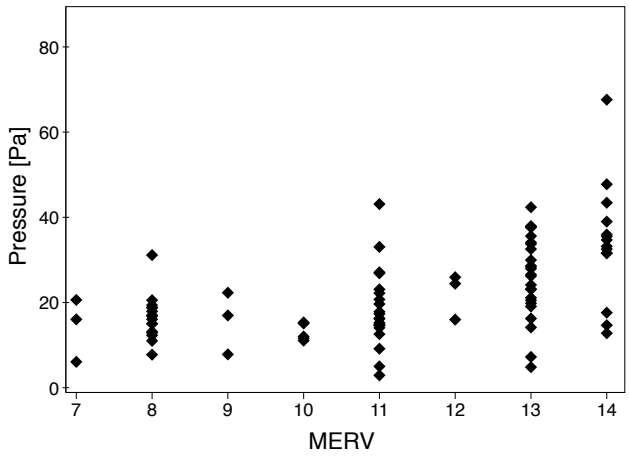

Figure 9. Filter initial pressure drop versus MERV filter categories

3.2.3 Impact of replacing a MERV 8 filter with a higher MERV filter on energy and IAQ

The impact of replacing a MERV 8 filter with a higher MERV filter on power consumption was calculated for cooling mode by incorporating the fan power, compressor power, and duty cycle. For fan-only mode, analysis considers only the fan power (e.g. economizer or ventilation modes). The impact on particle concentrations was investigated using the CADR for two PM sizes: $PM_{2.5}$ and PM_{10}. Figure 10 summarizes these findings for units equipped with no fan speed control and Figure 11 for units equipped with fan speed control. There are three columns in each of these two figures; each column corresponds to replacing MERV 8 with a higher MERV: MERV 11 (left), MERV 13 (middle), and MERV 14 (right). The distribution represents the range of impact for different filters with the same MERV category. The number of filters for each MERV is indicated at the top of each column. There are 5 distributions in each sub-figure that correspond to the impact on energy and air quality relative to 5 base case MERV 8 scenarios.

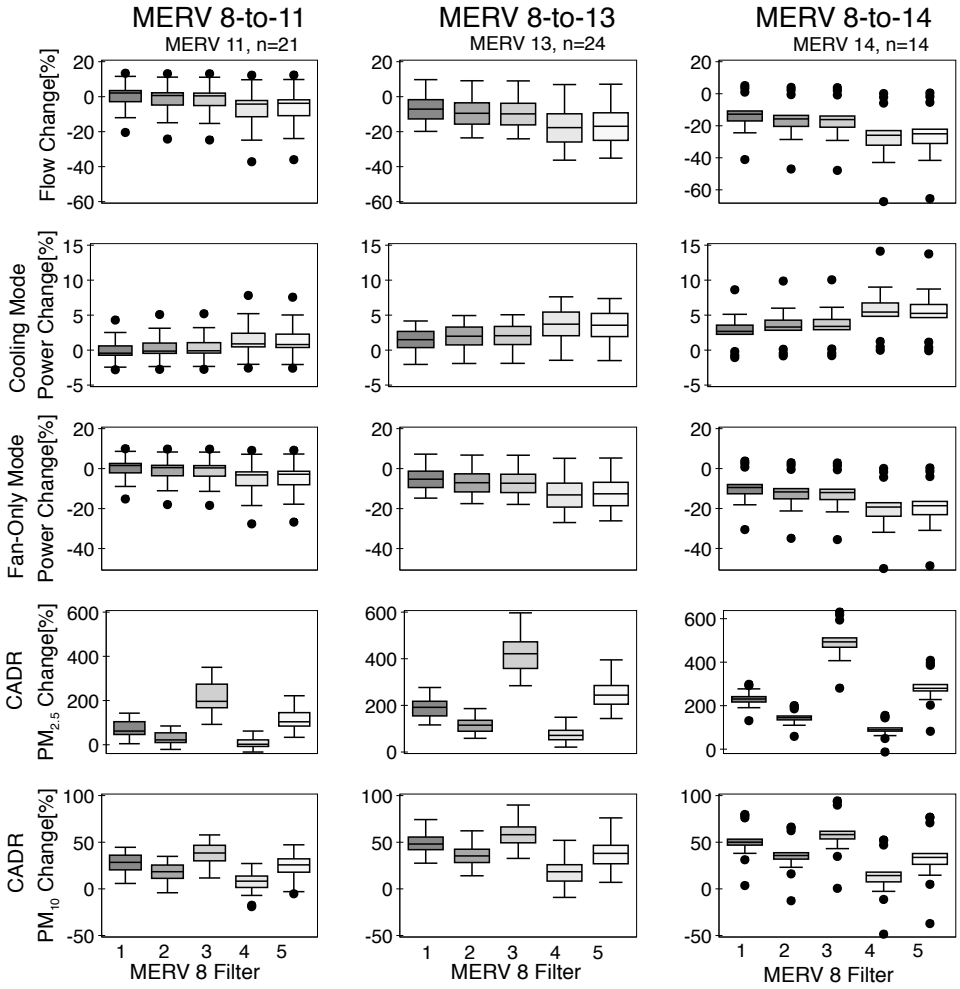

Figure 10. Box plots showing the impact of replacing MERV 8 with MERV 11 (left), 13 (middle), 14 (right) on flow, total power (compressor and fan), fan power, CADR $PM_{2.5}$ and CADR PM_{10} for five MERV 8 base cases for face velocity of 0.9 ms^{-1} for units equipped with no fan speed control.

Results in Figure 10 show that for units equipped with no fan speed control, replacing MERV 8 with MERV 11 marginally decreased the airflow; across all the MERV8 base cases, the median decrease in airflow was 1.9%. The maximum median airflow decrease of 4.2% was observed for MERV 8 - base case number 4. This was anticipated because the MERV 8 filter corresponding to this base case had the lowest pressure drop (Figure 1). As a result of the marginal flow decrease, the power during cooling mode increased

by 0.4% and during fan-only mode decreased by 1.4%. This indicates that the energy consequences of MERV 11 compared to MERV 8 filter are minimal. As for $PM_{2.5}$ CADR, median CADR increased by 63%. The increase relative to base case 4 was the lowest (median 2.2%) and the increase of CADR relative to base case number 3 was the greatest (median 214%). This could be explained by the fact that MERV 8 - base case number 4 had a relatively high (the largest) $PM_{2.5}$ efficiency and MERV 8 - base case number 3 had a very low $PM_{2.5}$ efficiency (Figure 1).

As for PM_{10}, median CADR increased by 23%. For some MERV 11 filters (5 out of 21 filters), $PM_{2.5}$ efficiency was lower than that of MERV 8, causing the CADR to decrease (example observed in Figure 10 for MERV 8 base case number 2 and 4). In addition, CADR was also decreased when the diminished flow was not compensated for by the increased efficiency (4 out of 21 filters for PM_{10}, and 3 out of 21 filters for $PM_{2.5}$). In summary, MERV 11 offered a moderate increase in CADR for most of the filters for minimal change in power consumption when compared to MERV 8 filter.

Overall, replacing MERV 8 with MERV 13 and MERV 14 filters had a larger effect on the studied parameters than those of MERV 11. The median decrease in flow was 10.8% when using MERV 13 and 17.6% when using MERV 14. For cooling mode, the increased duty cycle to meet the same cooling load requirements caused a median total power draw increase of 2.4% for MERV 13 and 3.7% for MERV 14. For fan-only mode, the fan power decreased 8.4% for MERV 13, and 13% for MERV 14, suggesting a potential benefit when switching from a low to a higher-pressure drop filter. The total energy impact of high efficiency filters (i.e., MERV 13, MERV 14) depends on the time

the system runs in cooling mode and fan-only mode. For example, if the system was running 100% in recirculation or economizer modes (i.e. fan-only with no cooling), then replacing a low efficiency filter with a higher efficiency filter may result in energy savings, especially given that fan energy contributes to at least 7% of total site energy consumption, as previously mentioned. As another example, if the system was running in 25% economizer mode, then replacing MERV 8 filter by MERV 14 may result in energy neutral consumption. Besides the energy consequences, replacing MERV 8 with high MERV filters offered largely better indoor air quality (at least 2.9 times increase in $PM_{2.5}$ CADR).

Comparing MERV 13 with MERV 14 filter reveals that the median $PM_{2.5}$ CADR that corresponds with MERV14 was only 1.2 times higher than that of MERV 13, and the median PM_{10} CADR for both MERV 13 and MERV 14 was comparable. The increased CADR for MERV 14 filter was accompanied with 1.5% increase in power consumption during cooling mode and similar percentage decrease in fan-only mode.

As for units with fan speed control, replacing MERV 8 with a higher MERV caused an increase in fan power to maintain a constant flow, as mentioned previously. In contrast to units without fan speed control, there was no saving in fan power; the fan power draw increased in both cooling mode and fan-only mode, as shown in Figure 11. For MERV 11, the fan power increased 2%. For MERV 13 and MERV 14, the increase in fan power was more important: 11% for MERV 13 and 18% for MERV 14. Accounting for the change in total power during cooling mode, the increase in power draw for units with and without fan speed control was comparable. However, during fan-only mode, using high

MERV filters incurred additional fan cost when using units with fan speed control and incurred energy savings when using a fan without speed control.

Since the flow in the units equipped with fan speed control remained the same, the CADR improvement was simply an efficiency improvement: $PM_{2.5}$ CADR increased 1.7 times for MERV 11, 3.1 times with MERV 13, and 3.8 times with MERV 14. The increase in PM_{10} CADR was lower: 1.3 times with MERV 11, 1.6 times with MERV 13, and 1.7 times with MERV 14. The constant flow for these units caused the CADR to be greater than the CADR in the units without fan speed control.

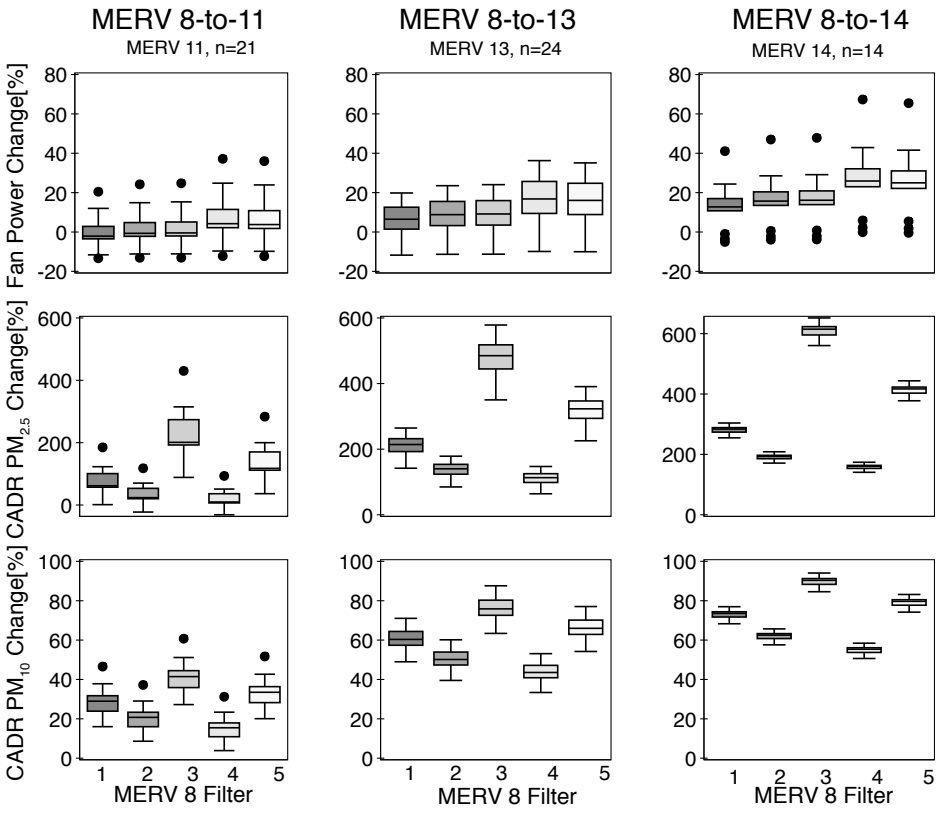

Figure 11. Box plots showing the impact of replacing MERV 8 with MERV 11 (left), 13 (middle), 14 (right) on fan power, CADR $PM_{2.5}$ and CADR PM_{10} for five MERV8 base cases for face velocity of 0.9 ms^{-1} for unit equipped with fan speed control.

Comparing these results with published studies in the literature, there are two studies that performed field measurements in residences equipped with small systems to test the impact of high MERV filters on energy use (Stephens et al. (2010[b]): 15 units without fan speed control and 2 units with fan speed control; Walker et al. (2013) , which also included simulation work: 7 units without fan speed control and 3 units with fan speed control). Both studies found minimal effect of mid/high MERV (range mid MERV: 6-8; range high MERV: 11-13) on fan power (<5%) when replacing low MERV filters (range low MERV 2-5). Additionally, Walker et al. [12] found that fan power draw became significant when MERV16 was used (about 20%). Also, Stephens et al. (2010[b]) found minimal energy reduction (0.07 kWh kWcapacity^{-1} day^{-1}) following the switch to high MERV, although the uncertainty was large (±1.3 kWh kWcapacity^{-1} day^{-1}). Direct comparison of findings form these two previous studies with the results presented in this paper is challenging due to the differences in the filters considered and the equipment. Also, other factors that are reasoned by these studies may affect the results (e.g., duct leakage, initial airflow resistance, capacity rating of the system, etc.). Yet the general consensus in the literature (including this work) is that using a mid MERV filter (up to MERV 11) has minimal impact on energy consumption in cooling or fan-only mode; this is for both units equipped with or without fan speed control.

3.3 Effect of filter loading

The work described above compared the energy and air quality performance of filters while taking into account the initial filter pressure resistances (i.e., clean conditions). In this section, artificial fouled filter data were used to verify if the conclusion for higher

MERV filter reached for initial (clean conditions) still holds at the end of life of the filter. Because the results from artificial fouling tests differ from those for filters subjected to natural loading, the results presented herein serve as estimates. In this analysis, we neglect filter bypass (e.g., Ward and Siegel, 2005; VerShaw et al., 2009) as the purpose was to compare different filters, although bypass could influence both energy and indoor air quality findings.

Furthermore, the analysis presented below uses data of filter loading conducted on uncharged media filters: the experimental data (efficiency and pressure drop) for artificial filter fouling are available from Rivers and Murphy (1999). For these filters, loading increases filter pressure drop and filter efficiency. For example, Rivers and Murphy (1999) reported that that MERV 13 and 14 filters increased at least to MERV 15 after being fully loaded. Also, this study tested two MERV 8 and two MERV 11 filters and found that these filters increased to MERV 12 and MERV 13/15, respectively. Thus, uncharged media filters present air quality advantage in terms of fractional efficiency (not considering the clean air delivery yet, i.e. the change in airflow rate) and increase in energy consumption because pressure drops increases with dust loading. The question remains if the increase in energy consumption is offset by higher clean air delivery rate: the answer to this question is presented in the next paragraph. The other broad type of filter classification is the charged media filter. Contrary to uncharged media filters, research shows that charged media filters exhibit significant efficiency reductions as filter is used, especially for particles less than 1.5 μm (e.g. Barrett and Rousseau, 1998; Raynor and Chae, 2001). Also, these studies showed that charged media filters had very

little change in pressure drop after loading. The question for this type of filters remains whether the decrease in fractional efficiency for MERV 13/14 after loading is still higher than the MERV 8 efficiency. We know of no available data to use for this analysis and therefore we only consider uncharged media filters.

To compare the impact of pressure drop on energy and air quality for clean and fouled mechanical filter conditions, energy and CADR calculations were repeated for two filter categories: MERV 8 and MERV 13 for clean and fouled conditions. Figure 12 summarizes the results from this calculation for units equipped with no fan speed control and a face velocity of 0.9 ms^{-1}. The results are provided for one MERV 8 base case and seven different MERV 13 filters.

Results show that the pressure drop of MERV 13 filters relative to the MERV 8 filter base case when filters were fully loaded was lower than when the filters were clean; nonetheless the pressure drop of MERV 13 was higher than that of MERV 8 for both clean and fouled conditions (Figure 12). This caused the total power to increase slightly (1.15 times) and the fan power savings to decrease (3.7 times) for fouled conditions when comparing the replacement of MERV 8 with MERV 13 for clean conditions. For fouled conditions, CADR improvement decreased for both PM sizes as both MERV categories approached high filtration efficiency. Nevertheless, MERV 13 still offered better air quality than MERV 8 (CADR change is positive) even at fully loaded conditions. In conclusion, at fully loaded conditions replacing MERV 8 with higher MERV filter efficiency such as MERV 13 in units with no fan speed control: 1) offers air quality advantages, but less than that observed at clean conditions; 2) causes higher energy

consumption than that observed at clean conditions.

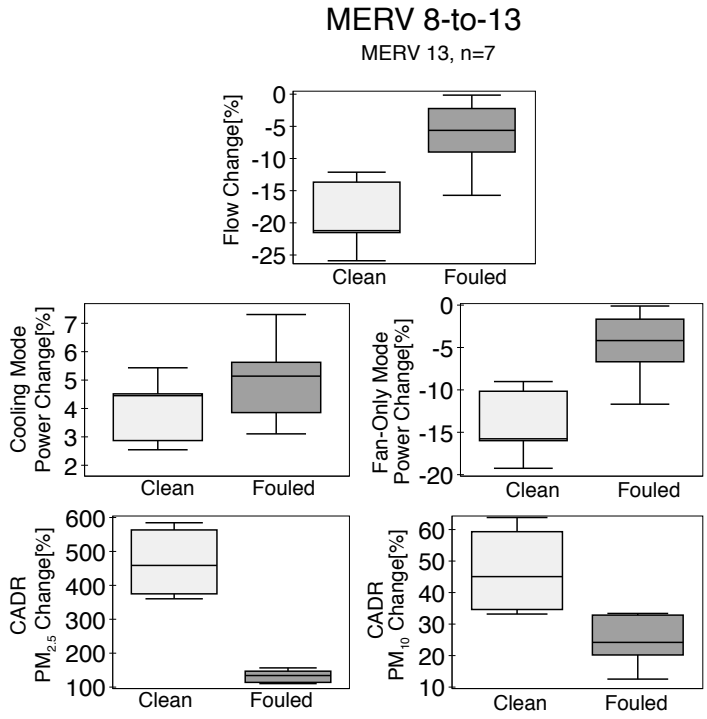

Figure 12. Box plots showing the impact of replacing MERV 8 with MERV 13 on fan airflow rate, fan power, CADR $PM_{2.5}$ and CADR PM_{10} for face velocity of 0.9 ms^{-1} for units without fan speed control.

As for units with fan speed control, fan power draw of MERV 13 relative to MERV 8 decreased (3.6 times) with fouling when compared to clean conditions; the absolute value of fan power draw for MERV 13 remained larger than that of MERV 8. CADR improvement for both PM sizes also decreased but remained positive. This suggests that at fouled conditions, higher efficiency filters installed in units with fan speed control still offer advantages related to air quality, with less energy consumption that that observed at clean conditions.

4. CONCLUSION

The goal of this study was to establish the overall energy and IAQ implications of

varying degrees of filter efficiencies in commercial systems. The following significant conclusions were obtained from the field measurements and data analysis:

(1) Commercial rooftop units equipped with no fan speed control generally behave like residential systems when subjected to a decrease in airflow. Doubling the pressure across filters resulted in a median airflow decrease of 15.9%, a median fan power decrease of 12.8%, and a minimal impact on EER. For units with fan speed control, the same increase in pressure drop resulted in the same magnitude change of fan power but in the opposite direction, and other parameters were unchanged.

(2) Replacing MERV 8 with MERV 13 or MERV 14 in units without fan speed control can result in energy savings, depending on the operation of the RTU. However, this is not the case for units with fan speed control, fan power draw increased during all modes (11-18%). Accounting for the change in total power during cooling mode, the increase in power draw for units with and without fan speed control is comparable.

(3) The increase in energy consumption with the use of high MERV filter is offset by improvement in CADR, especially for $PM_{2.5}$. The air quality benefits are larger for units equipped with fan speed control because the flow is maintained constant.

(4) Comparing MERV 13 with MERV 14 reveals that the median $PM_{2.5}$ CADR that corresponds with MERV 14 is only 1.2 times higher than that of MERV 13, and the median PM_{10} CADR for both MERV 13 and MERV 14 is comparable. The increase in CADR is accompanied with 1.5-1.6% increase in power consumption

during cooling mode and similar percentage decrease in fan-only mode for units with no fan speed control. For units with fan speed control, the increase in CADR was accompanied with a similar percentage increase in power draw for both cooling and fan-only mode.

(5) The differences in energy and air quality impact of high MERV filters were much smaller at fouled conditions; nonetheless, high MERV filters still offered advantages related to improved air quality. As for energy consumption, at fouled conditions high efficiency filters caused higher energy consumption for units without fan speed control (1.15 times increase in total power, and 3.7 times decrease in energy saving during fan-only mode), and lower energy consumption for units equipped with fan speed control (3.6 times decrease in fan power).

These results suggest the importance of understanding the energy and air quality impact of high efficiency filters. Deciding on the right filter efficiency is crucial for achieving acceptable indoor particulate matter concentration and low energy use.

5. ACKNOWLEDGMENTS

We would like to extend thanks to the ASHRAE RP-1596 project team for their assistance with field measurements, the stores that allowed the measurements to take place, and professors James Braun and Nabil Nassif for providing the vapor compression models.

Appendix D

Paper 3: Impact of ventilation and filtration strategies on energy consumption and exposures in retail stores

Marwa Zaatari, Atila Novoselac, Jeffrey Siegel

ABSTRACT

The impacts of different ventilation strategies on exposure to, and risks from, contaminants of concern (COCs) and energy usage in retail buildings are not sufficiently investigated. We used a multi-contaminant model of an area-normalized retail store, and developed best estimates for distributions of model inputs across the retail sector. We used these distributions in a six city Monte Carlo simulation to compare the impacts of the ASHRAE 62.1-2010 ventilation rate procedure (VRP), demand controlled ventilation (DCV), and indoor air quality procedure (IAQP), with or without using a high particulate efficiency filter. Results showed that for cities where outdoor $PM_{2.5}$ concentration is low, adopting the IAQP with low efficiency $PM_{2.5}$ filter in grocery stores and the VRP with high $PM_{2.5}$ efficiency in non-grocery stores yielded the highest health benefits. For cities with high outdoor $PM_{2.5}$ concentration, adopting the VRP with high $PM_{2.5}$ efficiency for both store types yielded the highest health benefits. Nonetheless, these exposure benefits translated into an increase in energy consumption, with the magnitude depending on the city and the retail type. We propose a new pollutant exposure control ventilation (PECV) strategy, where ventilation rates are weighed against exposure to

contaminants of concern, next a range of acceptable exposures is identified, then the ventilation rate that is most climatically advantageous is chosen. Selecting an appropriate ventilation strategy is key to achieving energy efficiency and acceptable air quality in retail buildings.

1. INTRODUCTION

Ventilation is mainly used to promote the comfort of occupants by diluting emissions of indoor-generated pollutants. The measurable benefits of increased ventilation rates are the decreased sick building syndromes and the improved perceived air quality, leading to major economical benefits (e.g., better productivity, positive impact on retail sales; Wargocki et al., 2002; Chebat and Michon, 2003; Seppänen et al. 2004; Sundell et al., 2011). However, ventilation may also play a negative role with respect to indoor air quality as it can transport ambient pollution indoors (e.g., Hänninen et al., 2005). Beside its impact on air quality, ventilation is closely tied to energy consumption: the use of ventilation in the existing building stock contributes 6.6% to the total energy use index across the sector (i.e., building's energy use as a function of its size; Benne et al., 2009). Thus, the goal is to balance ventilation energy cost and air quality without sacrificing the exposure and the economical benefits.

Over the past two decades, a lot of effort has been exerted to find the minimum ventilation rates that will reduce energy consumption while maintaining an acceptable indoor air quality, most commonly adopted are the ventilation rates specified by ASHRAE Standard 62.1-2010. ASHRAE Standard 62.1-2010 provides two alternative procedures for selecting the minimum ventilation rate for commercial buildings: 1) a

prescriptive approach: the ventilation rate procedure (VRP); 2) a performance-based approach: the indoor air quality procedure (IAQP).

The VRP is the more widely used procedure. The prescribed minimum ventilation rates are the sum of two quantities: the minimum rate of outdoor air supply per unit floor area, and the minimum rate of outdoor air supply per occupant. The VRP is assumed to maintain an acceptable indoor air quality as perceived by at least 80% of occupants. Bluyssen et al. (1996) tested 44 buildings with average ventilation rates of 25 L/s (far above the current ventilation rates specified by the VRP) and found that air quality in 64% of these buildings did not satisfy 80% of the occupants. In addition, a review of ventilation measurements in retail stores found that half of the stores tested met or exceeded the VRP; nonetheless, these ventilation rates were not sufficient to keep all pollutants below their most conservative limits (Zaatari et al., 2013). Specifically, there is no documentation of the adequacy of VRP in maintaining an acceptable indoor air quality in retail buildings.

One variation to the VRP is the use of demand control ventilation (DCV), often based on CO_2 concentrations in buildings with variable occupancy, to further save energy. The impact of DCV on indoor air quality remains less investigated. Only eight literature studies investigated whether controlling ventilation by measuring occupancy (DCV-based CO_2) could keep pollutants (e.g., formaldehyde, TVOC, radon) below their reference or regulatory limits (Gabel et al., 1986; Donnini et al., 1991; Carpenter, 1996; Enermodal, 1995; Persily et al., 2003; Chao & Hu, 2004; Jeong et al., 2010; Mui & Chan, 2006; Herberger & Ulmer 2012). Five of these studies found that DCV was not sufficient

to control the measured pollutants below their established limits; additional studies would have found that DCV did not control these pollutants if they considered the current reference exposure or standard limits for these pollutants. DCV-based CO_2 does not control outdoor-generated pollutants, nor does it account for pollutants generated indoors but independently of human activities.

Another approach to control ventilation is to follow the performance-based approach, the IAQP, specified in ASHRAE Standard 62.1-2010. In the IAQP, the minimum ventilation rate is defined to be the larger rate resulting from an objective assessment, based on pollutants' emission rates and concentration limits, and a subjective assessment of air quality. In the objective assessment, the IAQP requires the engineer to choose the larger ventilation rate required to keep each individual pollutant below its established limit. Thereby ignoring the fact that some pollutants (e.g. ozone, and some particles) can be generated outdoors and keeping the ventilation rate to a minimum may be more advantageous from both exposure and energy perspectives. Further, generally, there is a lack of knowledge of source strengths (used in the IAQP to calculate the required ventilation rate) in different types of buildings and poor understanding of how different sources of emissions should be added together.

The impacts of VRP, DCV-based CO_2, and IAQP on energy usage and exposure to contaminants of concern whether generated indoors, outdoors, or both are not sufficiently investigated, specifically for retail stores and their understudied energy, health, and economic influence. The main objective of this paper is to determine an exposure-based, energy-efficient control strategy for different retail types and locations.

Specifically this paper answers the following questions:

1. What are the effects of ventilation rates determined by VRP, DCV-based Co_2, and IAQP on contaminants of concern (COCs) found in retail buildings?
2. What happens to COCs concentrations if we increase the particle filter efficiency?
3. What is the optimal ventilation-filtration combination strategy that will lead to a balance between exposure to pollutants and energy consumption?

This work is useful for building designers and other researchers in understanding the impact of different ventilation strategies recommended by energy standards on indoor air quality and HVAC energy use in retail buildings. Additionally, this paper proposes a new ventilation strategy tailored for different retail types and locations that reduces energy consumption without compromising occupants' health, as well as minimizing negative sales impacts from poor perceived air quality.

2. SIMULATION METHODOLOGY

2.1. Overview

The methodology comprises four steps, (1) identifying contaminants of concern, (2) assessing the impact of control strategies on COCs concentrations, and quantifying (3) exposures and (4) energy consumption. Each step is summarized below.

2.1.1. Contaminants of concern in retail buildings

Zaatari et al. (2013) identified contaminants of concern in retail buildings by using data compilation from 15 literature studies (150 stores, a total of 34 pollutants), and found that $PM_{2.5}$ and acrolein are the main contaminants of concern for which control methods

should be prioritized. In the present paper, we used these identified contaminants of concern as well as two additional pollutants, formaldehyde and acetaldehyde, for further assurance that the selected control strategy will not increase concentrations of these pollutants to a level where they will considered as COCs.

2.1.2. Impact of control strategies on COCs concentrations

Two alternative exposure control scenarios were evaluated using a time-averaged mass balance multi-contaminant model. This model was applied on a well-mixed single-zone area-normalized retail store. The first control scenario calculates $PM_{2.5}$, acrolein, formaldehyde, and acetaldehyde concentrations based on different ventilation strategies. The second scenario complements the first scenario with increase of $PM_{2.5}$ filtration level. Best estimates for distributions of inputs across the retail sector were modeled in Monte Carlo simulations for multiple combinations of cities (6 major cities were chosen to cover different climates as well as different outdoor air quality, Austin, Philadelphia, Minneapolis, Seattle, Los Angeles, Phoenix), seasons (winter and summer), store types (grocery and non-grocery), and time of the day (store-open and store-closed).

The time-average pollutant mass-balance model is described in Equation 1, based on that provided by Riley et al. (2002) and adding an indoor source emission term.

$$C_{out}\left(\frac{Q_{OA}(1-\eta)}{V}+\frac{pQ_i}{V}\right)+E-C_{in}\left(\frac{Q_R\eta}{V}+\beta+\frac{Q_{EX}}{V}\right)=0 \quad (4)$$

Where C_{out} is the outdoor and indoor concentrations [$\mu g/m^3$], Q_{OA} is the mechanical outdoor air airflow rate [m^3/h], p is the penetration factor of PM through leaks in building envelopes and major openings (dimensionless, ranging from 0 to 1), Q_i is the infiltration

airflow rate [m^3/h], V is the volume of the store [m^3], E is the indoor emission rate [$\mu g/m^3 \cdot h$], Q_R is the recirculation airflow rate [m^3/h], η is the filter efficiency (dimensionless, ranging from 0 to 1), β is the first-order indoor loss rate of the pollutant by deposition to surfaces and/or reactions [per hour], and Q_{EX} is the exhaust airflow rate. Applying this model assumes a well-mixed space, no phase change processes (case of particulates, gas-to-particle transformations and vice versa), and losses by homogenous and heterogeneous reactions are negligible when compared to removal by ventilation (case of gas phase contaminants). The parameters shown in Equation 1 are discussed in details in Section 2.2.

2.1.3. Exposure assessment

The disability-adjusted life year (DALY) metric was chosen as the common metric of harm to compare exposures to contaminants of concern for different control strategies, instead of relying on the widely varying published health standards or guidelines. This metric is commonly used and is endorsed by the World Health Organization (WHO). Pollutant concentrations calculated from section 2.1.2 were used in Monte Carlo simulations to quantify the DALYs lost central estimates by sampling from uncertainty distributions of DALY factors reported in epidemiological or extracted from animal toxicity data studies, following the work of Logue et al., 2012.

2.1.4. Energy usage assessment

To assess the energy consumption of different ventilation and filtration scenarios, we built an energy model for a typical retail store (10,000 m^2) using e-Quest software. We used this model to compare the energy required to cool and distribute the air throughout

the building, over the range of different ventilation scenarios, and the six U.S. cities. As mentioned earlier, the second exposure control scenario repeats the assessment of different ventilation rates but replaces the low $PM_{2.5}$ efficiency filter (ASHRAE Standard 52.2-2012 Minimum Efficiency Reporting Value; MERV 8) with higher $PM_{2.5}$ efficiency filter (MERV 13). The energy used by fans when a low $PM_{2.5}$ efficiency filter is installed was adjusted to account for the additional fan energy used in case a high $PM_{2.5}$ efficiency filter was installed. Zaatari et al. (2013) reported through fieldwork and modeling that replacing a MERV 8 filter with a MERV 13 filter in units with fan speed control (i.e. units that maintain the supply airflow constant) increase the fan power draw by 11%. Because the goal is to discern the impact of a specific ventilation scenario (i.e. specific air exchange rate) on energy and air quality, the fan-only mode where air exchange rate can be increased beyond what is defined in ASHRAE Standard 62.1-2010 when free cooling is provided by the low outdoor temperatures (i.e. economizer mode) was disabled. In addition, the analysis takes into account cooling only as the filtration fieldwork analysis was conducted for cooling mode.

2.2. Simulation inputs

2.2.1. Locations, schedules, retail types, occupancy, dimensions

To cover multiple climatic and outdoor air quality scenarios, the exposure control scenarios were modeled in Austin, Philadelphia, Los Angeles, Minneapolis, Phoenix, and Seattle. Outdoor concentrations for each of the cities were divided into two seasons (summer and winter, by calendar) and two time periods that correspond to store open and store closed times. The retail schedule generally varies depending on the type of the

retail. In this paper, the store open hours were considered to be from 7:00 am till 10:00 am and closed otherwise. The outputs of pollutant concentrations were provided for both night and day period, whenever the emission data was known.

Retail stores were classified into two general types: grocery and non-grocery type stores. At the beginning four retail types were chosen: grocery, general merchandise, home improvement stores, other (including electronics, offices, and furniture stores). The division was made based on reported differences in indoor emission rates, and occupancy frequency. The results showed that there is not much difference between the retail types other than grocery, especially that the ventilation rates specified by the VRP is the same for these retail types.

Occupancy parameter is an important parameter to determine the ventilation rate specified by DCV. Generally, there is a lack of studies that measured occupancy in any environment, and specifically in retail buildings. To determine occupancy ranges, the maximum number of occupants per floor area reported in 20 unique retail stores in two studies was used (Chan et al., 2012; and Siegel at al., 2013). The hourly occupancy profile followed that provided by Ng et al. (2012) for stand-alone reference retail buildings; the proposed profile is similar to that observed by Siegel et al. (2013). In their report, Siegel et al. (2013) observed that the trend of occupancy was repeatable on a daily basis for the measured stores; also the occupancy did not vary by season when comparing values of different test weeks in the same store. Thus, no further adjustments for season was done. We added the number of employees to the number of customers following the

average number reported by EIA, (2003; Table B-1): 0.8 workers/100 m^2 (1 worker/1,246 ft) non-grocery stores and 1.3 workers /100 m^2 (1 worker/877 ft) for grocery stores.

It should be noted that some of the observed maximum occupancy numbers reported from literature were higher than what is reported as maximum number in ASHRAE. For example, Chan et al. (2012) reported the maximum number of people observed for a grocery store to be 10 people/100 m^2, which is higher than ASHRAE maximum number of 8 people/100 m^2. This will play a role in decreasing the difference between ventilation rates suggested by DCV and VRP.

For each retail type, the occupancy sample values included in the simulation were generated from uniform distributions with lower and higher range equal to the 10^{th} and 90^{th} percentile observed at the stores (10^{th} occupant frequency per 100 m^2= 0.8 for non-grocery, 2.3 for grocery; 90^{th} occupant frequency per 100 m^2= 5.9 for non-grocery, 12.5 for grocery).

The building volume over surface ratio was selected to be 6.1 m (20 ft), based on stand-alone retail reference building provided by Ng et al. (2012). The building height was used to normalize some input parameters by floor area.

2.2.2. HVAC parameters

Supply rate. The total supply rate was assumed to be equal to 1.94 per hour, as reported by Ng et al. (2012). This value is consistent with the values used in Apte et al. (2011) for a target store (1.99 per hour) and within the uncertainty of the value reported by Siegel et al. (2013) for 14 stores (1.67 ± 0.6 per hour).

Infiltration rate. The infiltration rate during the day was assumed to follow a lognormal

distribution with a geometric mean of 0.35 h^{-1} and geometric standard deviation of 2.1 as developed by Chan (2006) who used a set of leakage measurements a combination of the LBL and Shaw-Tamura infiltration models to predict infiltration in unpressurized buildings over the US commercial building stock using the monthly averaged climatic data. Chan's findings agree with findings from Lagus and Grot (1995) and Cummings et al. (1996). These studies measured commercial buildings in Florida and in California, and found that the infiltration rate alone produced air-exchange rates that were 10-80% of those with the ventilation system on, with a mean ratio of about 40%. From Siegel et al. (2013) report, the infiltration rates were calculated by subtracting the measured total supply rate from the mechanical ventilation rate. The median infiltration rate was found also to be consistent with Chan's finding 0.322 per hour.

Inside retail stores, infiltration is mainly driven by large openings such as entrance, exit, and loading docks. Most of the stores tested in Siegel et al. (2013) were unpressurized. At night, all openings were assumed to be closed, and since retail stores have very little to no windows the infiltration rate was assumed to be much less at night, and arbitrarily divided by 4.

Ventilation rate. Three main ventilation rate strategies were considered; these rates corresponded to those calculated from the VRP, DCV-based CO_2, and IAQP (as described in ASHRAE Standard-62.1-2010). Two additional strategies were investigated as alternatives to the VRP (VRP-C, VRP-NG), and are only reported for some of the analyses in this paper. Strategies other than the IAQP scenario are described in Figure 1. VRP-C differ from the VRP by setting the ventilation rate zero at night, and VRP-NG

differ from the VRP by reducing the ventilation rate specified for non-grocery stores to be equal to that specified for grocery stores.

For all scenarios, the recirculation rate (supply rate minus outdoor ventilation rate) during the store-closed period was set equal to zero.

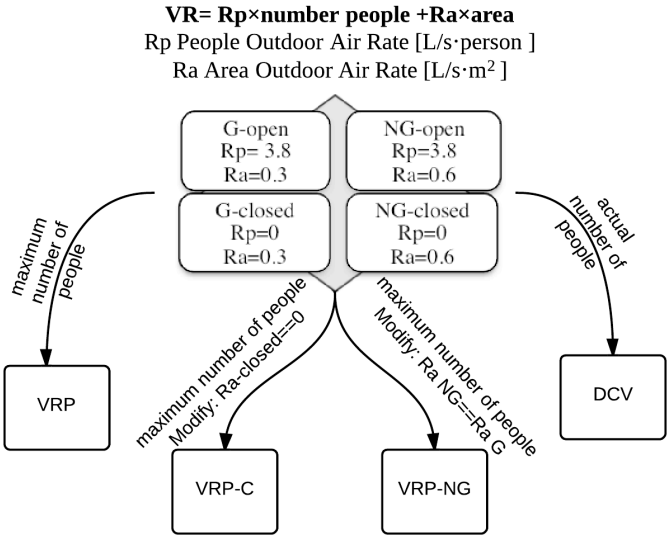

Figure 1: Summary of ventilation scenarios for grocery stores (designated as G), and non-grocery stores (designated as NG).

As mentioned previously, the IAQP depends on an objective and subjective assessment of the indoor air quality in the space. In this work, the objective assessment was based solely on $PM_{2.5}$ concentrations: we applied Equation 1 on $PM_{2.5}$ to find the ventilation rate that will result in the desired $PM_{2.5}$ concentration. This concentration is defined in ASHRAE Standard 62.1-2010 (Appendix B; ASHRAE, 2010) and corresponds to the National Ambient Regulatory limit (NAAQS) of 12 $\mu g/m^3$ over a one-year average. Acrolein (i.e., the other identified contaminants of concern) was not included in the objective assessment because indoor and outdoor acrolein concentrations were higher

than the acrolein established limits at all stores, limiting the value of ventilation to decrease its indoor concentration.

The subjective assessment was based on the value reported by Dutton et al., 2013. Dutton et al. (2013) concluded that an air exchange rate of 0.2 per hour could satisfy at least 80% of occupants in retail buildings. It is important to note that Dutton et al., (2013) only evaluated one big box retail store and relied on simulated shoppers (i.e., not actual shoppers or store employees) to evaluate satisfaction with IAQ.

2.2.3. Outdoor concentrations

Outdoor $PM_{2.5}$ was taken from EPA-hour resolved monitoring data for 2012 year. The samples included are all urban samples except for Austin and Minneapolis, where sub-urban samples were used because the data were not available. EPA reference method data were used for all cities except for Austin and Philadelphia, where none were available and non-reference method data were used. The sample values included in the simulation were generated from lognormal distributions that represent each of the six cities, two seasons, and two periods. Median, 10^{th}, and 90^{th} percentiles of $PM_{2.5}$ outdoor concentrations are provided in the Supporting Information.

For the remaining pollutants, outdoor concentrations were taken from the median measured in State of California (ARB, 2012): outdoor formaldehyde concentration equaled to 2.46 $\mu g/m^3$, outdoor acetaldehyde concentration equaled to 1.9 $\mu g/m^3$, outdoor acrolein concentration equaled to 0.79 $\mu g/m^3$. The outdoor California reference concentrations of formaldehyde and acetaldehyde were consistent with measurements done in Pennsylvania, reported in Siegel et al. (2013). However, these values are lower

(factor of 2 or less) than those provided by RIOPA study for Los Angeles data conducted between May 1999 and February 2001. It should be noted that the reported outdoor concentration of state of California (ARB) data prior to 2001 were higher than the recent years and consistent with values reported in the RIOPA study. For acrolein, we are not aware of available reliable data from sources other than the ARB.

2.2.4. Emission rates

For $PM_{2.5}$, indoor emission rate calculation was explained in details in Zaatari et al. (2013). In this paper, we repeated the emission rate parametric analysis to generate emission rates during day and night. Box plot showing emission rate by 14 different stores (24 different sites visits, n=5 grocery stores, n=14 non-grocery stores) for day and night period in the Supporting Information. In addition, emission rates from Chan et al. were also included (n=3 grocery stores, and n=1 non-grocery store). Median emission rates for grocery stores during the day ranged from 2.3 to 48.2 $\mu g/m^2 \cdot h$. and during the night 0.3 to 15.4 $\mu g/m^2 \cdot h$. Median emission rates for non-grocery stores during the day ranged from 0.01 to 13.8 $\mu g/m^2 \cdot h$. and during the night 0.01 to 3.3 $\mu g/m^2 \cdot h$.

For other pollutants (formaldehyde, acetaldehyde, and acrolein), only emission rates during the day period were available from literature studies. Median emission rates for formaldehyde reported by Grimsrud et al. (2011, n=3), Bennett et al. (2012, n=7), Chan et al. (2012, n=8), and Siegel et al. (2013, n=18) for non-grocery stores ranged from 64 to 110 $\mu g/m^2 \cdot h$; with median emission rate across all studies equal to 80.1 $\mu g/m^2 \cdot h$. Median emission rates for formaldehyde reported by Chan et al. (2012, n=3), and Siegel et al.

(2013, n=5) for grocery stores ranged from 26.3 to 53 μg/m^2·h; with median emission rate across these two studies equal to 39.65 μg/m^2·h.

Median emission rates for acetaldehyde reported by Bennett et al. (2012, n=6), Chan et al. (2012, n=7), and Siegel et al. (2013, n=12) for non-grocery stores ranged from 13 to 57.9 μg/m^2·h; with median emission rate across all studies equal to 14.8 μg/m^2·h. Median emission rates for acetaldehyde reported by Chan et al. (2012, n=3), and Siegel et al. (2013, n=7) for grocery stores ranged from 130 to 266 μg/m^2·h; with median emission rate across all studies equal to 198 μg/m^2·h.

Emission rates for acrolein were reported only by one study (Chan et al., 2012, n=6: 3 grocery stores and 3 non-grocery stores): median emission rate for grocery stores was 23 μg/m^2·h and average was 30.4 μg/m^2·h and for non-grocery stores was 9 μg/m^2·h and average was 14.49 μg/m^2·h.

For each retail type, the sample values included in the simulation were generated from uniform distributions with lower and higher range equal to 10th and 90th percentiles observed at the considered stores.

2.2.5. Filtration, deposition, and penetration

Particulate filtration efficiency, deposition rate, and penetration factor are essentially unknown for commercial buildings. Methods for estimating these parameters are explained in details is in Zaatari et al. (2013). In summary, the 10th, and 90th percentile of $PM_{2.5}$ filter efficiency corresponds to 17% and 32% respectively; and the 10th, and 90th percentile of $PM_{2.5}$ deposition rate corresponds to 0.17 [per hour] and 0.36 [per hour]

respectively. Particle penetration factor was estimated to be equal to unity. For gas phase contaminants (acrolein, formaldehyde, and acetaldehyde), the depositional, filtration and penetration losses are negligible.

3. RESULTS AND DISCUSSION

The result section is divided into three sub-sections. The first section explores the effects of different ventilation scenarios on the identified contaminants of concern (i.e., $PM_{2.5}$ and acrolein); formaldehyde and acetaldehyde were also analyzed to ensure that the selected control strategy did not increase the concentration of these pollutants beyond a level where they will be considered as COCs. The second section repeats the work of the first section but replaces the low $PM_{2.5}$ efficiency filter with a high $PM_{2.5}$ efficiency filter. The third section presents the DALYs lost and the energy results for the different considered scenarios and determines the optimal ventilation-filtration combination strategy that led to a balance between the DALYs lost and energy consumption.

Effect of ventilation scenarios on pollutants concentrations: The case of low $PM_{2.5}$ efficiency filter

Figure 2 displays distributions of $PM_{2.5}$ indoor concentrations for six major U.S. cities (Austin, Philadelphia, Minneapolis, Seattle, Los Angeles, Phoenix), two seasons (summer and winter), and are differentiated for store-open and store-closed periods, grocery and non-grocery retail types, and four different ventilation strategies.

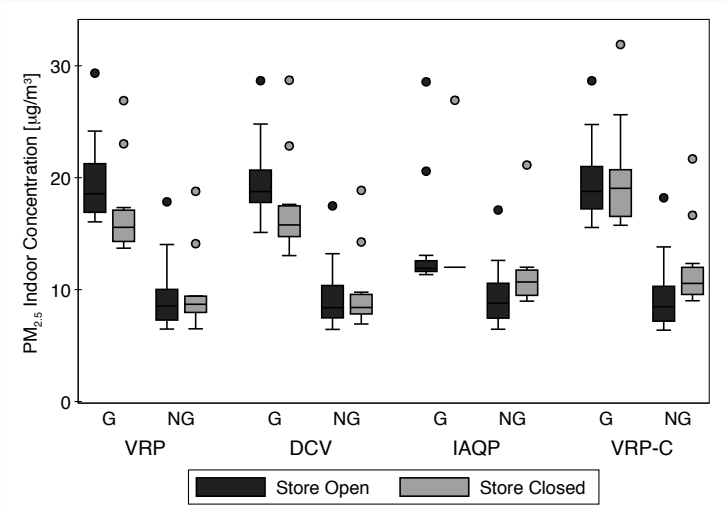

Figure 23: Box and whisker plots of modeled $PM_{2.5}$ indoor concentrations for six U.S. cities (Austin, Philadelphia, Minneapolis, Seattle, Los Angeles, Phoenix), two seasons (summer and winter) over store schedule, retail type (G: grocery, and NG: non-grocery), and different ventilation strategies.

$PM_{2.5}$ concentrations in grocery stores were higher than those reported for non-grocery stores for any ventilation scenario, mainly due to higher indoor emission sources. These results are consistent with those reported in Zaatari et al. (2013) and Chan et al. (2012). Across all cities, median $PM_{2.5}$ indoor concentrations were comparable when applying the VRP or the DCV strategy. The VRP-C yielded higher concentrations when compared to the VRP for store-closed period because of lower ventilation rates (VRP-C assumes zero ventilation rate when the store is closed). Comparing IAQP with the other strategies revealed that IAQP resulted in lower median $PM_{2.5}$ concentration for grocery stores and higher median value for non-grocery stores. For all ventilation strategies, stores located in at least one city had indoor $PM_{2.5}$ concentrations higher than 12 $\mu g/m^3$ ($PM_{2.5}$ concentration of interest defined in ASHRAE Standard 62.1-2010). This finding is less expected when following the IAQP because the ventilation rates were calculated by setting $PM_{2.5}$ concentration equal to 12 $\mu g/m^3$. To explore this issue, Figure 3 shows $PM_{2.5}$

concentrations and air exchange rate for two cities (Austin and Los Angeles), differentiated by store type and season.

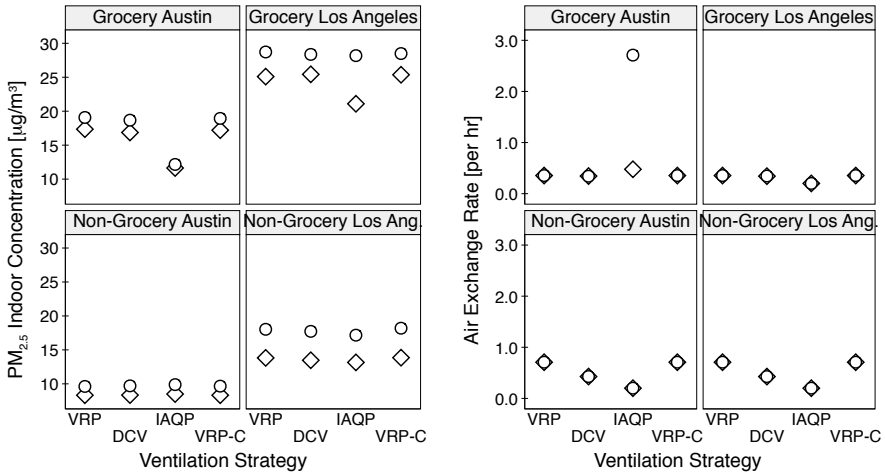

Figure 24: $PM_{2.5}$ indoor concentrations (left graph) and air exchange rate (right graph) for store-open period for two cities (Austin and Los Angeles), differentiated by store type and season. Hollow circles represent the summer season and hollow diamonds represent the winter season.

Results in Figure 3 show that in Austin, it was favorable to increase the air exchange rate in grocery stores to dilute indoor-generated $PM_{2.5}$ because the $PM_{2.5}$ indoor-to-outdoor ratios in these stores was larger than unity. Considering the IAQP, $PM_{2.5}$ concentration in grocery stores is Austin was maintained at 12 $\mu g/m^3$ with average air exchange rate of 1.6 per hour (350% increase than the VRP). In Los Angeles, the outdoor concentration was higher than the indoor concentration ($PM_{2.5}$ IO ratios lower than unity); this required the air exchange rate to be reduced to the minimum allowed to minimize admission of airborne pollution indoors. According to ASHRAE standard 62.1-2010, the minimum air exchange rate in an occupied zone can not be zero; rather, it should be equal to the air exchange rate identified by a subjective assessment of IAQ identified based on occupants' survey responses collected for a similar type of building. The minimum air exchange rate that matches the satisfaction of 80% of occupants in retail buildings corresponds to 0.2 per hour (Dutton et al., 2013). This minimum air exchange rate was not sufficient to keep $PM_{2.5}$ indoor concentration in grocery stores

below 12 μg/m^3: the combination of high infiltration rate along with high outdoor concentration in these stores was likely the cause.

For non-grocery stores, following the IAQP, both cities (Austin and Los Angeles) were assigned the minimum air exchange of 0.2 per hour. In Austin, $PM_{2.5}$ indoor and outdoor concentrations were low (lower than 12 μg/m^3) and no extra dilution was needed. In Los Angeles, again, $PM_{2.5}$ indoor concentrations were greater than the desired concentration because of the elevated outdoor concentration and the high infiltration rate.

To explore the effects of these ventilation rates on acrolein, formaldehyde, and acetaldehyde concentrations, Monte Carlo simulations were repeated to calculate the concentrations of these pollutants, given the different ventilation strategies. Figure 4 shows the average indoor concentrations of formaldehyde, acetaldehyde, and acrolein during store-open period for six U.S. cities over retail type and different ventilation strategies.

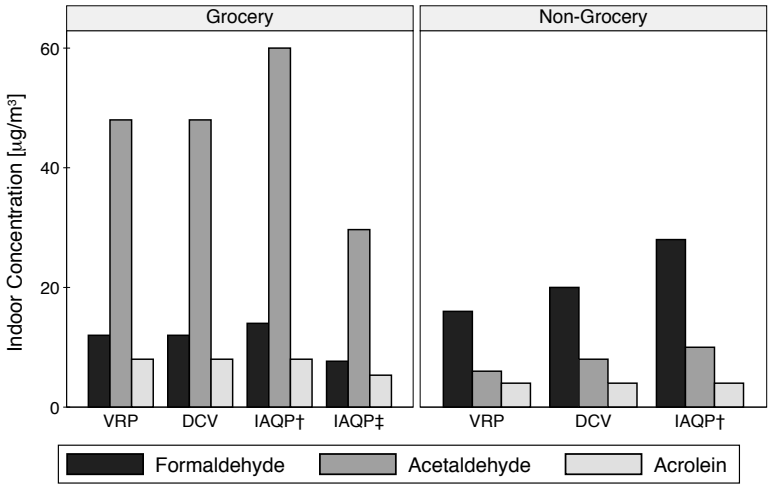

Figure 25: Bar charts of formaldehyde, acetaldehyde, and acrolein indoor concentrations for six U.S. cities over retail type and different ventilation strategies.

As shown in Figure 4, the IAQP applied to grocery stores with $PM_{2.5}$ indoor-to-outdoor ratios less than unity yielded the highest pollutant concentrations because the

recommendation for these stores is to keep the air exchange rate to a minimum. Since formaldehyde, acetaldehyde, and acrolein are pollutants mainly generated indoors, lowering the air exchange increased these pollutants (formaldehyde and acrolein concentrations were 22% and acetaldehyde was 27% higher that those reported using the VRP). By analogy, applying the IAQP on grocery stores with $PM_{2.5}$ indoor-to-outdoor ratios larger than unity (i.e., increasing air exchange rate above that required by the VRP or the DCV) yielded the lowest formaldehyde, acetaldehyde, and acrolein concentrations (formaldehyde and acrolein concentrations were 31% and acetaldehyde was 38% lower that those reported for the VRP). For non-grocery stores, indoor concentrations were low; thus, air exchange rates were kept to a minimum, causing an increase of the concentrations of pollutants other than $PM_{2.5}$.

The question remains of how much the increase or decrease in pollutants concentrations translates into gain or loss in exposure benefits. Before answering this question, the next section explores the change in pollutants concentrations when using a high $PM_{2.5}$ efficiency filter.

Effect of ventilation scenarios on pollutants concentrations: The case of high $PM_{2.5}$ efficiency filter

This section investigates the impact of using a high $PM_{2.5}$ filter efficiency on pollutant concentrations: in the calculation, MERV 8 was switched to MERV 13. Figure 5 is a repeat of Figure 2 but with a high efficiency filter. Note that the ventilation rate determined by the IAQP-based $PM_{2.5}$ is the ventilation rate calculated to reduce $PM_{2.5}$ below 12 $\mu g/m^3$ with a high efficiency filter in place.

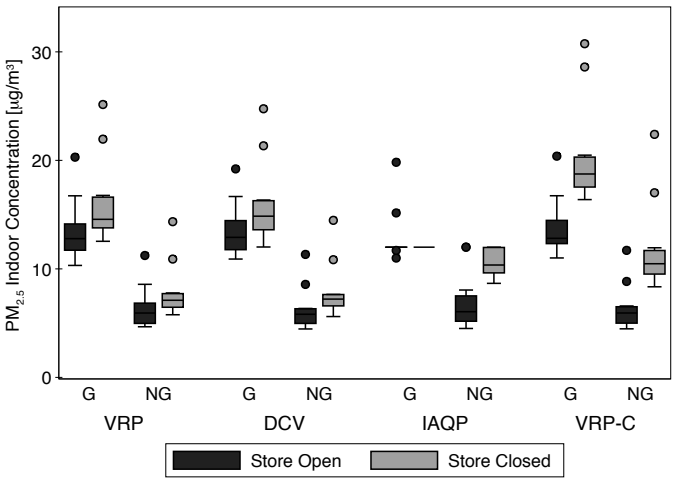

Figure 26: Box and whisker plots of modeled $PM_{2.5}$ indoor concentrations for six U.S. cities (Austin, Philadelphia, Minneapolis, Seattle, Los Angeles, Phoenix), two seasons (summer and winter) over store schedule, retail type (G: grocery, and NG: non-grocery), and different ventilation strategies

As expected, using a high $PM_{2.5}$ efficiency filter (efficiency = 80%) decreased $PM_{2.5}$ indoor concentrations for all ventilation strategies; nonetheless, $PM_{2.5}$ indoor concentrations were higher than 12 $\mu g/m^3$ in at least one city for any of the ventilation strategies. For IAQP, the calculated air exchange rate was 0.2 per hour (minimum allowed) for all cities and for both retail types. This air exchange rate was sufficient to reduce elevated $PM_{2.5}$ concentrations in stores with $PM_{2.5}$ IO ratios larger than unity. For the stores with high outdoor concentrations, the high infiltration rates and emission rates present at these stores diminished the effect of high filtration efficiency, resulting in $PM_{2.5}$ concentrations higher than 12 $\mu g/m^3$.

Figure 6 is a repeat of Figure 4 but with using a high efficiency filter. The only difference between both figures is that the IAQP increases formaldehyde, acetaldehyde, and acrolein concentrations for both stores with $PM_{2.5}$ IO ratios higher and lower than unity, suggesting the danger of focusing on single pollutant strategies. As mentioned above, the calculated air exchange rate from the IAQP for all stores was 0.2 per hour,

resulting in a decrease of dilution of these pollutants (i.e., higher formaldehyde, acetaldehyde, and acrolein concentrations).

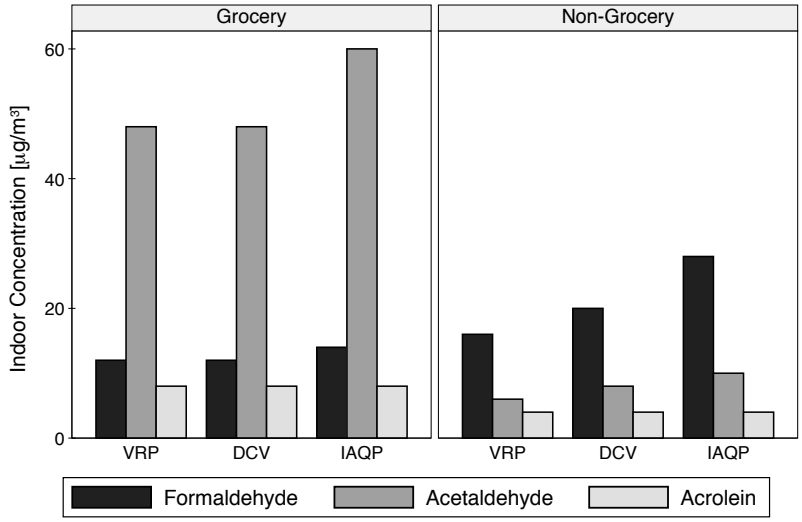

Figure 27: Bar charts for formaldehyde, acetaldehyde, and acrolein indoor concentrations for six U.S. cities over retail type and different ventilation strategies.

The next section puts these results in context of exposure and energy consumption.

Energy and exposure consequences of different ventilation and filtration scenarios

This section evaluates the control options of indoor exposures to pollutants by presenting the burden of disease (i.e., DALYs lost) attributable to each pollutant and the corresponding energy consumption.

Figure 7 shows the DALYs lost from exposure to acetaldehyde, acrolein, formaldehyde, and $PM_{2.5}$ for different ventilation scenarios and filter efficiencies (top figure), and a zoom of DALYs attributed to exposure to $PM_{2.5}$ and acrolein (bottom figure). Note that the y-axis on the top figure is a log-scale. A new ventilation scenario was introduced in this figure (VRP-NG) that reduced the ventilation rate required for non-grocery store to that required by the grocery store (as explained earlier in Figure 1).

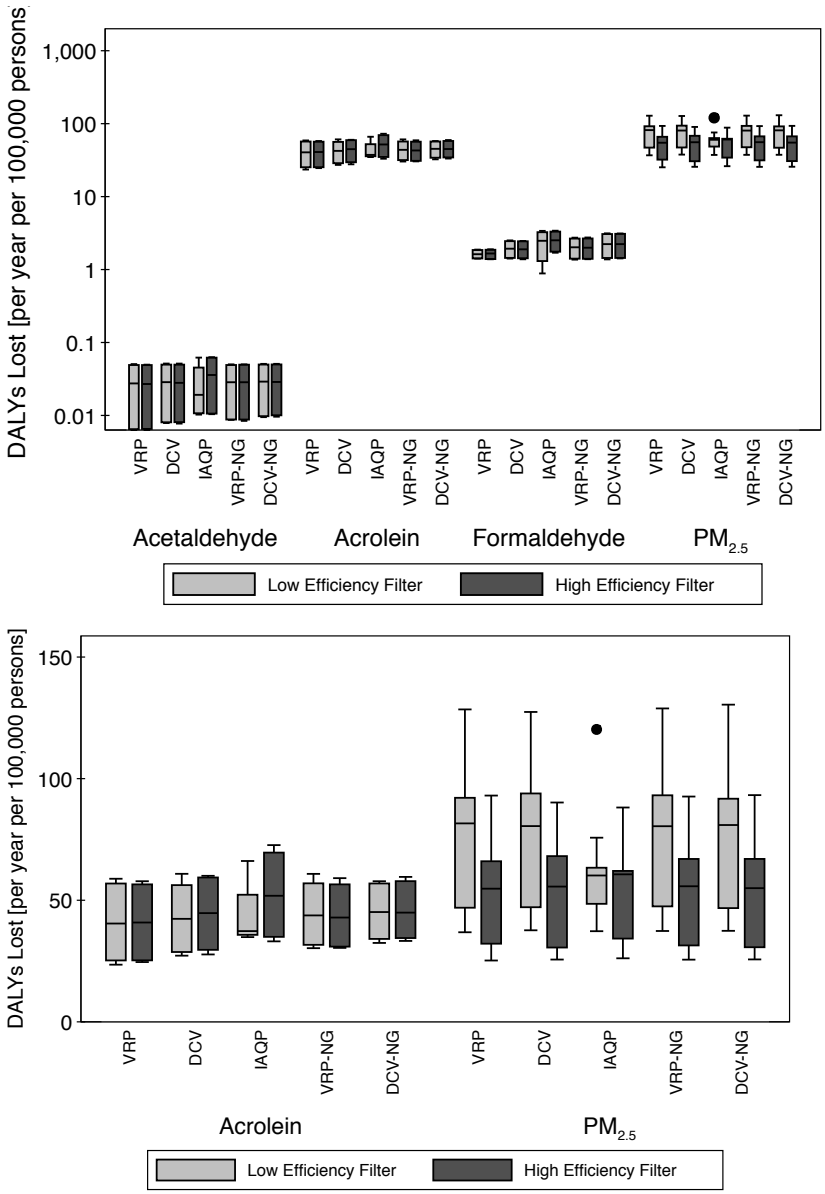

Figure 28: Box and whisker plots of modeled DALYs lost from exposure to acetaldehyde, acrolein, formaldehyde, and $PM_{2.5}$ for different ventilation scenarios and filter efficiencies (top figure). The y-axis is a log-scale. A zoom of DALYs attributed to exposure to $PM_{2.5}$ and acrolein is presented in the bottom figure.

The top figure in Figure 7 shows that the DALYs attributed to formaldehyde and acetaldehyde for all ventilation scenarios were negligible when compared to those attributed to acrolein and $PM_{2.5}$, and the bottom figure shows that the DALYs attributed to $PM_{2.5}$ were 60% higher than those attributed to acrolein (e.g., median DALYs per

100,000 persons across all scenarios attributed to formaldehyde = 1.79; acetaldehyde = 0.019; acrolein = 37.4; $PM_{2.5}$ = 60 DALYs).

Table 4 summarizes the total DALYs lost calculated by summing the DALYs from exposure to all pollutants. It is important to remember that the total DALYs were essentially the sum of DALYs from exposure to acrolein and $PM_{2.5}$. This table also shows the percentage change of different ventilation and filtration scenarios relative to the VRP/low $PM_{2.5}$ filter efficiency scenario.

Table 6: Summary of DALYs lost for different control scenarios and percentage change in DALYs lost relative to the VRP/low $PM_{2.5}$ filter efficiency scenario.

	DALYs Lost per year per 100,000 persons [% Change switching from VRP Low PM η]									
City	VRP Low PM η	DCV Low PM η	IAQP Low PM η	VRP-NG Low PM η	DCV-NG Low PM η	VRP High PM η	IAQP High PM η	DCV High PM η	VRP-NG High PM η	DCV-NG High PM η
Grocery										
Aus	147	152	100			124	132	127		
	(0%)	(3%)	(-32%)			(-15%)	(-10%)	(-14%)		
PA	152	150	118			125	130	129		
	(0%)	(-1%)	(-23%)			(-18%)	(-14%)	(-15%)		
LA	186	185	187			151	160	151		
	(0%)	(0.5%)	(0.4%)			(-19%)	(-14%)	(-19%)		
Mn	143	140	104			115	132	121		
	(0%)	(-2%)	(-27%)			(-20%)	(-8%)	(-16%)		
Ph	146	154	113			121	136	124		
	(0%)	(6%)	(-23%)			(-17%)	(-6%)	(-15%)		
Stl	160	160	132			127	133	132		
	(0%)	(-0.2%)	(-17%)			(-21%)	(-17%)	(-18%)		
Non-Grocery										
Aus	75	78	87	83	84	60	74	65	67	71
	(0%)	(4%)	(16%)	(11%)	(13%)	(-19%)	(-2%)	(-14%)	(-11%)	(-5%)
PA	74	81	88	82	86	59	71	64	65	68
	(0%)	(9%)	(20%)	(11%)	(16%)	(-20%)	(-3%)	(-14%)	(-12%)	(-8%)
LA	106	108	114	111	115	80	100	82	87	88
	(0%)	(2%)	(8%)	(5%)	(9%)	(-25%)	(-5%)	(-22%)	(-18%)	(-16%)
Mn	65	69	77	72	75	52	63	58	59	64
	(0%)	(6%)	(18%)	(10%)	(15%)	(-20%)	(-4%)	(-12%)	(-10%)	(-2%)
Ph	67	71	83	74	77	56	70	61	63	65
	(0%)	(5%)	(23%)	(10%)	(15%)	(-17%)	(3%)	(-9%)	(-7%)	(-3%)
Stl	74	79	88	82	85	59	74	64	64	67
	(0%)	(6%)	(18%)	(10%)	(14%)	(-21%)	(-1%)	(-15%)	(-14%)	(-11%)

Table 4 shows that the DALYs lost resulting from the IAQP with high efficiency filter were higher than those reported for IAQP with low efficiency filter for grocery stores. This finding is counter-intuitive. The reason is that using a high efficiency filter decreased $PM_{2.5}$ indoor concentration substantially so that the ventilation rate required was the minimum ventilation rate (i.e. 0.2 per hour). Setting the ventilation rate in grocery stores to a minimum level increased the concentrations of other pollutants generated indoors; of particular interest is acrolein because it contributed significantly to the DALYs lost.

Table 5 summarizes the electrical consumption by city for different ventilation and filtration scenarios.

Table 7: Summary of electricity consumption for different control scenarios and percentage change in electricity consumption relative to the VRP/ low $PM_{2.5}$ filter efficiency scenario.

	Electricity Consumption-Cooling kWh x 1000 [% Change switching from VRP Low PM η]									
City	VRP Low PM η	DCV Low PM η	IAQP Low PM η	VRP-NG Low PM η	DCV-NG Low PM η	VRP High PM η	IAQP High PM η	DCV High PM η	VRP-NG High PM η	DCV-NG High PM η
Grocery										
Aus	175 (0%)	173 (-1%)	399 (128%)			186 (7%)	156 (-11%)	184 (6%)		
PA	81 (0%)	81 (0%)	168 (108%)			90 (11%)	87 (7%)	90 (11%)		
LA	62 (0%)	63 (1%)	75 (20%)			72 (16%)	85 (36%)	73 (17%)		
Mn	81 (0%)	81 (0%)	88 (9%)			90 (11%)	89 (10%)	90 (11%)		
Ph	179 (0%)	178 (-1%)	238 (33%)			193 (8%)	171 (-4%)	192 (7%)		
Stl	42 (0%)	43 (2%)	60 (43%)			52 (25%)	70 (68%)	53 (27%)		
Non-Grocery										
Aus	245 (0%)	191 (-22%)	145 (-41%)	175 (-29%)	155 (-37%)	256 (5%)	156 (-36%)	202 (-17%)	186 (-24%)	167 (-32%)
PA	94 (0%)	84 (-11%)	78 (-17%)	81 (-14%)	79 (-17%)	103 (10%)	87 (-8%)	93 (-1%)	90 (-4%)	88 (-7%)
LA	33 (0%)	56 (68%)	75 (126%)	62 (89%)	70 (113%)	43 (30%)	85 (156%)	66 (99%)	72 (119%)	80 (144%
Mn	86 (0%)	82 (-5%)	80 (-7%)	81 (-6%)	80 (-7%)	95 (11%)	89 (4%)	91 (5%)	90 (5%)	89 (4%)
Ph	238 (0%)	191 (-20%)	157 (-34%)	179 (-25%)	165 (-31%)	252 (6%)	171 (-28%)	205 (-14%)	193 (-19%)	178 (-25%)
Stl	18 (0%)	3 (-94%)	60 (226%)	42 (128%)	52 (186%)	29 (58%)	70 (284%)	46 (152%)	52 (186%)	63 (244%

Before identifying the optimal control strategy, it is important to mention that (1) lowering the air exchange in Los Angeles and Seattle increased electrical consumption because outdoor air conditions helped in cooling the store, and (2) on average DCV offered no advantage over VRP in grocery stores from either energy or air quality perspectives; the air exchange specified for DCV is approximately equal to that specified for VRP.

Synthesizing the data displayed in Table 4 and Table 5 shows that for grocery stores located in cities where outdoor $PM_{2.5}$ concentration is low (median $PM_{2.5}$ outdoor

concentration less than 12 μg/m^3, all sampled cities except Los Angles in our case), adopting the IAQP with low efficiency $PM_{2.5}$ filter led to the highest health benefits (on average 26% DALYs less than the VRP). This gain in health benefits translated into 113% average increase in air exchange rate (72%, on average, increase in electrical consumption of ventilation alone, $8,800 average increase in annual cost, with the annual increase in Austin being the highest at $25,000). For grocery stores located in Los Angeles, adopting the VRP with high $PM_{2.5}$ filter efficiency led to the highest health benefits, with electrical consumption equal to $1,100 (additional energy cost of higher efficiency filtration) more than that realized by the VRP with low efficiency filter.

For non-grocery stores, it is most beneficial from health perspective to adopt the VRP with high $PM_{2.5}$ efficiency (on average 20% less DALYs than the VRP with low efficiency filter), with electrical consumption equal to $1,200 (additional energy cost of higher efficiency filtration). Adopting the reduced VRP with high $PM_{2.5}$ efficiency filter instead of the VRP with high $PM_{2.5}$ efficiency filter led to sacrificing 8% of health benefits for half the air exchange rate ($1,700 decrease in annual cost).

A comparison between the ventilation rates prescribed by the VRP (ASHRAE Standard 62.1-2010) for grocery and non-grocery stores revealed that non-grocery stores are recommended higher ventilation rates than grocery stores (approximately double). As for the IAQP (ASHRAE Standard 62.1-2010), results contradict the VRP results because grocery stores have higher emissions and higher indoor concentrations, and thus naturally they should be recommended higher ventilation rates (with the caveat that outdoor air concentration are low). A revision of ASHRAE Standard 62.1 is warranted given the new information available about retail stores; ventilation rates in grocery stores should be higher.

Pollutant control ventilation strategy. For all cities and for all retail types, adopting a tailored ventilation/filtration strategy (specific to retail type and location) led to substantial decrease in DALYs lost. A tailored control strategy should take into consideration all contaminants of concern: a single pollutant-approach to determine ventilation rates may overlook significant exposures from other pollutants generated from a different source (indoors vs. outdoors). Although the indoor air quality procedure (IAQP) specified in ASHRAE Standard 62.1-2010 requires the determination of the minimum ventilation rate to keep each contaminant of concern below its established limits, this method converges to the single pollutant-approach as it requires the engineer to choose the larger ventilation rate determined by the different COCs. Thereby ignoring the fact that some pollutants (e.g. $PM_{2.5}$) can be generated outdoors and keeping the ventilation rate to a minimum might be more advantageous from both exposure and energy perspectives. Further, the established limits for each pollutant vary widely and it might not be feasible to calculate the ventilation rate that will drive the pollutant below its limit. For example, acrolein levels indoors and outdoors were higher than the REL and depending on ventilation alone will not drive acrolein below its REL. As demonstrated in results shown in Table 4 and Table 5, following the VRP or DCV specified in ASHRAE Standard 62.1-2010 might not be the optimal solution from both energy and exposure perspectives.

Instead of relying on a single pollutant (IAQP-ASHRAE Standard 62.1-2010) or a prescribed number (VRP-ASHRAE Standard 62.1-2010), a proposed approach is the pollutant exposure control ventilation (PECV) strategy. The PECV finds an optimal ventilation rate based on weighing the exposures of different contaminants of concern, following the same DALY approach for the contaminants of concern identified in this work. However, ventilation rates are not limited only to those recommended by the

standards. An example of applying PECV on a grocery store located in Los Angeles is shown in Figure 8.

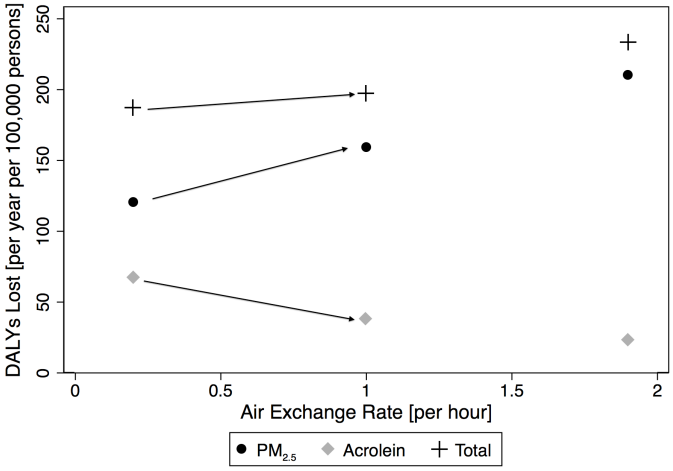

Figure 29: DALYs lost function of air exchange rate for a grocery store in Los Angeles.

Figure 8 shows that the total DALYs lost for air exchange rate between 0.2 and 1 per hour are comparable. The low air exchange rate (i.e., 0.2 per hour) increased DALYs lost from acrolein and decreased those for $PM_{2.5}$; the opposite happened when applying the higher air exchange rate (i.e., 1 per hour). Thus, the proposed strategy offers the flexibility to choose the air exchange rate that will lead to lower energy consumption. As mentioned previously, in Los Angeles, increasing the air exchange rate is beneficial to decrease the energy consumption because the conditions of the outdoor air are favorable for free cooling. Note that an additional decrease in DALYs lost can be obtained if a high efficiency filter is used, but this translates into additional cost as mentioned previously. The suggested ventilation strategy depends mainly on the uncertainties associated with the estimated exposures (i.e. DALYs lost). The uncertainty bounds on exposure to some of the pollutants were higher than one order of magnitude. Besides, there were limitations associated with the use of ambient rather than indoor epidemiological studies, assuming linear relationship between exposure to acrolein and formaldehyde and health effects (i.e., eliminating the threshold effect), excluding health effects with incomplete

quantitative information (i.e., might be excluding effects from important pollutants such as ultrafine and SVOCs). However, as those available health factors and the most recent scientific knowledge are used, the order of magnitude of the results should be sufficiently reliable for prioritizing pollutant control strategies.

4. CONCLUSION

This study modeled the impact of (1) adjusting ventilation only; (2) adjusting filtration of supply air only; and (3) adjusting ventilation and filtration together on exposure and energy consumption in retail buildings. All approaches were able to provide substantial reductions in the health risks (19-26% decrease in DALYs lost); the magnitude of the reductions depended on the ventilation/filtration scenario, the retail type, and the city. The magnitude of energy cost to achieve the maximum health benefits depended on the city and the retail type (ranging from $1,100 in annual cost for filtration energy in a grocery store in Los Angeles to $24,000 as the annual cost of ventilation in a grocery store in Austin; results are for a 10,000 m^2 store). A proposed strategy as alternative to the ventilation strategies recommended by the standards is the pollutant exposure control ventilation (PECV) strategy. This strategy is based on (1) weighing the exposures of different contaminants of concern found in retail buildings, (2) identifying the range of ventilation rates that lead to low DALYs lost, and (3) choosing the optimal ventilation rate that leads to energy usage savings in the climate considered.

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