MEASURING THE CONCENTRATION
OF WOODSTOVE SMOKE PARTICULATE MATTER PM$_{2.5}$
IN A SINGLE-ROOM FIRST NATIONS HOUSE

by

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Abstract

This research study examined the degree to which residents are exposed to $PM_{2.5}$ concentration levels due to wood smoke using several instruments, including six Dusttrak II 8530, a Dylos DC1700 and a UNI-T. The parametric variations in this study included distance from the woodstove, height above floor level, duration of time the woodstove door remained open for refueling, time between refueling stages, and the effectiveness of various types of air purifiers in reducing $PM_{2.5}$. An extensive instrument calibration and error analysis were also conducted as a way of evaluating the accuracy of the data obtained with different instruments. The results showed that the $PM_{2.5}$ distribution varies with distance from the woodstove and height above floor level. The results also showed that the use of air purifiers incorporating HEPA filters were most effective in reducing $PM_{2.5}$. This research study adds to the body of knowledge and has practical implications in the monitoring and mitigation of indoor levels of $PM_{2.5}$.

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CHAPTER 1. INTRODUCTION

1.1 Background to the Investigation

Fifty percent of the world population and approximately 90% of the rural population in developing countries use wood as an energy source (Bruce, Perez-Padilla, & Albalak, 2000). Literature shows that exposure to wood smoke poses a major health problem to children, including pneumonia, chronic respiratory disease, and Tuberculosis, among others (Torres-Duque, Maldonado, Pe´rez-Padilla, Ezzati, & Viegi, 2008). According to the World Health Organization, 4 million people die each year worldwide due to substandard indoor air quality from biomass burning (WHO, 2010). Residential combustion of wood contains carcinogens, irritants, and cancer-causing agents (Government of Canada, 2016). Furthermore, exposure to particulate matter, often described as fine airborne particles that consist of smoke and soot, is also major cause of concern. Generally, $PM_{2.5}$ and PM_{10} are two measures often widely discussed and found within the literature. These measures consist of particles that are 2.5 and 10 microns in diameter, respectively. Such particles can easily be inhaled deep into the lungs causing difficulty in breathing and leading to cardiovascular damage (Government of Canada, 2016).

Acceptable limits for wood smoke particles have changed over the last few years and vary from jurisdiction to jurisdiction. The U.S. Environmental Protection Agency (EPA) and the Canadian Standards Association have adopted similar standards for safe outdoor and indoor particulate levels as well as recommendations for average 24-hr limits for healthy living (CCME, 2012). According to EPA, safe levels of $PM_{2.5}$ over a 24-hr period are 35 $\mu g/m^3$ while safe levels of

PM$_{2.5}$ over an entire year are 12 µg/m^3 (EPA, 2016). An air quality index for PM$_{2.5}$ levels is given in Table 1.1, while related health implications are shown in Table 1.2 (AQIT, 2018).

Table 1.1 Air Quality Index for PM$_{2.5}$ (Source: EPA, 2016)

AQI Category	Index Values	Previous Breakpoints (1999 AQI) (µg/m^3, 24-hour average)	Revised Breakpoints (µg/m^3, 24-hour average)
Good	0 - 50	0.0 - 15.0	0.0 – 12.0
Moderate	51 - 100	>15.0 - 40	12.1 – 35.4
Unhealthy for Sensitive Groups	101 – 150	>40 – 65	35.5 – 55.4
Unhealthy	151 – 200	> 65 – 150	55.5 – 150.4
Very Unhealthy	201 – 300	> 150 – 250	150.5 – 250.4
Hazardous	301 – 400	> 250 – 350	250.5 – 350.4
	401 – 500	> 350 – 500	350.5 – 500

The research study presented in the thesis is community-based and was conducted at Sagkeeng First Nation in the province of Manitoba and involves an extensive monitoring of the PM$_{2.5}$ concentration in a single room house, shown in Figure 1.2. The testing was carried out between April and June 2016. The single room house was equipped with an operational woodstove, shown in Figure 1.2. The wood used for fuel in this study was primarily well-seasoned tamarack. It was commercially available and was purchased from a single source. The kindling used was comprised of well-seasoned recycled wood and was also purchased from a single source. The test house was originally used to house a family of four, but was not used at the time of the testing. The house is approximately 800 sq. ft. and consists of a single room, as shown in Figure 1.3. Other than a table, a couch, and a small counter top near the stove, the room was mostly empty space.

Table 1.2. Air Quality Index Scale as Defined by the U.S.-EPA 2016 Standard (Source: AQIT, 2018).

AQI	Air Pollution Level	Health Implications	Cautionary Statement (for PM2.5)
0 - 50	Good	Air quality is considered satisfactory, and air pollution poses little or no risk	None
51 -100	Moderate	Air quality is acceptable; however, for some pollutants there may be a moderate health concern for a very small number of people who are unusually sensitive to air pollution.	Active children and adults, and people with respiratory disease, such as asthma, should limit prolonged outdoor exertion.
101-150	Unhealthy for Sensitive Groups	Members of sensitive groups may experience health effects. The general public is not likely to be affected.	Active children and adults, and people with respiratory disease, such as asthma, should limit prolonged outdoor exertion.
151-200	Unhealthy	Everyone may begin to experience health effects; members of sensitive groups may experience more serious health effects	Active children and adults, and people with respiratory disease, such as asthma, should avoid prolonged outdoor exertion; everyone else, especially children, should limit prolonged outdoor exertion
201-300	Very Unhealthy	Health warnings of emergency conditions. The entire population is more likely to be affected.	Active children and adults, and people with respiratory disease, such as asthma, should avoid all outdoor exertion; everyone else, especially children, should limit outdoor exertion.
300+	Hazardous	Health alert: everyone may experience more serious health effects	Everyone should avoid all outdoor exertion

(a) Side View

(b) Front Entrance

Figure 1.1. Test House

(a) View of Woodstove Being Refueled (b) View of Woodstove with Fire

Figure 1.2. Operational Woodstove in the Test House
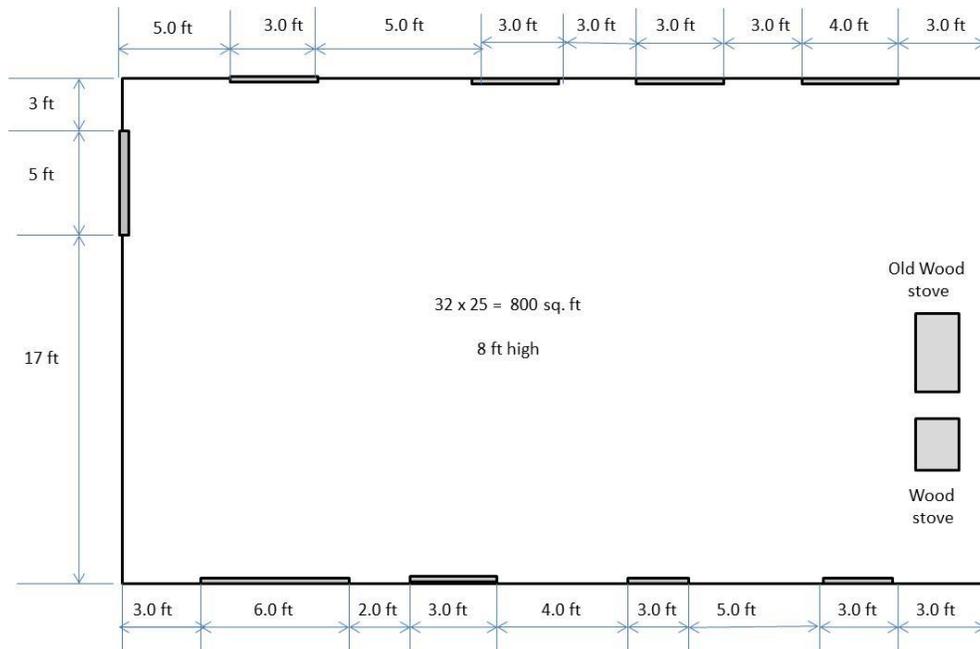
Figure 1.3. Floor Plan of Test House

Woodsmoke particulate consists of both $PM_{2.5}$ and PM_{10} particles. However, $PM_{2.5}$ is characterized in the literature as the more harmful of the two due to their smaller size and, with

regards to the respiratory tract, their increased penetrative capability. Additionally, woodsmoke is predominantly made up of $PM_{2.5}$ with little additional added concentration when detecting PM_{10}. According to Gibson et al. (2015. p. 5), "… there was no significant difference (P = 0.524) between the Dusttrak $PM_{2.5}$ and Dusttrak PM_{10} data, suggesting that the PM_{10} size fraction is dominated by the fine PM mode ($PM_{2.5}$)". This observation is further corroborated by findings in this research study, as described in Appendix B.

The objective in this research study was to assess the degree to which residents are exposed to $PM_{2.5}$ concentration levels in a home. More specifically:

1. How does the $PM_{2.5}$ concentration vary with **distance** from the woodstove, measured normal to the woodstove and at fixed heights from the floor?

2. How does the $PM_{2.5}$ concentration vary with **height** above floor level and at fixed distances from the woodstove?

3. How does varying the amount of time the woodstove door remains open during the refueling process affect indoor $PM_{2.5}$ concentration levels?

4. How does the duration of time the woodstove door remains closed between refueling sessions affect $PM_{2.5}$ concentration levels?

5. How effective are air purifiers (HEPA, ozone) in reducing $PM_{2.5}$ concentration levels?

6. How accurate are the instruments used in monitoring $PM_{2.5}$ concentration levels?

7. Can the cost associated with monitoring $PM_{2.5}$ concentration levels be reduced without compromising accuracy?

The monitoring of the $PM_{2.5}$ concentration levels was conducted using six Dusttrak II instruments manufactured by TSI Inc. (TSI, 2018), a Dylos DC1700 manufactured by the Dylos Corporation, and a UNI-T 938A manufactured by Uni-Trend Technology (China) Co. Ltd. The last two instruments were purchased in the open market while the Dusttrak II instruments were rented for the study. The Dusttrak II cost more than \$10,000 CAD while the Dylos DC1700 cost approximately \$500 CAD, and the UNI-T 938C cost approximately \$200 CAD. This instrument comparison was conducted to determine the effectiveness of significantly cheaper $PM_{2.5}$ detection strategies when compared to the costly Dusttrak II. The Dusttrak II instruments and the Dylos DC1700 are equipped with internal logging capability. The UNI-T 938A however, is not. Data acquisition using this instrument was accomplished through a combination of image manipulation, using character recognition, and various software programs, freely available on the internet. To examine the effectiveness of air purifiers in reducing $PM_{2.5}$ concentration levels, nine different such devices were purchased in the open market, varying in price from \$70 to \$730 CAD.

1.2 Scope of the Thesis

Chapter 1 of the thesis provides an introduction to the research study describing the background to the investigation, outlining the key objectives, and summarizing the methodology, including the various instruments used. Chapter 2 provides a review of the pertinent academic literature. The experimental investigation is detailed in Chapter 3, while results are given in Chapter 4. The analysis and the discussion of the results are provided in Chapter 5. Chapter 6 summarizes the

key findings from this study, discusses the impact of the study on practice, highlights its limitations, and provides recommendations for future research.

This study, for the most part, utilized rented equipment that is mostly used in environments that do not always require the same precision as that demanded by a research investigation. It, thus, became necessary to: carry out instrument calibration according to the manufacturer's requirements, perform an error analysis in order to gauge the accuracy of the results, and make the necessary adjustments to the data. This analysis is provided in Appendix A.

Studies involving indoor $PM_{2.5}$ concentrations are quite expensive to conduct, mainly due to the high cost associated with the monitoring equipment required. An instrument performance evaluation was conducted using data obtained from three instruments: an inexpensive UNI-T 938C and Dylos DC1700, with a rather expensive one: a Dusttrak II instrument. This performance evaluation is discussed in Appendix B.

CHAPTER 2: LITERATURE REVIEW

2.1. Indoor Air Quality Standards for Homes with Woodstoves

In Canada, people can spend up to 90% of their time indoors, which is mainly attributed to the colder climate when compared to other countries around the world (Health Canada, 2015). There is no doubt that the air one breaths in while indoors may pose some serious health concerns that can have a lasting impact on one's overall health, not only for the short term, but also for the long term, if special attention is not paid and care is not taken (Health Canada, 2015; Government of Canada, 2015a).

For individuals living in a cold climate, as in Canada, particularly for people living in remote northern communities and other underserved regions, this is a major cause of concern (CCOSH, 2018). Vulnerable individuals, particularly young school-aged children and the elderly, often can be faced with serious allergies that negatively impact their quality of life. For school-aged children, this can lead to frequent visits to a doctor and absence from school for prolonged periods of time (Daisey, Angell, & Apte, 2003).

Oftentimes, large amounts of mold and mildew, due to buildup of condensation and poor ventilation, can be observed inside the homes of people living in Northern communities (Daisey et al., 2003; Government of Canada, 2015b). This is usually due to improperly maintained heating and ventilation systems which places the residents at greater health risk. Also, the poor type of materials used in the construction of the homes in Northern communities aggravate the housing situation.

Moreover, with different sources of heating often employed during the harsher winters months in such remote communities, variable effects on air quality can be observed (Government of Canada, 2015b). Some of the heating sources used in the north include pellet stoves, fireplaces, fireplace inserts, and in some rare instances central heating.

Standards for measuring indoor air quality vary from region to region and are set by different governing bodies based upon the jurisdiction they fall under (NHDES, 2015). For instance, in Europe, EUROVEN scientists have suggested an outdoor air supply rate of 30 L/s per person, whereas in the U.S., ASHRAE recommends 10 L/s per person (HealthLinkBC, 2018; Charles, Magee, Won & Lusztyk, 2015). The National Research Council of Canada recommends the removal of the contaminants at the source and to rely upon ventilation to remove the air-borne Volatile Organic Compounds or VOCs (Charles et al., 2005).

Nevertheless, in Canada, the Canadian Standards Association (CSA) and/or the Canadian Council of Ministers of the Environment (CCME) are often responsible for providing guidelines on the required standards for air quality, while the Environmental Protection Agency (EPA) is responsible for providing such measures in the U.S.

2.1.1 Sources of Heating

There are several sources of heating for homes in Northern communities that could impact indoor air quality. These include woodstoves, pellet stoves, and fireplaces.

Woodstoves are free-standing appliances used as a source for space heating or to supplement an existing heating system (CCME, 2012). Typical woodstoves can hold a fuel load of 15 kg to 40 kg, which can result in an operation time of 4 to 12 hrs before requiring refuelling. Two main categories of woodstoves exist: conventional and advanced combustion types.

Conventional stoves tend to have relatively high smoke emissions and heat loss through a chimney system, while advanced combustion woodstoves generally have higher energy efficiency with lower particulate emissions (CCME, 2012). There are two sub-types of advanced combustion stoves: catalytic and non-catalytic stoves. A catalytic stove is equipped with a ceramic combustor coated with palladium (CCME, 2012). This acts as a catalyst and reduces the ignition temperature of the smoke so it can burn more completely all while cutting down on smoke emissions at normal stove-operating temperatures. On the other hand, a non-catalytic stove includes an insulated firebox, or an internal baffle to form a reflective surface, which separates the firebox from a secondary combustion chamber. This system allows pre-heating and a method to distribute a secondary air supply above the fuel bed source. In recent years, both sub-types have proved to be successful in lowering emissions, but the non-catalytic category has come to dominate the market.

Pellet Stoves burn wood or biomass pellet where sawdust or other waste biomass is compressed into small cylinders producing pellets (CCME, 2012; EPA, 2015). Steady burning is achieved in this system and allows for adjusting the fuel and air mixture. Pellet stoves can burn with lower emissions on average when compared to a dedicated woodstove. Exhaust in a pellet stove is forced into a vent by a fan.

Fireplaces are not strictly a decorative appliance as is often considered. Fireplaces have been manufactured with advanced heating technologies as well as heating ducts for use as a central heating source (CCME, 2012). Two broad categories of fireplaces exist: conventional fireplaces and advanced technology fireplaces (CCME, 2012).

Conventional fireplaces can be either masonry or factory built and do not incorporate any emission reduction technologies (CCME, 2012). Thus, they generally have higher emissions when compared to advanced technology fireplaces. Furthermore, they are not effective for home heating purposes, due to inefficient heat transfer characteristics as well as ineffective air flow issues (CCME, 2012). Advanced technology fireplaces are factory-built units which consist of emission reducing technologies and, thus, have low emissions (CCME, 2012). They can be used for home heating by using a series of ducts to distribute heat to different areas of a home.

2.1.2 Indoor Air Quality and Standards

With woodstoves as the major source of heating in Northern communities, the harmful emissions and exhaust from the burning of wood often lead to various health problems and serious respiratory issues (Government of Canada, 2015c). Generally, $PM_{2.5}$ and PM_{10} are two types of fine particles associated with wood smoke. These particles 2.5 and 10 microns in diameter, respectively, can be easily inhaled deep into the lungs causing difficulty breathing and various other cardiovascular damage (Government of Canada, 2015c). Figure 2-1 shows $PM_{2.5}$

emissions from residential wood combustion in 2010 in various provinces in Canada. $PM_{2.5}$ is a major cause of concern in households that utilize wood as their source of heating (CCME, 2012).
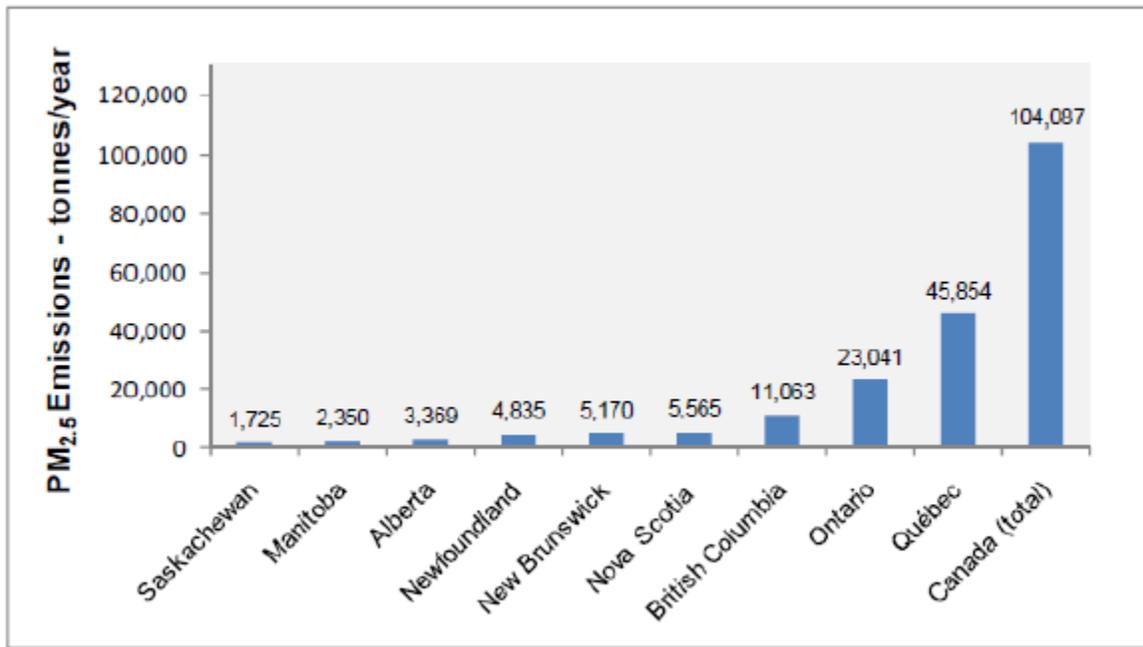
Figure 2.1. Measures for $PM_{2.5}$ Emissions in Various Provinces of Canada
from Residential Combustion of Wood in 2010 (CCME, 2012)

Authorities with the Government of Canada have provided residential indoor air quality guidelines for sampling and exposure to various contaminants (Government of Canada, 2015b).

In Canada, PM emissions for woodstoves and pellet stoves are set at 2.0 g/hr. For non-catalytic wood burning appliances the limit is 7.5 g/hr, while for catalytic wood burning appliances the limit is 4.1 g/hr. However, these standards have been revised and are more stringent when compared to EPA (Environmental Protection Agency) standards, since the EPA does not closely regulate indoor air (CCME, 2012).

Table 2.1 has been adapted from a paper by the CCME on guidelines for residential wood burning appliances and provides a comparison between EPA and CSA standards (CCME, 2012).

Table 2.1. Comparison of U.S. EPA and CSA Wood-Burning Appliance Particulate Matter (PM) Emission Limits (CCME, 2012).

Type of Device	US EPA Standard (2012 to 2016)	CSA Standard (2012 to 2015)
Catalytic wood burning appliances	4.1 g/hr (1988) 2.5 g/hr (2013)	2.5 g/hr
Non-catalytic wood burning appliances	7.5 g/hr (1988) 4.5 g/hr (2013)	4.5 g/hr
Factory built fireplaces	5.1 g/hr (2012)	Currently no limit
Masonry heaters	Currently no limit	Currently no limit
Masonry fireplaces	Currently no limit	Currently no limit
Indoor boilers and furnaces	0.32 lb/MMBtu (2014)	0.4g/MJ

Both manufacturers and consumers alike have been encouraged to consider new codes and practices for the use of wood-burning appliances (CCME, 2012). By meeting new stringent requirements, PM and VOCs have been estimated to drop by nearly 70% or 8,300 tons/year and 9,300 tons/year, respectively. The EPA has estimated the benefits of new residential heaters at $3.4 billion to $7.6 billion annually with costs estimated at $46 million. Also, these new codes and practices will yield $74 to $165 in benefits for every dollar spent, which includes avoidance of lost school and work days, emergency room visits, and hospital stays.

As a further precaution, the CSA and EPA have provided guidelines on the type of wood to be used in wood-burning appliances. This includes wood that is well seasoned and does not have high moisture content, as the majority of the heat goes into the evaporation of the water content within the wood instead of heating. Furthermore, green wood is not ideal as it has a high moisture content. Presently, woodstoves sold in the U.S. all require to have a permanent label

indicating they are EP5-certified and meet emission limits. This assists consumers in the decision-making process when it comes to the purchase of a wood burning stove.

The World Health Organization (WHO) European Centre for Environment and Health has also provided detailed guidelines regarding indoor air quality, particularly for selected pollutants. In regards to $PM_{2.5}$ healthy concentration levels, the WHO recommends a 24-hr average of no more than 0.025 μg/m³ and an annual average of no more than 0.010 μg/m³ (WHO, 2010).

2.2 The Use of Air Purifiers to Reduce the Concentration of Indoor Woodstove Smoke Particulate Matter $PM_{2.5}$

Air purifiers are used to reduce the indoor suspended particulate matter that is known to affect human health. The use of air purifiers has been steadily increasing. According to Shaughnessy and Sexto (2006), one in three U.S. households in 2006 had a portable air purifier. The key parameter characterizing an air purifier is its clean air delivery rate (CADR) (Mang, Walser, Nizkorodov, & Laux, 2009), defined as:

$$CADR \ = \ \gamma F \qquad\qquad\qquad [2\text{-}1]$$

where F is the air flow rate through the purifier (m^3/h of ft^3/min) and γ is the probability of capture for a given contaminant. For a plain fan that does not remove any air particulates, $\gamma=0$. For an air purifier that captures 100% of the particulates, $\gamma = 1$. For a high efficiency particulate air (HEPA) filter, $\gamma > 0.99$. However, Mang et al. (2009) point out that certain commercially available air purifiers emit ozone (O_3). These authors cite evidence by Bell, Peng, and Dominici,

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(2006) that exposure to ozone, even at low concentration is known to have detrimental health effects on humans, including elevated mortality, aggravation of asthma symptoms, and lung cancer in children. Nevertheless, research into developing effective indoor air purifiers continues unabated. Tests performed by the Association of Home Appliance Manufactures (AHAM) and reported in the CADR Ratings Guide (APG, 2010) show that the mean CADR value for dust (particle size 0.5-3.0 μm in diameter) was approximately 200 ft^3/min (see Figure 2.2). It can be concluded from the test results that 85% of the air purifiers had a CADR between 40 and 300 ft^3/min.

Shaughnessy and Sexto (2006) reviewed published documents on the effectiveness of portable air cleaning devices and reported that, for a large room (20' x 30' x 8'), an air cleaner with the highest CADR rating had a smoke removal effectiveness of 74%. Allen et al. (2011) reported that portable HEPA air filters reduced indoor concentrations of PM$_{2.5}$ by 60% and were associated with improved health risks. There was limited evidence in their study that suggests these benefits were more pronounced among residents in homes that burned wood.
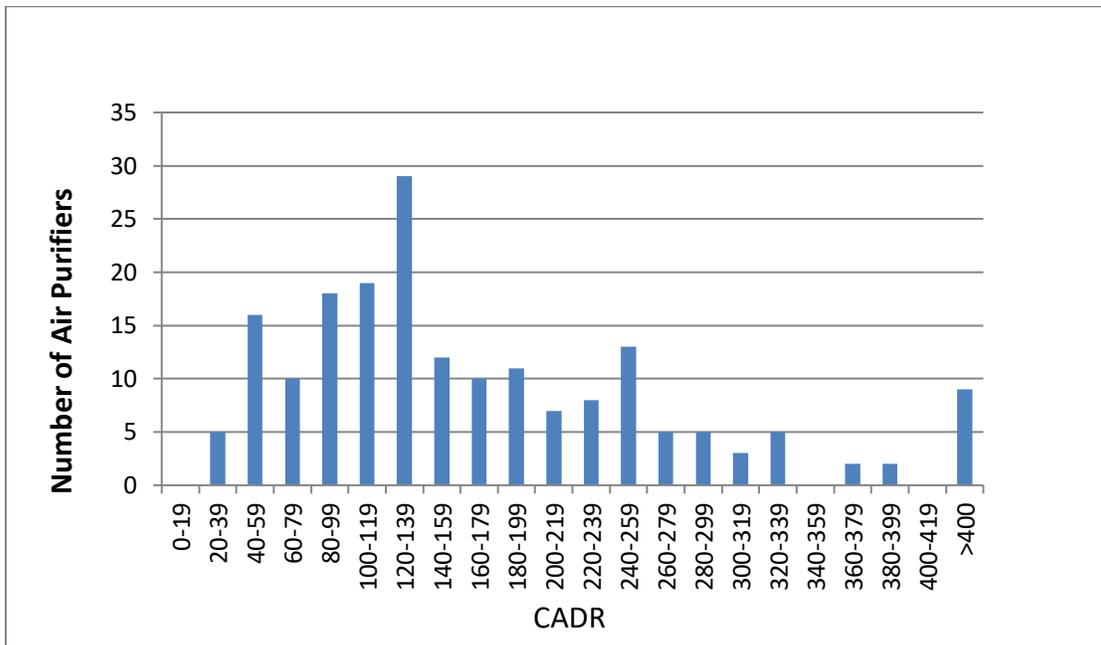
Figure 2.2. CADR Rating of 189 Air Purifiers for Dust (Data Source: APG, 2010)

The great majority of the published literature related to indoor levels of $PM_{2.5}$ deals primarily with ambient/outdoor $PM_{2.5}$ concentrations entering homes or business, as this poses unique challenges in polluted cities. Other sources of $PM_{2.5}$ are also discussed across a wide variety of published work, including tobacco smoking, smoke due to cooking, road dust, air pollution in the form of smog, and smoke from forest fires. In this thesis, the main focus is particulate levels in the form of $PM_{2.5}$ created from woodstoves used as the primary heating source in homes. When ambient levels of $PM_{2.5}$ are typically higher than those found indoors, such as those found in polluted cities in China, the primary strategy for mitigation is to ensure that windows and doors are properly sealed, not allowing $PM_{2.5}$ to enter the home.

Alternatively, when ambient levels of $PM_{2.5}$ are negligible, the $PM_{2.5}$ detected indoors is almost entirely due to residual particulate created from within the home. This was the case in the test home selected for this thesis. Ensuring proper window and door sealants, in this instance, would actually result in higher levels of $PM_{2.5}$ within the home. It is clear that there is no single solution for mitigating indoor $PM_{2.5}$. However, it is obvious that proper maintenance of the home can only benefit residents. In the case of an apartment in a polluted city in China, proper maintenance of window/door sealants will always result in cleaner indoor air. In contrast, in woodstove burning homes, proper maintenance of the woodstove will almost always result in cleaner air. Proper maintenance with respect to indoor air quality requires proper education on maintenance, which is something that residents may not have acquired nor always follow. Furthermore, once proper maintenance begins, consistency over time is not assured.

Identifying ways of controlling high concentrations of indoor $PM_{2.5}$ is essential to reducing the risks associated with respiratory illness, especially for those individuals most at risk – the elderly and young children. Mitigation strategies for $PM_{2.5}$ discussed in the literature include ensuring better ventilation strategies, using newer and cleaner pellet woodstoves, and introducing air filtration units. The literature shows that air purifiers can be quite effective in reducing $PM_{2.5}$ concentrations (Rice et al., 2018; Cheng et al., 2016; Polidori, Fine, White, & Kwon, 2013; Liu et al., 2017; Hart et al., 2011; McNamara et al., 2017).

A study conducted by Rice et al. (2018) showed that $PM_{2.5}$ concentrations were successfully reduced through utilization of air purifiers in combination with proper education. The study

included intervention along with pre-/during-/post-intervention data in conjunction with interviews. The homes in the study were occupied by either pregnant women or women with young infants as well as by at least one cigarette smoker. The educational intervention proved successful in reducing the tendency by residents to smoke indoors, indicating that resident behaviour can be positively influenced by proper education of indoor air quality (Rice et al., 2018).

Air purifiers were found to reduce average $PM_{2.5}$ concentrations in homes in Fresno, California, by an average of 96% (Cheng et al., 2016). Data were collected over a 12-week period in each home, with 5-min data logging intervals. An air purifier was utilized for only part of the entire test period in order to examine the effectiveness of such a device in reducing $PM_{2.5}$ concentration. The results indicated that although the average $PM_{2.5}$ concentrations dropped with the use of an air purifier, the magnitudes of the $PM_{2.5}$ spikes were not reduced (Cheng et al., 2016).

In a different study, Polidori et al. (2013) measured $PM_{2.5}$ concentrations in classrooms in Southern California over a four-week period. The effectiveness of implementing several $PM_{2.5}$ abatement strategies was also determined. First, a baseline $PM_{2.5}$ concentration was measured without the use of any HVAC or air purifier present. When a stand-alone air purifier was used with no HVAC system running, an overall reduction in indoor $PM_{2.5}$ of 90% was observed. When an HVAC system was active, in addition to use of an air purifier, this reduction in $PM_{2.5}$ was only slightly improved on average (between 87 – 96% reduction). This is understandable,

given the fact that fresh particulate is continuously circulated into the room through the HVAC system. Interestingly, the effectiveness of the HVAC alone was equivalent to reducing $PM_{2.5}$ to that of the stand-alone air purifier. Indoor concentrations of $PM_{2.5}$, in this case, were primarily due to biomass-burning pollution in the atmosphere, not entirely similar to a woodstove-burning home (Polidori et al., 2013).

When it comes to reducing high $PM_{2.5}$ concentrations utilizing air filtration techniques, in combination with increased air ventilation, can be very effective (Liu et al., 2017). The problem, however, with allowing for increased ventilation within a woodstove-burning home, by opening a window, is that the room temperature can sharply drop, which is counter to the purpose of using the woodstove. Additionally, thermal comfort is negatively impacted by an increased air velocity at cooler temperatures (Liu et al., 2017). Although better ventilation is a well utilized strategy in reducing $PM_{2.5}$ in many situations, it cannot be a reliable strategy in the case of many northern wood-burning homes where outdoor temperatures can reach dangerously low levels. Perhaps, a better strategy for managing indoor $PM_{2.5}$ levels is through better management and maintenance of woodstoves, including their proper ventilation. Ensuring that the chimney is properly cleaned and the ventilation openings kept open is also important.

In a study conducted by Hart et al. (2011), air purifiers were utilized in two woodstove-burning homes during a winter season. Sample durations for testing were 10 days for each home. Twenty tests were conducted, each 12-hrs long. $PM_{2.5}$ concentrations with an air purifier in use were, on average, 61-85% lower than $PM_{2.5}$ concentrations without an air purifier (Hart et al., 2011).

In a separate study by Klepeis et al. (2017), where airborne $PM_{2.5}$ concentrations were measured in low-income homes in San Diego, California, there was a strong correlation between the presence of cigarette/marijuana smoke within the home and mean airborne $PM_{2.5}$ concentrations. Additionally, an increase in the size of the home itself was strongly correlated with a decrease in $PM_{2.5}$. Interestingly, no strong correlation was found between $PM_{2.5}$ concentrations and an increase in reported ventilation activity, such as opening windows or turning on exhaust fans (Klepeis et al., 2017).

In another study dealing with the benefits of using an air purifier in woodstove-burning homes, 48 homes were chosen for a double-blind intervention (McNamara et al., 2017). In this study, a subset of homes was examined from the Asthma Randomized Trial of Indoor Wood Smoke (ARTIS) study. ARTIS is an intervention-based study designed to improve the quality of life of children with asthma living in woodstove homes by reducing in-home levels of $PM_{2.5}$ (Noonan & Ward, 2012). Real HEPA air purifiers were used in 23 homes, while placebo air purifiers without built-in HEPA air filtration were used in the remaining 25 homes. Comparisons between data from the homes, which have 60-min average $PM_{2.5}$ concentrations over a 48-hr testing period, yielded some interesting results. Homes utilizing the HEPA air purifiers reported, on average, 66% lower $PM_{2.5}$ concentrations than those homes utilizing the dummy air purifiers. The reason for using the placebo units was to minimize the effect of resident behavior on the data. This in turn, allowed for a more accurate representation of the effect of introducing a HEPA air purifier to reduce $PM_{2.5}$ concentrations in the home (McNamara et al., 2017).

Air filtration interventions that reduce indoor exposure to $PM_{2.5}$ have been the subject of a number of investigations (Offermann et al., 1984; Brauner et al., 2008; Barn et al., 2008; Hart et al., 2011; Zuraimi, Nilsson, & Magee, 2011; Weichenthal et al., 2013; Fisk and Chan, 2016; Barn et al., 2016).

Offermann et al. (1985) report on a study they conducted to evaluate the rate of decay of indoor concentration of respirable particles, such as tobacco smoke (with a median diameter of 0.5 μm), using various portable air cleaning devices. Their performance studies show a substantial variation in the abilities of various air cleaning devices to remove particles form indoor air. Hart et al. (2011) carried out a preliminary study to evaluate the effectiveness of a commercial portable air purifier in two homes with wood-burning stoves. Twenty 12-hr trials were conducted in each home. Ten trials were conducted with an air purifier and ten without. In Home A, where the woodstove was the primary source of heating, the occupants added wood to the stove every 2 to 3 hrs during the day and once at midnight. In Home B, where the woodstove was used as a secondary source for heating, the mass of wood burned was significantly lower than that in Home A. Reduction in particle count concentration in Home A ranged from 61% to 66% while in Home B, reduction in particle concentration was higher than Home A (78-86%).

Studies examining the effect of wood burning smoke on the health of First Nations residents in Canada is very limited. Weichenthal et al. (2013) conducted a study involving 37 residents in 20 homes in a First Nations community in Manitoba. Each home received an electrostatic filter and a placebo filter for one week. Indoor $PM_{2.5}$ in homes using an electrostatic filter decreased substantially relative to homes using a placebo filter (mean difference: 37 μg/m^3, 95% CI:

10.64). In general, the authors report that commercially available indoor air filters may substantially reduce the indoor levels of $PM_{2.5}$ and improve lung functions.

A similar randomized double-blind crossover trial of air purifiers was conducted by Chen et al. (2015) in China. They found that, on average, air purifiers reduced $PM_{2.5}$ concentrations by 57% from 96.2 to 41.3 $\mu g/m^3$ within hours of operation.

A number of studies have also addressed the health risks associated with smoke infiltration into homes near forest fires. The health benefits associated with filtration interventions that reduce indoor exposure to $PM_{2.5}$ were the subject of a recent study by Fisk and Chan (2016). Using analytical models, the authors concluded that a portable air cleaner reduced the predicted mean of $PM_{2.5}$ concentration by 45% in homes without forced HVAC systems. The authors also addressed the economic benefits associated with infiltration.

The infiltration factor of wood smoke from forest fires was also the subject of a study by Barn et al. (2008) who examined the HEPA filter air cleaners in reducing indoor levels of $PM_{2.5}$. While the authors were not able to use all the data obtained during their investigation, they concluded that air cleaners were found to be effective across a wide range of housing characteristics. In a more recent publication, Barn et al. (2016, p.1)) concluded that "portable air cleaners should be at the forefront of the public health response to landscape fire smoke."

Resident behavior directly impacts indoor air quality. Decisions that residents make in biomass-burning homes will inherently affect the $PM_{2.5}$ concentrations. In a study conducted in a village

in Tibet (Xiao et al., 2015), where yak dung is the primary biomass fuel utilized for heating homes as well as for cooking, $PM_{2.5}$ concentrations were measured over 6-hr periods. It was found that although some residents tried to ventilate their homes via makeshift chimneys, the chimneys in many cases proved inadequate in reducing $PM_{2.5}$ within the home. In this study, particulate concentrations were found to be higher before and after snow events, a time when moisture content in the yak dung was higher (Xiao et al., 2015).

Educational intervention programs, when implemented alone, are not always successful in reducing average indoor $PM_{2.5}$ concentrations in woodstove burning homes. In a study conducted in Alaska dealing primarily with correlating health risks in young children against IAQ indicators, such as $PM_{2.5}$ concentrations, an educational intervention was implemented guiding residents how to better ventilate the room, better maintain their woodstove, and properly manage the wood fire so as to not allow particulate leaks. $PM_{2.5}$ was monitored in the homes two weeks prior, two weeks after, and one year and two weeks after the intervention. Data were logged for a 24 to 96-hr duration test time in 60 homes. No significant drop in $PM_{2.5}$ was detected when comparing pre-intervention to the post-intervention concentrations despite fewer reported respiratory symptoms. In contrast, a significant drop in VOCs was detected. It should be noted that no air purifiers were used as part of the intervention, which was strictly educational in nature (Singleton et al., 2018).

In a separate study conducted by Chan et al., (2017), the effectiveness of air purifiers against rising $PM_{2.5}$ concentration in hotel rooms that allows cigarette smoking is weighed against the

effectiveness of increased room ventilation. Cigarette smoke and wood-burning smoke are similar in nature as they both are created from biomass combustion. Given the restricting dimensions of hotel rooms, a mere 36 square meters, $PM_{2.5}$ concentrations were extremely high even after a single cigarette was lit and smoked. Data was logged for 150-min sessions, during the burning stage and for the duration after that. $PM_{2.5}$ concentrations were detected to be at their highest level when no ventilation or air purifier was used. Concentrations rose to an average level of 0.586 mg/m³ during the burning process and as high as 1.386 mg/m³ during the test period. Ventilating the small room proved to be extremely helpful in reducing $PM_{2.5}$ concentration levels during the burning and smoke permeation stages to an average of 0.1 mg/m³. Without modifying room ventilation, however, the presence of the air purifier only helped reduce smoke concentrations, as $PM_{2.5}$ spikes remained high. Opening a window after the burning stage for 30 min yielded only partial reduction in average $PM_{2.5}$ concentrations. Levels remained at 70% even 70 min into the burn test with the increased room ventilation, indicating fresh air intake limitations due to the room orientation and window dimensions. Also, $PM_{2.5}$ appears to linger in the room for extended periods of time, even long after the cigarette had been put out. When both increased ventilation and use of an air purifier were utilized together, $PM_{2.5}$ concentrations rose from a pre-burn level of 0.045 mg/m³ to only 0.072 mg/m³. The post-burn smoke-permeation stage, on average, yielded an even lower concentration than the pre-burn stage: 0.032 mg/m³ (Chan et al., 2017).

It is clear that when ambient $PM_{2.5}$ levels are low and thermal comfort is not at risk, increased ventilation for the room is an excellent and cost-effective strategy in reducing harmful particulate counts indoors. Utilizing an air purifier in rooms with less ventilation potential or when outdoor

temperatures are too low to allow fresh air intake, would be most beneficial. The strategy chosen is one that best suits the unique lifestyles and needs of residents in their homes.

Woodstove change-out programs, where older woodstoves are replaced with modernized cleaner burning ones, have also been reported to be effective, in many cases reducing overall $PM_{2.5}$ concentrations in homes. In a study by Pope et al. (2017), results from 42 separate stove change-out program intervention studies were analyzed and compared. The data show that, on average, there is a clear reduction in $PM_{2.5}$ concentrations after the change-outs were completed. On average, $PM_{2.5}$ exposure by residents dropped approximately 55% across all studies. However, the authors report that the extent by which $PM_{2.5}$ dropped is not necessarily an accurate representation for the long-term effectiveness of the change-out program (Pope et al., 2017).

In a study conducted by Chowdhury et al. (2013) in a rural town in Yuyan China, older and less efficient cook stoves were swapped for newer cleaner burning cook stoves along with other new supplemental technologies such as solar water heaters, in 30 homes. Comparing $PM_{2.5}$ concentrations before-the-intervention to after-the-intervention $PM_{2.5}$ concentrations, there was a clear reduction in such concentrations by, approximately, 90%. It should be noted that changing older stoves/chimneys when accompanied by better ventilation of particulate to the outside results in a reduction of average indoor $PM_{2.5}$ concentration (Chowdhury et al., 2013).

The majority of the case studies examined in this literature review provide data taken shortly after the intervention and as such, they provide a best-case scenario. Because health effects are typically developed over a longer period of time, long-term changes in $PM_{2.5}$ concentrations must also be tracked to obtain a more complete picture of the effectiveness of any change-outs. This best-case scenario, however, in many instances is still not enough to guarantee safe $PM_{2.5}$ levels within the home. This is a further indication that the implementation of a woodstove change-out program alone is not sufficient in developing a strategy for a reduction of indoor $PM_{2.5}$ concentrations. Although effective in reducing overall $PM_{2.5}$ concentrations in the home, most if not all of these change-out programs are part of research studies or pilot programs. In reality, however, many residents in First Nation communities cannot afford new woodstoves or justify the cost of replacing their old ones. In some cases, a new clean burning woodstove could cost well over several thousand dollars. In a study conducted in a village in Mexico, a woodstove change-out program was implemented for 100 homes. In the long term, the new change-out program proved to have little or no overall benefit in terms of reduction in indoor $PM_{2.5}$, as residents began to utilize additional makeshift woodstoves within the home to supplement their newer, cleaner burning woodstoves (Ruiz-Mercado & Masera, 2015). The reason for this was strictly cultural, as the newer woodstoves could not fulfill their cooking needs. Resident comfort ultimately becomes a priority within the home. The potential benefits of utilizing air purifiers to reduce $PM_{2.5}$ concentrations in homes must also be considered.

How $PM_{2.5}$ concentrations due to wood smoke vary with height above floor level, at several distances away from the woodstove, has not been previously studied. Additionally, there is no evidence in the related literature of any study that examines how varying the duration of time the

woodstove door remains opened for refuelling stages affect the concentration of $PM_{2.5}$ levels. Nor, is there evidence of any similar research that examines the effect that air purifiers might have on the sustainability of $PM_{2.5}$ levels within the home. The absence of such evidence, thus, characterizes the uniqueness of the present study and its important contribution to the body of knowledge.

CHAPTER 3. EXPERIMENTAL INVESTIGATION

3.1   Background: Positional Spatial Analysis - $PM_{2.5}$ Particulate Dispersion in a Closed Space

Despite the fact that, under certain lighting conditions, wood smoke is visible to the naked eye, particulate matter in the form of $PM_{2.5}$ is mostly invisible. Additionally, $PM_{2.5}$ can remain airborne in an unventilated home for extended periods of time and long after any visible smoke dissipates.

This chapter addresses the question of how $PM_{2.5}$ levels are distributed in a closed space environment. Intuitively, it might be assumed that, as wood smoke permeates a closed single room environment, $PM_{2.5}$ levels would spread evenly in such a short time frame as to negate any spatial effects when $PM_{2.5}$ average values are determined over longer periods of time. Alternatively, one could assume $PM_{2.5}$ levels would be higher in the immediate vicinity of a heat source, such as a woodstove. Other parameters may also affect the permeation of wood smoke in a room, such as the height above floor level and distance from the woodstove. Smoke will generally rise because heat rises, but how does this translate to differences in detected $PM_{2.5}$ levels in a single room home? Keeping in mind that the harmful effect of particulate matter, in the form of $PM_{2.5}$, on respiratory health is well documented, this parameter is very critical. Because children might be restricted to a space that could in fact harbor higher $PM_{2.5}$ levels than what would typically be experienced by an adult, they can potentially be subjected to even greater risks. Also, children can be exposed to $PM_{2.5}$ levels differently than adults. For example, when children sleep in bunk beds, each child can potentially be exposed to different levels of

$PM_{2.5}$ concentrations sleeping at different heights above floor level for extended periods of time. Thus, these kinds of spatial postulations should be tested using synchronized measurements of $PM_{2.5}$ obtained from several $PM_{2.5}$ detection pieces of equipment placed at unique fixed heights and/or distances from the woodstove inside the home.

In recent years, introduction of newer, cleaner burning-woodstove technologies has been predominantly pushed as the go-to solution for reducing $PM_{2.5}$ concentrations in many communities across the world. This strategy can be an excellent long-term solution that is known to have been quite successful in several pilot studies (Noonan, Navidi, Sheppard, Palmer, Bergauff, Hooper, et al., 2012; Gómez, Chávez, Salgado, & Vásque, 2017; CARB, 2017). However, these new woodstoves are quite costly and beyond the affordable level of many communities. However, preventative measures can be taken by residents to reduce the risk associated with high $PM_{2.5}$ levels regardless of the type of woodstove used. By better understanding how $PM_{2.5}$ disperses and settles in a single-room home over short (1.5-hr) and long (24-hr) test periods, better woodstove practices as well as useful and cost effective preventative strategies or counter measures against high levels of $PM_{2.5}$, can be developed. With this in mind, the following questions are addressed in this chapter.

In a single-room house that utilizes a woodstove as its primary heat source:

1. How does the $PM_{2.5}$ concentration vary with distance from the woodstove?

2. How does the $PM_{2.5}$ concentration vary with height above floor level while the woodstove is fired up?

By answering these questions, a clearer understanding of how positional differences affect $PM_{2.5}$ concentrations can be observed. However, positional factors do not provide a complete picture of the potential risk associated with wood smoke concentration. If high levels of $PM_{2.5}$ are allowed to enter the home due to poor maintenance of the woodstove, residents will one way or another be subjected to unnecessary pollution.

3.1.1 Residents' Actions (Directly and/or Indirectly Related to Woodstove Use) that Could Affect $PM_{2.5}$ Concentration Levels at Home

A resident with a woodstove in operation is exposed to different $PM_{2.5}$ levels throughout the day while at home. A resident's actions directly related to woodstove use within the home could have significant effect on $PM_{2.5}$ levels in the home. These include:

a) Duration of time the woodstove is in use during the day,

b) Frequency of opening the woodstove door,

c) Duration of time the woodstove door remains open during refueling,

d) Use of firewood and kindling with substantial moisture content, and

e) Operation of a woodstove that is poorly maintained.

In addition to the residents' actions listed above, a number of other factors, which are indirectly related to woodstove use within the home, could also affect how $PM_{2.5}$ is experienced by residents. For example:

a) the amount of time residents stay indoors,

b) the volume of air generally inhaled per min - lung capacity,

c) resident's position with respect to the woodstove,

d) the height above floor level of a resident's mouth during the time the woodstove is in operation,

e) the level of thermal comfort of residents (temperature and relative humidity indoors/outdoors),

f) type of ventilation (windows open or closed),

g) use of the front door while the woodstove is in operation (door open/closed), and

h) use of air purifiers.

Furthermore, some residents' actions, not directly related to woodstove use, can also affect the degree of exposure to $PM_{2.5}$ particulates in a home. For example, smoking by residents inside the home can increase the $PM_{2.5}$ level substantially. In addition, the type of cooking preferences and whether the woodstove is also used as a cook stove could affect $PM_{2.5}$ levels (Hu et al., 2012).

It should be noted that the amount of $PM_{2.5}$ being absorbed by a person's lungs varies from person to person. This is, in part, due to biochemistry variabilities among residents. The position of a child's mouth/nose, for instance, will be different than that of an adult and thus will have exposure to different $PM_{2.5}$ levels. This is the reasoning behind the choice of the different heights used in the testing program of this study. Residents will have their mouth and nose positioned at different heights within the home throughout the day, depending on their chosen behaviour. Residents can be standing, sleeping, or sitting, each of which will mean a different breathing

position. Also, height of nasal and mouth breathing passages will depend on the age of residents as well, as will their chosen behaviours. Because children and elderly are inherently at greater risks of respiratory illness caused by particulate matter absorption, no one numerical value of $PM_{2.5}$ can consistently characterize a hazard for all. In this sense, standards such as those developed by EPA can only be used as a guide (EPA, 2016).

Even if an ideal location in the home is found to have the least $PM_{2.5}$ concentration, a significantly greater amount of particulate can build up through woodstove misuse. Thus, any positional recommendations alone would be misleading. Resident behaviour parameters that are indirectly associated with woodstove use could outweigh parameters directly related to woodstove use in the way they affect $PM_{2.5}$ levels absorbed by residents, or vice versa. Because constant monitoring of resident behaviour is outside of the scope of this study, the only behavioural testing done was for spatial parameters which are discussed in this section (resident mouth/nose position: distance/height), as well as the behaviour regarding the duration of woodstove door is kept open during refueling, discussed in Sections 3.3.4 and 3.3.5. Once the data at different heights above floor level and distances from the woodstove are compiled, some preliminary conclusions regarding resident behaviour, indirectly related to woodstove use, can be made with implications for potential behavioral changes. No direct monitoring of any residents' behaviour was made during this project.

In order to begin developing a strategy to help mitigate high $PM_{2.5}$ levels, the effect of several residents' actions will be examined. If high smoke levels are allowed to permeate the home for

any reason, residents could choose to mitigate the amount inhaled by better positioning themselves in the home during the burn cycle. If, however, the smoke levels (and thus the $PM_{2.5}$ levels) entering the home are reduced, through changes made in resident behaviour directly related to the woodstove's use (such as time the woodstove door is left open), the indoor air quality can be improved and the need for changes in resident position can be minimized. Of course, mitigation of indoor particulate becomes increasingly important when high levels of $PM_{2.5}$ are produced. It is up to residents to enact change in their home, whether it be positional or behavioral, and ultimately it is through action that these concentrations be avoided.

3.2   Experimental Investigation

Tests were conducted in the present study to measure the effect of $PM_{2.5}$ concentrations, as defined in Chapter 1. More specifically, to assess the degree to which residents are exposed to $PM_{2.5}$ levels in the home, answers to the following questions were sought:

a) How does the $PM_{2.5}$ concentration vary with **distance** from the woodstove, measured normal to the woodstove and at several heights (1', 2', 3', 4', 5', and 6' above the floor level), while the woodstove is fired up and in operation for a 90-min period?

b) How does the $PM_{2.5}$ concentration vary with **height** above floor level and at fixed distances (3', 8', 13', and 18') from the woodstove, while the woodstove is fired up and in use over a 90-min period?

In order to assess the potential impact that resident behaviour, with respect to the woodstove's use, will have on $PM_{2.5}$ concentrations, answers to the following question was sought:

c) How does varying the amount of time the woodstove door remains open during the refueling process affect indoor $PM_{2.5}$ concentration levels?

Other factors affecting $PM_{2.5}$ permeation include moisture content of the wood/kindling and the total time the woodstove is in use. These will be discussed later in the chapter as their effect was not measured quantitatively. Nevertheless, their effect was at times noticeable during testing. In this sense, these will be discussed qualitatively.

3.2.1 Procedures in Determining Indoor Levels of $PM_{2.5}$

In order to determine how spatial parameters (Questions (a) and (b) above) as well as how varying the duration of time the woodstove door remained open during refueling (Question (c) above) will affect the air quality, as measured in $PM_{2.5}$ concentration levels in the home, two different test procedures were used. National air quality standards typically focus on 24-hr average data. Thus, a more complete examination will be made when discussing $PM_{2.5}$ concentrations by viewing the data under the two different test procedures:

1. Real-time data directly measured by Dusttrak II instruments with a built-in data logging interval of 2 seconds. These data will be characterized by noticeable sharp peaks and valleys in $PM_{2.5}$ concentrations over the duration of the test period.

2. Synthesized 24-hr average data obtained from 1.5-hr test period real-time data, which resembles real-life woodstove use.

The key objective in this study was to examine how particulate matter in the form of $PM_{2.5}$ can permeate a single-room rural home with a woodstove burning. Additionally, this study sought to determine the feasibility of using air purifiers to reduce the $PM_{2.5}$ concentration levels in the home. The discussion in this study will focus on the 24-hr measurement values (Procedure (b) above), as this is the scale in which health standards are determined (EPA, 2016). Live data (Procedure (a) above) is also of interest in this study in order to understand the nature of $PM_{2.5}$ as it disperses within the home. While discussing dispersion, settling, or air purification of particulate matter, this study will additionally include discussion of real-time data for the purposes of measuring effectiveness in real time. In the end, it is the live data that are used to synthesize the 24-hr data. The limitation in this procedure is that the $PM_{2.5}$ average from a 90-min burn cycle will only approximate the $PM_{2.5}$ average from a 24-hr burn cycle.

It is not uncommon in most northern communities to have single-room homes with woodstoves as the sole source of heating. Residents in those homes staying inside during the long cold winters could be exposed continuously to high levels of $PM_{2.5}$.

The home used during testing was at one time a principal residence of a family where children and adults lived, indicating that, during that time, they all had been exposed to particulate levels in line with those observed. It is for this reason, that height testing was included as children are

exposed to levels that are introduced at different heights. Along with heights, concentration levels at different distances from the woodstove were measured, as described earlier. These spatial tests may not only provide useful information for possible particulate concentration alleviation strategies for individuals, but also can help lead to the development of a best practice method for detection of $PM_{2.5}$ in a woodstove burning home. By understanding how $PM_{2.5}$ levels vary inside a single home, a best-practice method would help to maximize consistency for comparison between different homes in potential broader studies in the future.

3.3   Positional/Locational Testing

Children and adults living in a wood burning house are exposed to different $PM_{2.5}$ levels throughout the day. The level of exposure to wood smoke will depend on the rate in which air is inhaled and the position of the residents in the home. By measuring $PM_{2.5}$ concentrations simultaneously at several different locations during a single burn, we can assess the level of exposure by different residents.

In this section, the test set-up is described for the various tests conducted. These include: (a) tests conducted to determine $PM_{2.5}$ distribution over height above floor level; (b) tests conducted to determine $PM_{2.5}$ distribution over distance from woodstove; (c) tests conducted to determine $PM_{2.5}$ distribution during various durations of time the woodstove door was kept open without the use of an air purifier; (d) tests conducted to determine $PM_{2.5}$ distribution during various durations of time the woodstove door was kept open for refueling, with the use of an air purifier;

and, (e) tests conducted to determine dissipation of $PM_{2.5}$ concentration with the use various types of air purifiers.

The equipment used to monitor $PM_{2.5}$ consisted of four Dusttrak II instruments. These instruments were positioned at various distances from the woodstove and at various heights above floor level. All distances were measured perpendicular to the woodstove orientation and perpendicular to the floor, which we determined to be level to within a quarter inch using a carpenter's level. All Dusttrak II instruments were used with a $PM_{2.5}$ impactor which was placed at the nozzle of the instrument to prevent 50% of particles larger than $PM_{2.5}$ from entering the unit. Currently, there is no perfectly accurate measurement tool for $PM_{2.5}$. The best that can be done, according to the manufacturer, TSI, is to estimate it using this 50% reduction.

Each experimental test set-up is described in the following sections.

3.3.1 Tests Conducted to Determine $PM_{2.5}$ Distribution over Height above Floor Level

$PM_{2.5}$ concentrations were measured, in mg/m^3, using four Dusttrak II instruments, positioned at different heights above floor level, as shown in Figure 3.1. These locations represent, roughly, the position of a person's mouth and nose inhaling wood smoke when the woodstove is on (resident behavior). The complete list of the tests and instruments used is given in Table 3.1.

Figure 3.1. Location of Instruments used to Monitor PM$_{2.5}$ Distribution
at Various Heights above Floor Level, 3' from the Woodstove

Table 3.1. Tests Conducted to Determine PM$_{2.5}$ Distribution over Height Above Floor Level

Test #	Distance from Woodstove (ft-in)	Height Above Floor Level (ft-in)	Instrument Used
H1	3-0	2-5.5	Dusttrak II #4
H1	3-0	3-5.5	Dusttrak II #2
H1	3-0	4-5.5	Dusttrak II #6
H1	3-0	5-5.5	Dusttrak II #5
H2	8-0	2-5.5	Dusttrak II #4
H2	8-0	3-5.5	Dusttrak II #2
H2	8-0	4-5.5	Dusttrak II #6
H2	8-0	5-5.5	Dusttrak II #5
H3	13-0	2-5.5	Dusttrak II #4
H3	13-0	3-5.5	Dusttrak II #2
H3	13-0	4-5.5	Dusttrak II #6
H3	13-0	5-5.5	Dusttrak II #5
H4	18-0	2-5.5	Dusttrak II #4
H4	18-0	3-5.5	Dusttrak II #2
H4	18-0	4-5.5	Dusttrak II #6
H4	18-0	5-5.5	Dusttrak II #5

Each test was conducted over a period of 90 min with a 2s data logging interval.

As explained earlier (Section 3.1.1), the reasoning behind why these different heights were used is that residents will have their mouth and nose positioned differently within the home throughout the day, depending on their chosen behaviour. The range of heights between 2'-5.5" and 5'-5.5" will cover almost all possible mouth positions for residents of all ages.

3.3.2 Tests Conducted to Determine PM$_{2.5}$ Distribution over Distance from Woodstove

PM$_{2.5}$ concentrations were also measured, in mg/m^3, using four Dusttrak II instruments, positioned at different distances from the woodstove, as shown in Figure 3.2. The complete list of the tests and the instruments used is given in Table 3.2.

Figure 3.2. Location of Instruments Used to Monitor PM$_{2.5}$ Distribution at 4'-5.5" Above Floor Level and at Various Distances from the Woodstove

Table 3.2. Tests Conducted to Determine PM$_{2.5}$ Distribution over Distance from Woodstove

Test #	Distance from Woodstove (ft-in)	Height Above Floor level (ft-in)	Instrument Used
D1	3-0	1-5.5	Dusttrak II #4
	8-0	1-5.5	Dusttrak II #5
	13-0	1-5.5	Dusttrak II #6
	18-0	1-5.5	Dusttrak II #2
D2	3-0	2-5.5	Dusttrak II #4
	8-0	2-5.5	Dusttrak II #5
	13-0	2-5.5	Dusttrak II #6
	18-0	2-5.5	Dusttrak II #2
D3	3-0	3-5.5	Dusttrak II #4
	8-0	3-5.5	Dusttrak II #5
	13-0	3-5.5	Dusttrak II #6
	18-0	3-5.5	Dusttrak II #2
D4	3-0	4-5.5	Dusttrak II #4
	8-0	4-5.5	Dusttrak II #5
	13-0	4-5.5	Dusttrak II #6
	18-0	4-5.5	Dusttrak II #2
D5	3-0	5-5.5	Dusttrak II #4
	8-0	5-5.5	Dusttrak II #5
	13-0	5-5.5	Dusttrak II #6
	18-0	5-5.5	Dusttrak II #2
D6	3-0	6-5.5	Dusttrak II #4
	8-0	6-5.5	Dusttrak II #5
	13-0	6-5.5	Dusttrak II #6
	18-0	6-5.5	Dusttrak II #2

3.3.3 Tests Conducted to Determine Reliability of Equipment

Tests were conducted to determine the reliability of cheaper and more readily available equipment than Dusttrak II instruments. These are listed in Table 3.3.

The PM$_{2.5}$ equipment was placed 10' from the woodstove on a large flat table and as close to each other as possible. The maximum spacing of the equipment from each other is listed in Table

3.3. One-minute average data logging intervals were used to eliminate any possible random error between instruments.

Figure 3.3. Tests Conducted to Determine Reliability of Equipment

Table 3.3. Tests Conducted to Determine Reliability of Equipment

Test #	Equipment	Distance Above Floor Level (ft-in)	Distance from Woodstove (ft-in)	Maximum Spacing from Each Other (in)

R1	Group 1:
	  Dusttrak II #1
	  Dusttrak II #4
	  Dusttrak II #6
	  Dusttrak II #2
	
	Group 2:
	  Dylos DC1700
	
	Group 3:
	  UNI-T 936A
		3'-0"	10'-0"	Among Group 1 Equipment: 6"
				
				Between Group 1 and Group 2: 18"
				
				Between Group 1 and Group 3: 18"

For an identical burn test, the following correlations between instruments were examined:

- Between the four Dusttrak II instruments measuring $PM_{2.5}$ concentrations, in mg/m^3, over a period of 3 hrs at 1-min logging interval.

- Between the Dusttrak II instruments and the Dylos DC1700 over a period of 3 hrs at 1-min logging interval.

- Between the Dusttrak II instruments and the UNI-T 938C over a 3-hr test period, at 5s interval, which must be manually adjusted in order to determine the equivalent 1-min logging interval data.

The results and analysis for this section are laid out in Appendix B.

3.3.4 Tests Conducted to Determine $PM_{2.5}$ Distribution at Various Durations of Time the Woodstove Door Was Kept Open Without the Use of an Air Purifier

These tests were conducted to determine what effect the duration that the woodstove door remained open during refueling has on $PM_{2.5}$ concentration. During each burn cycle test, which lasted 60 min, the woodstove door was opened every 20 min for refueling, each lasting between 5s and 45s, as shown in Table 3.4. Test D4-N lasted 90 min with a refueling period lasting 45s every 20 min.

Table 3.4. Tests Conducted to Determine PM$_{2.5}$ Distribution
at Various Durations of Time that the Woodstove Door was Kept Open
Without the Use of an Air Purifier

Test #	Distance from Woodstove (ft-in)	Distance above Floor Level (ft-in)	Instrument Used	Duration Woodstove Door was Kept Open for Refueling
D1-N	3-0	3-5.5	Dusttrak II #4	5s every 20 min
	8-0	3-5.5	Dusttrak II #5	
	13-0	3-5.5	Dusttrak II #6	
	18-0	3-5.5	Dusttrak II #2	
D2-N	3-0	3-5.5	Dusttrak II #4	15s every 20 min
	8-0	3-5.5	Dusttrak II #5	
	13-0	3-5.5	Dusttrak II #6	
	18-0	3-5.5	Dusttrak II #2	
D3-N	3-0	3-5.5	Dusttrak II #4	30s every 20 min
	8-0	3-5.5	Dusttrak II #5	
	13-0	3-5.5	Dusttrak II #6	
	18-0	3-5.5	Dusttrak II #2	
D4-N	3-0	3-5.5	Dusttrak II #4	45s every 20 min
	8-0	3-5.5	Dusttrak II #5	
	13-0	3-5.5	Dusttrak II #6	
	18-0	3-5.5	Dusttrak II #2	

The tests were conducted using four Dusttrak II instruments (#4, #5, #6, and #2) over a period of 1 hr, with 2s data logging intervals. All four tests were conducted without the use of an air purifier.

3.3.5 Tests Conducted to Determine PM$_{2.5}$ Distribution at Various Durations of Time the Woodstove Door was Kept Open and With the Use of an Air Purifier

These tests were identical to those discussed in Section 3.3.4 except that an air purifier was used to examine the dissipation of PM$_{2.5}$ during various durations of refueling. The air purifier used in these tests was a BluAir 503 positioned at a distance of 23' from the woodstove, as shown in

Figure 3.4. The tests conducted are listed in Table 3.5. As in the case of Test D4-N, Test D4-W

lasted 90 min.

Figure 3.4. Set-Up Used to Determine PM$_{2.5}$ Distribution
at Various Durations of Time that the Woodstove Door Was Kept Open
and with the Use of an Air Purifier

Table 3.5. Tests Conducted to Determine PM$_{2.5}$ Distribution at Various Durations of Time that the Woodstove Door Was Kept Open <u>with</u> the Use of An Air Purifier

Test #	Distance from Woodstove (ft-in)	Distance Above Floor Level (ft-in)	Instrument Used	Duration Woodstove Door was Kept Open
D1-W	3-0	3-5.5	Dusttrak II #4	5s every 20 min
	8-0	3-5.5	Dusttrak II #5	
	13-0	3-5.5	Dusttrak II #6	
	18-0	3-5.5	Dusttrak II #2	
D2-W	3-0	3-5.5	Dusttrak II #4	15s every 20 min
	8-0	3-5.5	Dusttrak II #5	
	13-0	3-5.5	Dusttrak II #6	
	18-0	3-5.5	Dusttrak II #2	
D3-W	3-0	3-5.5	Dusttrak II #4	30s every 20 min
	8-0	3-5.5	Dusttrak II #5	
	13-0	3-5.5	Dusttrak II #6	
	18-0	3-5.5	Dusttrak II #2	
D4-W	3-0	3-5.5	Dusttrak II #4	45s every 20 min
	8-0	3-5.5	Dusttrak II #5	
	13-0	3-5.5	Dusttrak II #6	
	18-0	3-5.5	Dusttrak II #2	

3.3.6 Tests Conducted to Determine Dissipation of PM$_{2.5}$ Concentration with the Use of Various Types of Air Purifiers

Several tests were conducted with the use of various types of air purifiers to determine their effectiveness in removing particulate matter from a closed space while the woodstove was in operation. Each test was conducted with a different air purifier. Four Dusttrak II 8530 devices were used to monitor PM$_{2.5}$ concentrations. These are labeled Dusttrak II #4, #5, #6 and #2 in Figure 3.5 and were positioned as shown in Figure 3.6. Each test lasted 12 hrs and data were recorded every 5s. Using the woodstove, the air was first saturated with a high level of wood smoke particulate and then the air purifier was turned on. The air purifier was placed in line with the Dusttrak II devices, as shown in Figures 3.5 and 3.6. Each Dusttrak II device recorded

14,400 readings of $PM_{2.5}$ concentration in mg/m^3. An impactor was used with the Dusttrak II devices to more accurately determine $PM_{2.5}$ concentrations, by allowing the system to distinguish these concentrations from larger particulate, such as PM_{10}.
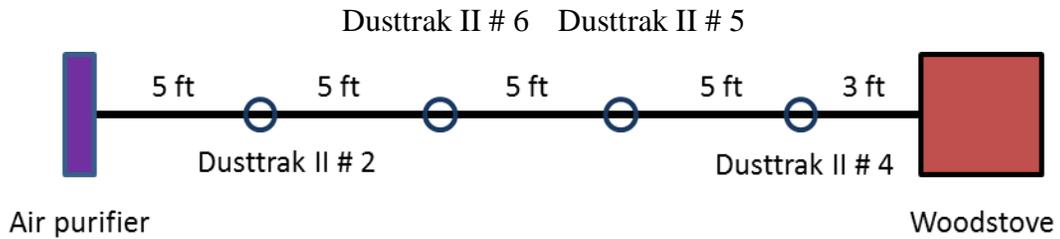
Figure 3.5. Position of both the Air Purifier and Dusttrak II Devices
with Respect to the Location of the Woodstove

Figure 3.6. Testing of IAQ Using Various Air Purifiers

The method of air filtration and the cost of these purifiers, labeled A through I, are shown in Table 3.6. The purifiers used are not identified by their trade name, but are commercially available and were purchased in the open market. Tests were also conducted without the use of an air purifier. Many of the air purifiers used also have proprietary pre-filters installed. The total number of filters in each of the purifiers range from one to five.

Table 3.6.   Purifier, Method of Air Filtration, and Price

Air Purifier	Method of Air Filtration	Price (CAD)
A	Carbon + HEPA filter	$700.00
B	HEPA filter + Ionization	$70.00
C	Carbon + HEPA filter	$300.00
D	Carbon Filter + Ionization + Ozone Generation	$200.00
E	Carbon + HEPA	$170.00
F	Carbon + HEPA filter	$300.00
G	Carbon + HEPA filter	$280.00
H	Carbon + HEPA filter + Toxin remover filter	$730.00
I	Carbon + HEPA filter + Odour remover filter	$730.00

After the woodstove was lit and the smoke allowed to permeate the closed space by leaving the woodstove door open for several minutes, and after the $PM_{2.5}$ level reached at least 0.5 mg/m³ on the Dusttrack II instruments, the woodstove door was closed for the entire duration of the test. Once $PM_{2.5}$ levels reached a minimum of 0.5 mg/m³ (and in some cases much higher), the air purifier was turned on and the research team left the room for the duration of the test, which lasted 12 hrs. The tests conducted with the use of an air purifier are listed in Table 3.7.

Table 3.7. Tests Conducted to Determine the Dissipation of PM $_{2.5}$ Concentration
with the Use Various Types of Air Purifiers

Test #	Distance from Woodstove (ft-in)	Height Above Floor Level (ft-in)	Instrument Used	Air Purifier Used
P1	3-0	3-5.5	Dusttrak II #4	A
	8-0	3-5.5	Dusttrak II #5	
	13-0	3-5.5	Dusttrak II #6	
	18-0	3-5.5	Dusttrak II #2	
P2	3-0	3-5.5	Dusttrak II #4	B
	8-0	3-5.5	Dusttrak II #5	
	13-0	3-5.5	Dusttrak II #6	
	18-0	3-5.5	Dusttrak II #2	
P3	3-0	3-5.5	Dusttrak II #4	C
	8-0	3-5.5	Dusttrak II #5	
	13-0	3-5.5	Dusttrak II #6	
	18-0	3-5.5	Dusttrak II #2	
P4	3-0	3-5.5	Dusttrak II #4	D
	8-0	3-5.5	Dusttrak II #5	
	13-0	3-5.5	Dusttrak II #6	
	18-0	3-5.5	Dusttrak II #2	
P5	3-0	3-5.5	Dusttrak II #4	E
	8-0	3-5.5	Dusttrak II #5	
	13-0	3-5.5	Dusttrak II #6	
	18-0	3-5.5	Dusttrak II #2	
P6	3-0	3-5.5	Dusttrak II #4	F
	8-0	3-5.5	Dusttrak II #5	
	13-0	3-5.5	Dusttrak II #6	
	18-0	3-5.5	Dusttrak II #2	
P7	3-0	3-5.5	Dusttrak II #4	G
	8-0	3-5.5	Dusttrak II #5	
	13-0	3-5.5	Dusttrak II #6	
	18-0	3-5.5	Dusttrak II #2	
P8	3-0	3-5.5	Dusttrak II #4	H
	8-0	3-5.5	Dusttrak II #5	
	13-0	3-5.5	Dusttrak II #6	
	18-0	3-5.5	Dusttrak II #2	
P9	3-0	3-5.5	Dusttrak II #4	I
	8-0	3-5.5	Dusttrak II #5	
	13-0	3-5.5	Dusttrak II #6	
	18-0	3-5.5	Dusttrak II #2	

Because drifting in the Dusttraks II' internal airflow may cause some error in its detection of $PM_{2.5}$, airflows were measured before and after each test for both Dusttrak II devices. This was to ensure the error correlates to less than the 5% equipment error value provided by the manufacturer.

It should be noted that all windows and doors were closed during testing. A control test was also done without the use of an air purifier.

CHAPTER 4. RESULTS FROM THE EXPERIMENTAL STUDY

4.1. PM$_{2.5}$ Concentration as a Function of Height above Floor Level

In this study, four individual burn cycles were conducted to measure differences in PM$_{2.5}$ concentrations at various heights above floor level, as shown in Table 3.1. Each burn cycle corresponds to different distances measured perpendicular to the woodstove. Data from one of the burn-cycles recorded at various heights above floor level at a 18' distance from the woodstove are shown in Figure 4.1
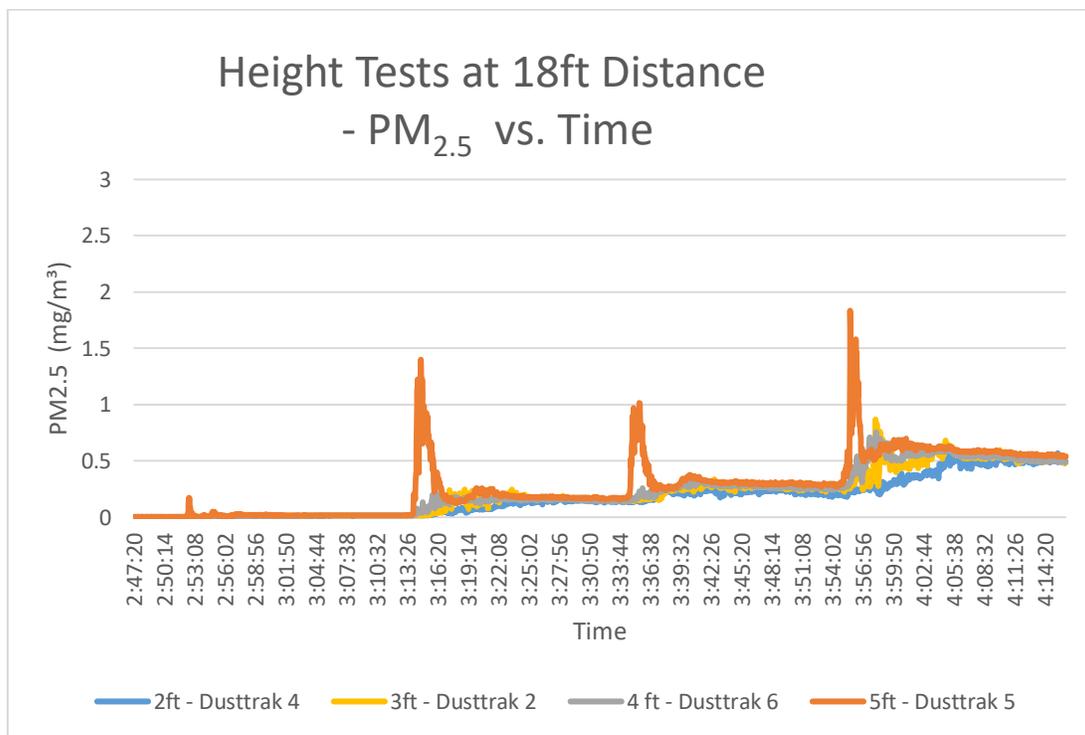
Figure 4.1. PM$_{2.5}$ Concentration as a Function of Time Measured at Various Heights Above Floor Level at 18ft. from the Woodstove.

(The actual position of the nozzle of the instruments was 5.5" above
the position of the actual instrument location shown in the figure.)

The spikes in PM$_{2.5}$ shown in Figure 4.1 occur shortly after the woodstove was lit for the first time and shortly after the door was reopened for refueling, which was every 20 min. A similar pattern was observed in all four burn cycles. Higher PM$_{2.5}$ concentration spikes were observed by instruments at higher positions, as demonstrated by the results in Figure 4.1. Concentrations at the lowest position of the instrument (Dusttrak II #4) did not spike. Instead a steady rise was observed. Also, the spikes in PM$_{2.5}$ seem to occur at the same time at the various heights. Interestingly, PM$_{2.5}$ concentration levels typically level off to an identical magnitude after about twenty min from the fueling/refueling time.

4.2   PM$_{2.5}$ Concentrations Due to Changes in Distance from the Woodstove

A series of tests were conducted to assess how the PM$_{2.5}$ concentration varies with distance away from the woodstove. These are listed in Table 3.3. A separate set of burn cycles was used for this part of the study. In total, six data sets from six individual burn cycles were logged. During each burn cycle, PM$_{2.5}$ concentrations were measured by four Dusttrak II 8530 instruments positioned at various heights, as described in the previous chapter. At each of the six burn cycles, the instruments were placed at four different distances from the woodstove: 3', 8', 13', and 18'.

Determining how distance from the woodstove can affect PM$_{2.5}$ concentration is of interest because it, again, adds a new layer in further understanding how residents may experience higher PM$_{2.5}$ concentrations when moving around the house. Depending on their proximity to the

woodstove, or even if they are responsible for fueling or refueling the fire, residents may be exposed to different levels of $PM_{2.5}$. Ultimately, by comparing the changes in average $PM_{2.5}$ concentration levels due to spatial parameters height and distance away from the woodstove, a clearer quantification can be made of how different counter measures, such as the use of air purifiers, may be used. Understanding the scale of the counter measure effects will help develop more effective strategies in reducing any potential health risks associated with wood smoke.

An example data set of $PM_{2.5}$ concentration at a height of 3'- 5.5" at distances of 3', 8', 13', and 18' from the woodstove is shown in Figure 4.2.
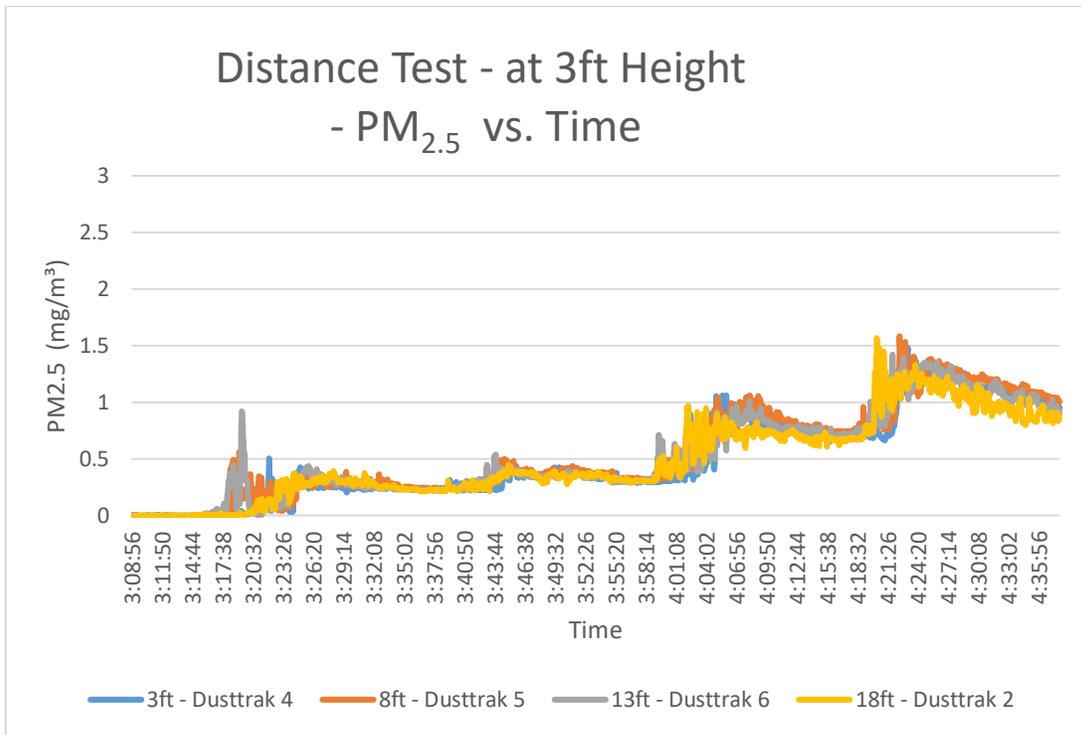
Figure 4.2. $PM_{2.5}$ Concentration Measured at a 3'- 5.5" Height above the Floor Level over a 90-min Burn Cycle (Peaks in Concentration are Caused by Refueling Every 20 Min)

Over the course of the burn cycle, each Dusttrak II instrument detected rising $PM_{2.5}$ concentrations at different times. It appears as if the particulate matter rises and travels along the ceiling and then curls back down upon reaching the back of the room. Dusttrak #2, located the furthest from the woodstove, at 18', detected rising particulate earlier than the other Dusttrak instruments closer to the fire. It is clear that this phenomenon will have an affect only when observing the data on a second-to-second basis and not over an extended period of time where data are averaged over the entire test period.

4.3 $PM_{2.5}$ Concentration as a Function of Woodstove Door Opening for Refueling

While using a woodstove, several resident behaviours come into play. Whoever is responsible for starting and maintaining the fire will have to open the woodstove door regularly in order to refuel. The time increment between refueling stages depends a great deal on factors such as wood moisture content, size of wood, kindling arrangement, air flow, etc. Testing the effects of these parameters is outside the scope of the present study. However, the refueling increment was fixed at 20 min. Most burn cycles lasted approximately 60 min.

In terms of its effect on indoor $PM_{2.5}$ concentrations the parameter of interest in this study was the time the woodstove door was left open during each refueling cycle. Because in a real-world environment the amount of time the woodstove door is left open can vary significantly, the periods chosen in this study were: 5s, 15s, 30s, and 45s.

The main underlying question in this section of the study is whether $PM_{2.5}$ concentrations are sustainable in the home or not. By establishing the particulate sustainability, it can be determined whether newly released $PM_{2.5}$ in the home is cumulative or the amount dissipates over time. By comparing the sustainability trend of $PM_{2.5}$ between different burn cycle data, conclusions can be drawn regarding the effect of the duration of time the woodstove door remains opened during refueling. It is critical to understand whether, given the limitation of a fixed 20-min refueling interval, $PM_{2.5}$ levels continue to rise, level off, or decrease towards safe levels over a 24-hr burn cycle. The data used for this purpose are based on eight burn cycles, as shown in Tables 3.4 and 3.5. Four of the tests (Table 3.4) were conducted without the use of an air purifier in order to determine the degree of $PM_{2.5}$ accumulation over successive fueling/refueling stages. The other four tests (Table 3.5) were conducted with the use of an air purifier in order to determine the degree of $PM_{2.5}$ dissipation over successive stages of fueling/refueling of the woodstove.

4.3.1 Effect of Duration of Time the Woodstove Remained Open During Refueling Without the Use of an Air Purifier

Data from Burn Cycle D2-N are shown in Figure 4.3. Arrows indicate concentrations of $PM_{2.5}$ just before refueling.
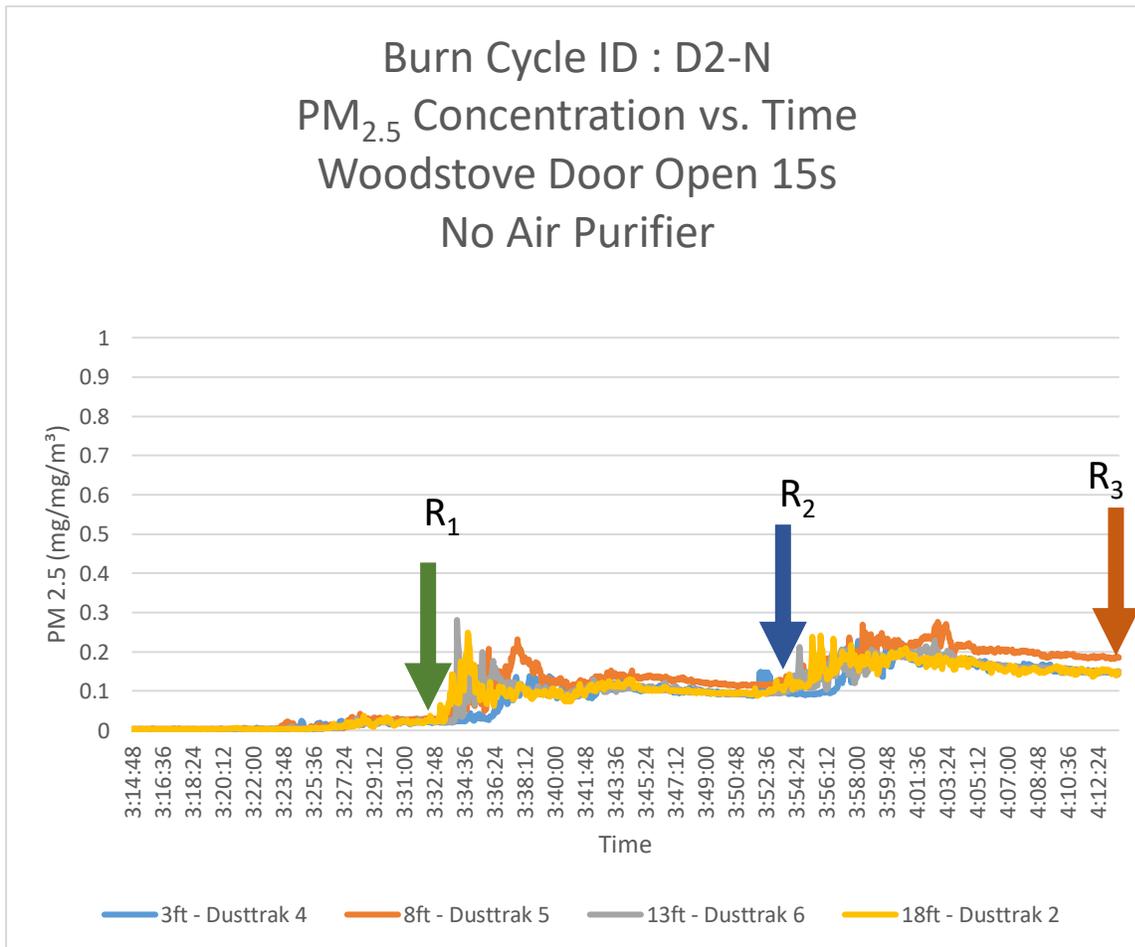
Figure 4.3. PM$_{2.5}$ Concentrations Over Time for Burn Cycle D2-N
with a 15s Woodstove Door-Open Period.

(Dusttrak II Instruments were Positioned at Several Distances from the
Woodstove. No Air Purifier was Used.)

The interval between refueling stages was 20 min while the woodstove door was left open for 15s for refueling. The woodstove door was opened for the first refueling, 10 min into the burn cycle. Dusttrak instruments were positioned at various distances from the woodstove, as indicated in Figure 4.3.

The $PM_{2.5}$ concentrations at each of the different distances of the instruments from the woodstove are shown in Tables 4.1 to 4.4. The significance of these values will be discussed in Section 5.3.1.

Table 4.1. PM2.5 Concentrations R_1, R_2, and R_3 Prior to Refueling Stages
Corresponding to Burn Cycle D1-N with <u>No</u> Air Purifier
(Woodstove Open 5s Every 20 min for Refueling)

$PM_{2.5}$ Prior to Refueling (mg/m³) at Various Distances from the Woodstove					
$PM_{2.5}$ Prior to Refueling (mg/m³)	@ 3'	@ 8'	@ 13'	@ 18'	Average
R_1	0.0381	0.0459	0.0393	0.0408	0.0410
R_2	0.1027	0.1306	0.1056	0.1037	0.1106
R_3	0.1108	0.1354	0.1118	0.1076	0.1164

Table 4.2. PM2.5 Concentrations R_1, R_2, and R_3 Prior to Refueling Stages
Corresponding to Burn Cycle D2-N with <u>No</u> Air Purifier
(Woodstove Open 15s Every 20 min for Refueling)

$PM_{2.5}$ Prior to Refueling (mg/m³) at Various Distances from the Woodstove					
$PM_{2.5}$ Prior to Refueling (mg/m³)	@ 3'	@ 8'	@ 13'	@ 18'	Average
R_1	0.0192	0.0207	0.0194	0.0212	0.0201
R_2	0.0880	0.1136	0.0937	0.0880	0.0958
R_3	0.1437	0.1830	0.1440	0.1403	0.1528

Burn Cycle D3-N lasted 90 min, compared to other burn cycles which lasted 60 min. As a result, for Burn Cycle DN-3, values of R_4 are given.

Table 4.3. PM2.5 Concentrations R_1, R_2, and R_3 prior to Refueling Stages
Corresponding to Burn Cycle D3-N with <u>No</u> Air Purifier.
(Woodstove Open 30s Every 20 min for Refueling)

PM$_{2.5}$ Prior to Refueling (mg/m³) at Various Distances from the Woodstove					
PM$_{2.5}$ Prior to Refueling (mg/m³)	@ 3'	@ 8'	@ 13'	@ 18'	Average
R_1	0.13942	0.14886	0.14192	0.13858	0.1422
R_2	0.18875	0.20545	0.19742	0.19248	0.1960
R_3	0.42557	0.48562	0.50846	0.42927	0.4622
R_4	0.60719	0.64624	0.57924	0.5508	0.5959

Table: 4.4. PM2.5 Concentrations R_1, R_2, and R_3 Prior to Refueling Stages
Corresponding to Burn Cycle D4-N with <u>No</u> Air Purifier.
(Woodstove Open 45s Every 20 min for Refueling)

PM$_{2.5}$ Prior to Refueling (mg/m³) at Various Distances from the Woodstove					
PM$_{2.5}$ Prior to Refueling (mg/m³)	@ 3'	@ 8'	@ 13'	@ 18'	Average
R_1	0.07728	0.07303	0.06835	0.07723	0.0740
R_2	1.27493	1.64362	1.3112	1.3172	1.3867
R_3	1.32657	1.64922	1.3559	1.33877	1.4176

4.3.2 Effect of Duration of Time the Woodstove Remained Open During Refueling with the Use of an Air Purifier

An example of data from Burn Cycle DW-4 with a 45s woodstove door-open period is shown in Figure 4.6. The woodstove door was opened 10 min into the burn cycle for the first refueling stage. Arrows indicate concentrations recorded just before each refueling stage. These are labeled R_1, R_2, R_3, and R_4, in Figure 4.7. These concentrations were averaged for each distance away from the woodstove and normalized, as shown in Tables 4.5 to 4.8.
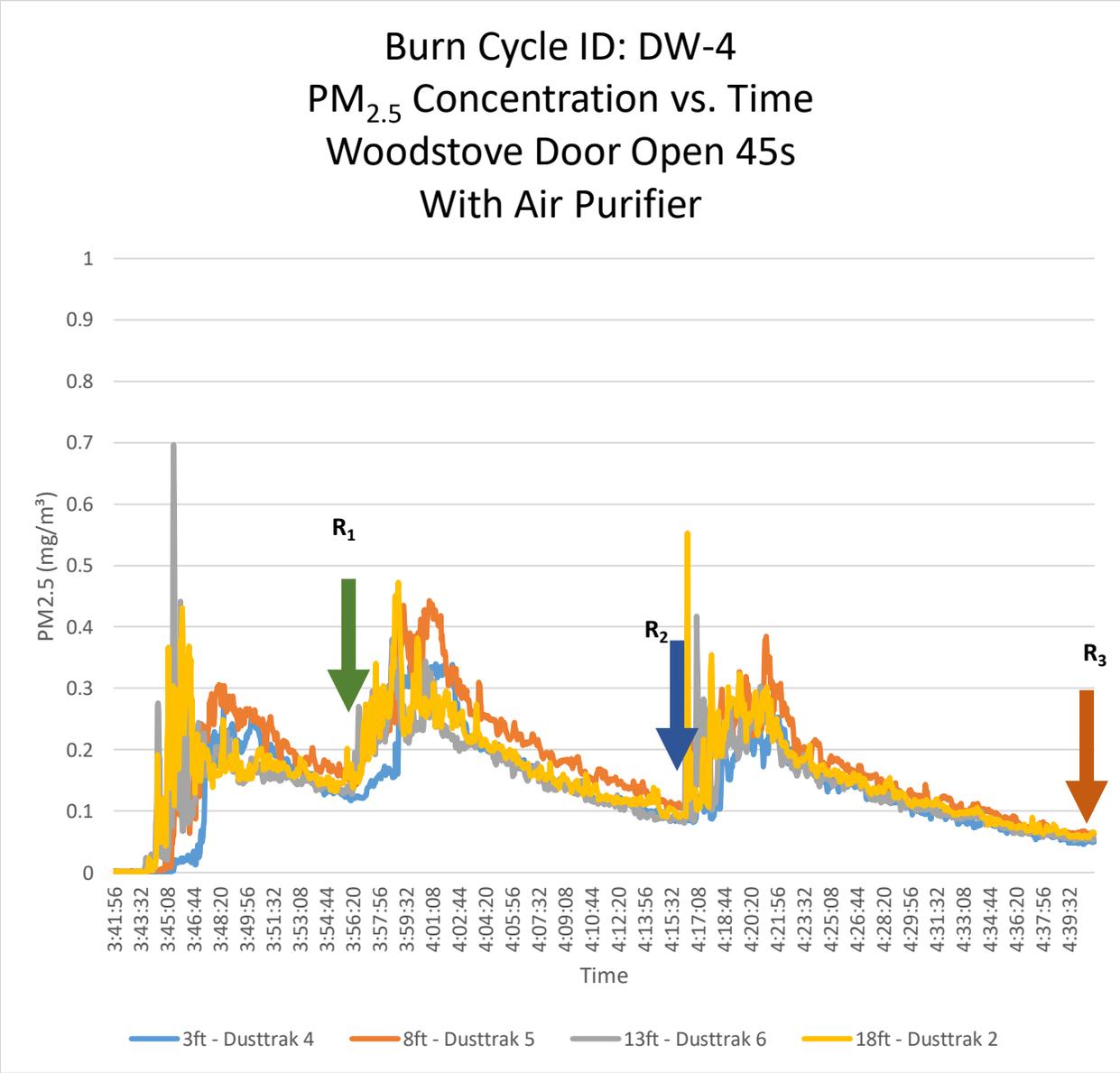
Figure 4.4. PM$_{2.5}$ Concentrations over Time for Burn Cycle D4-W
with a 45s Woodstove Door-Open Period.

(Dusttrak II instruments were positioned at various distances from the woodstove.
The time interval between refueling stages was 20 min. An air purifier was used.
Arrows indicate concentrations R$_1$, R$_2$, and R$_3$ recorded just before refueling.)

Table 4.5. PM$_{2.5}$ Concentrations R_1, R_2, and R_3 Prior to Refueling Stages
Corresponding toBurn Cycle D1-W <u>with</u> an Air Purifier.
(Woodstove Open 5s Every 20 min for Refueling)

PM$_{2.5}$ Prior to Refueling (mg/m³) at Various Distances from the Woodstove					
PM$_{2.5}$ Prior to Refueling (mg/m³)	@ 3'	@ 8'	@ 13'	@ 18'	Average
R_1	0.03846	0.04875	0.04116	0.04822	0.0441
R_2	0.04398	0.04894	0.03855	0.04352	0.0437
R_3	0.0244	0.03007	0.02645	0.02529	0.0265

Table: 4.6. PM$_{2.5}$ Concentrations R_1, R_2 and R_3 Prior to Refueling Stages
Corresponding to Burn Cycle D2-W <u>with</u> an Air Purifier.
(Woodstove Open 15s Every 20 min for Refueling)

PM$_{2.5}$ Prior to Refueling (mg/m³) at Various Distances from the Woodstove					
PM$_{2.5}$ Prior to Refueling (mg/m³)	@ 3'	@ 8'	@ 13'	@ 18'	Average
R_1	0.05787	0.0749	0.06314	0.06998	0.0665
R_2	0.08209	0.10142	0.09015	0.09252	0.0915
R_3	0.0365	0.04688	0.03706	0.03587	0.0391

Table: 4.7. PM$_{2.5}$ Concentrations R_1, R_2 and R_3 Prior to Refueling Stages
Corresponding to Burn Cycle D3-W <u>with</u> an Air Purifier
(Woodstove Open 30s Every 20 min for Refueling)

PM$_{2.5}$ Prior to Refueling (mg/m³) at Various Distances from the Woodstove					
PM$_{2.5}$ Prior to refueling (mg/m³)	@ 3'	@ 8'	@ 13'	@ 18'	Average
R_1	0.03437	0.03493	0.02831	0.02803	0.0314
R_2	0.02813	0.03792	0.03073	0.0343	0.0328
R_3	0.07425	0.08685	0.07506	0.07841	0.0786

Table: 4.8. PM$_{2.5}$ concentrations, R_1, R_2 and R_3 Prior to Refueling Stages
Corresponding to Burn Cycle D4-W <u>with</u> an Air Purifier
(Woodstove Open 45s Every 20 min for Refueling)

PM$_{2.5}$ Prior to Refueling (mg/m³) at Various Distances from the Woodstove					
PM$_{2.5}$ Prior to Refueling (mg/m³)	@ 3'	@ 8'	@ 13'	@ 18'	Average
R_1	0.11717	0.14326	0.12721	0.1327	0.1301
R_2	0.08155	0.10553	0.08288	0.08977	0.0899
R_3	0.04576	0.06014	0.05234	0.05645	0.0537

4.4   Effect of Various Types of Air Purifiers on Concentration Levels of PM$_{2.5}$

In this set of tests, the effectiveness of air purifiers on filtering PM$_{2.5}$ due to wood smoke was examined. The tests conducted are listed in Table 3.7. As previously stated, the tests conducted in this section have a test period of 12 hrs, and a data logging interval of 5s. Each instrument was positioned at a fixed height of 3'- 5.5", at distances 5', 10', 15', and 20' from the air purifier, as shown in Figure 3.5.

Figures 4.8 and 4.9 show typical PM$_{2.5}$ concentration levels with no air purifier and with an air purifier, respectively. It is clear that there is a spike in PM$_{2.5}$ concentration once the smoke hits the Dusttrak II devices, after which the levels drop as the PM$_{2.5}$ spreads across the room. Because high amounts of particulate are allowed to enter the room during each test, by the time PM$_{2.5}$ levels drop below 0.5 mg/m^3, it can be established the smoke has had more than enough time to distribute across the room. Comparing the results from the two 12-hr data sets, one from a burn cycle without the use of any air purifier, Figure 4.5, and one from a burn cycle with the use of air purifier, Figure 4.6, one can see a clear difference in dissipation or decay behaviour of the particulate between the two cases.
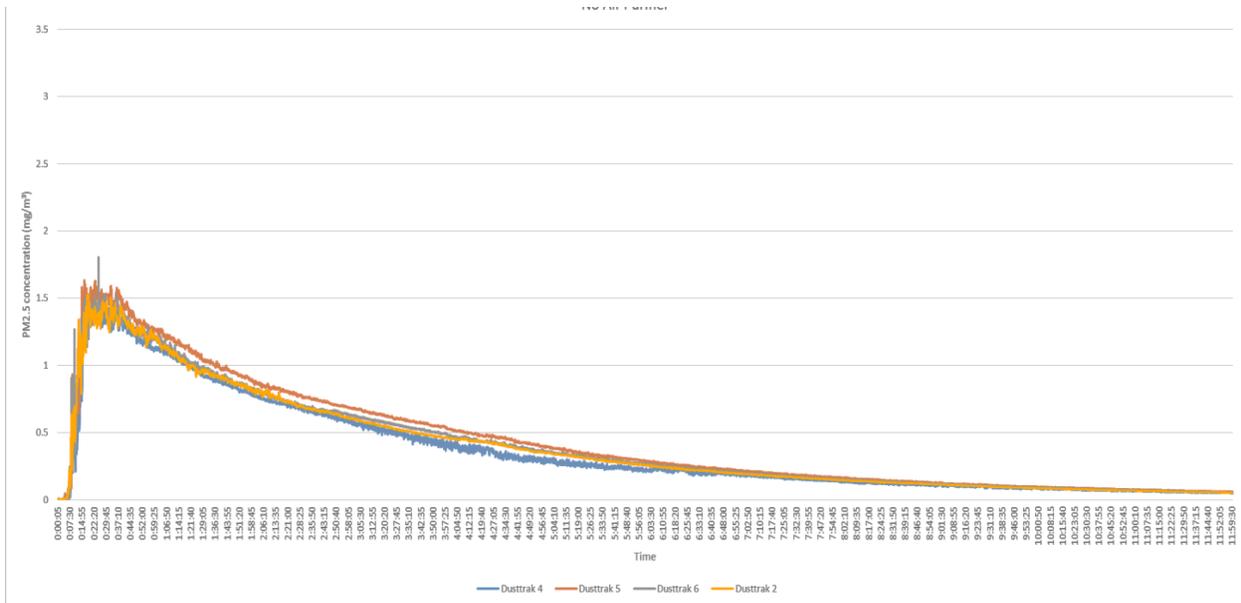
Figure 4.5. $PM_{2.5}$ Concentration Levels (mg/m^3) with <u>No</u> Air Purifier

Figure 4.6. $PM_{2.5}$ Concentration Levels (mg/m^3) <u>with</u> the Use of Air Purifier A.

(The spike in $PM_{2.5}$ is significantly higher than that in Fig 4.5 since this burn test took place on a different day and all parameters between days cannot be controlled.)

The exact amount of particulate allowed to disperse in the room when the woodstove was lit was not controlled. Instead, a general minimum threshold of at least 0.5 mg/m³ was ensured in all burn tests. In this sense, for comparison purposes between burn cycle data uniquely characterized by the presence of different air purifiers, or lack thereof, the maximum $PM_{2.5}$ detected is not important. Instead, what is of interest is the difference in time the various air purifiers took to reduce the $PM_{2.5}$ concentration levels from 0.5 mg/m^3 to an acceptable level of 0.035 mg/m^3. Dusttrak II instruments were positioned at various distances from the air purifiers in order to monitor any differences in the levels of particulates during the air purification process. The time it took each purifier to reduce the $PM_{2.5}$ in the smoke-filled room from 0.5 mg/m^3 to 0.035 mg/m^3 is referred to in this thesis as T-drop. T-drop values for all air purifiers are listed in Table 4.9.

Table 4.9. Average T-Drop Values for Each Test, Across all 4 Dusttrak II Positions, with and Without an Air Purifier

Air Purifier: A-I (Refer to Table 3.6)	T-drop at Dusttrak 4 – 20 feet from the Air Purifier (hr: min: s)	T-drop at Dusttrak 5 – 15 feet from the Air Purifier (hr: min: s)	T-drop at Dusttrak 6 – 10 feet from the Air Purifier (hr: min: s)	T-drop at Dusttrak 2 – 5 feet from the Air Purifier (hr: min: s)	Average T-drop (hr: min: s)
A	0:41:40	0:44:25	0:42:10	0:42:10	**0:42:36**
B	2:23:45	2:37:30	2:30:55	2:32:05	**2:31:04**
C (test 1)	1:02:40	1:03:20	1:03:45	1:04:00	**1:03:26**
C (test 2)	1:01:20	1:02:40	1:02:45	1:02:40	**1:02:21**
D	9:55:55	11+ hrs	11:00:55	10:14:55	**10:30+ hrs (test duration limitation)**
E	2:12:35	2:16:30	2:18:25	2:15:35	**2:15:46**
F	1:13:40	1:17:30	1:16:25	1:16:35	**1:16:03**
G	0:51:50	0:53:20	0:52:50	0:52:15	**0:52:34**
H	1:12:35	1:12:15	1:13:35	1:12:00	**1:12:36**
I	1:19:10	1:20:25	1:20:15	1:21:30	**1:20:20**
None	12+hrs	12+hrs	12+hrs	12+hrs	**12+ hrs (test duration limitation)**
None (Gravi-metric test)	19+hrs	19+hrs	19+hrs	19+hrs	**19+hrs (test duration limitation)**

CHAPTER 5.   ANALYSIS AND DISCUSSION

5.1. PM$_{2.5}$ Concentration as a Function of Height above Floor Level

In discussing the relevance of how height effects PM$_{2.5}$ concentration levels, as detected by the Dusttrak II instruments, it is important to understand the significance of the recorded spikes on residents exposed to wood smoke. The spikes in PM$_{2.5}$, shown in Figure 4.1, occur only over a short period of time. Concentrations level off to near identical levels after about 20 min across all instruments. It is clear then, that the overall 24-hr estimate for the average concentration would depend heavily on the duration of time and the frequency in which the woodstove door is left open. Alternatively, an estimated 24-hr equivalent concentration may be obtained from the data, as shown in Table 5.1.

Table 5.1.   Estimated 24-hr Equivalent PM$_{2.5}$ Concentration Data
at 18' from the Woodstove Obtained from a 90-min Burn Cycle*

Height Above Floor Level (ft-in)	Average PM$_{2.5}$ Concentration (mg/m³)	Normalized Average PM$_{2.5}$ Concentration (mg/m³)
2' - 5.5"	0.173886	1
3' - 5.5"	0.212895	1.224
4' - 5.5"	0.222763	1.288
5' - 5.5"	0.281218	1.617

*This theoretical 24-hr data set is strictly for comparison reasons and can only act as a rough estimate for a true burn test.

As shown in Table 5.1, there is a wide range of average PM$_{2.5}$ concentration values. The concentration level at 5'-5.5" above the floor is 61.7% higher than that at the 2'-5.5". A pattern is evident in this case. The higher the instrument is, the higher the average PM$_{2.5}$ concentration

value. Average $PM_{2.5}$ concentrations, normalized to the concentration level recorded by instruments located at 2'- 5.5" as function of height across all burn cycles, are shown in Table 5.2.

Table 5.2. Estimated 24-hr Normalized Equivalent $PM_{2.5}$ Concentrations (mg/m^3)
Obtained from all Burn Cycles and Normalized to the Concentration Level
Recorded by Instruments Located at a 2'- 5.5" Height above Floor Level

Height Above Floor Level (ft-in)	$PM_{2.5}$ Level at **3'** from the Woodstove (Burn Cycle H1)	$PM_{2.5}$ Level at **8'** from the Woodstove (Burn Cycle H2)	$PM_{2.5}$ Level at **13'** from the Woodstove (Burn Cycle H3)	$PM_{2.5}$ Level at **18'** from the Woodstove (Burn Cycle H4)	Average $PM_{2.5}$ Level over All Burn Cycles Normalized to level Recorded by Instruments Located at 2'- 5.5"
2'- 5.5"	1	1	1	1	1
3' -5.5"	1.118	0.995	1.112	1.224	1.112
4'- 5.5"	1.311	1.118	1.184	1.281	1.223
5'-5.5"	1.407	1.212	1.247	1.617	1.371

In every case, there was a significant increase in $PM_{2.5}$ concentration at heights higher than 2'-5.5" On average, there was a 37.1% increase at 5'- 5.5" compared to that at 2'- 5.5". The normalized averages for each burn cycle are shown in Figure 5.1. Only once, during Burn Cycle D2-N (instruments located at 8'), did the average $PM_{2.5}$ concentration drop, albeit, very slightly.
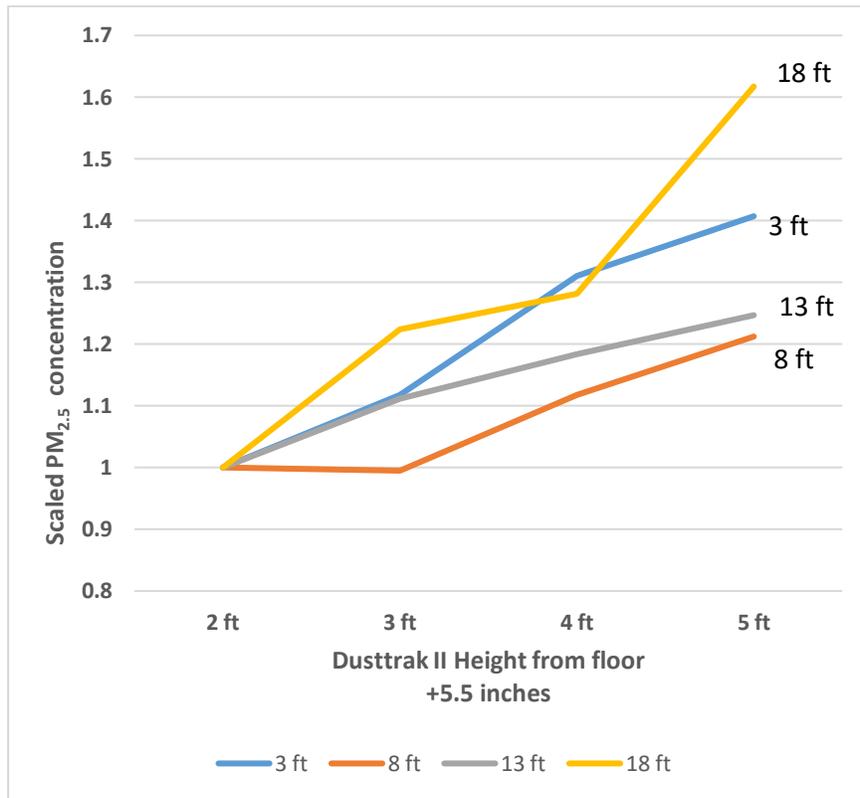
Figure 5.1. Estimated PM$_{2.5}$ Equivalent Concentrations for All Burn Cycles
Normalized to a Concentration Level Recorded by Instruments Located at a Height of 2'- 5.5"

These results are significant in that adult residents, being taller than young children, will be exposed to higher average PM$_{2.5}$ concentrations than children. Measuring a single PM$_{2.5}$ concentration in a home as a way of identifying potential risks, however, may not provide a complete picture. Only in combination with individual behaviour parameters can potential risks be accurately determined. It has yet to be shown how average PM$_{2.5}$ concentrations scale with height, and how this change compares to that from changes in distance from the woodstove, or with variations in residents' behaviour dealing with the woodstove door. These are discussed in the following sections.

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5.2   PM$_{2.5}$ Concentrations Due to Changes in Distance from the Woodstove

A typical PM$_{2.5}$ concentration measured at 3'-5.5" above floor level over a 90-min burn cycle is shown in Figure 4.2. Estimated 24-hr equivalent PM$_{2.5}$ concentration data from this 90-min data set are summarized in Table 5.3. As shown in Table 5.3, no significant pattern is observed in PM$_{2.5}$ concentration variance due to distance from the woodstove. Despite second-to-second differences in PM$_{2.5}$ detection between the instruments spanning the test room, when the PM$_{2.5}$ data are averaged over the entire period, there is little or no noticeable difference in PM$_{2.5}$ concentrations.

Table 5.3. 24-hr Equivalent PM$_{2.5}$ Concentration Data
from a 90-min Burn Cycle Recorded at a 3'-5.5" Height

Distance from Woodstove (ft)	Average PM$_{2.5}$ Concentration (mg/m³)
3	0.496293
8	0.541665
13	0.519918
18	0.489036

Table 5.4 displays how average PM$_{2.5}$ concentration varies with distance from the woodstove across all burn cycles. PM$_{2.5}$ concentration was measured at different heights varying from 1' to 6', as shown in Table 5.4.

Table 5.4. Determining the Effect that Distance from the Woodstove has
on $PM_{2.5}$ Concentration (mg/m³)*

Distance from the Woodstove (ft)	Height from the Floor Level						Average $PM_{2.5}$ Level over All Burn Cycles
	1'-5.5"	2'-5.5"	3'-5.5"	4'-5.5"	5'-5.5"	6'-5.5"	Normalized Concentration
3	1	1	1	1	1	1	1
8	1.2183	0.9716	1.0915	1.3786	1.1324	1.1366	1.155
13	1.095	0.9561	1.0477	1.2854	1.0463	1.0744	1.084
18	1.0579	0.8839	0.9855	1.0964	0.8948	0.9888	0.9846

*Obtained from 90-min burn cycles and normalized to the concentration level recorded by instruments located at 3' from the woodstove.

Unlike the results related to $PM_{2.5}$ concentration as a function of height above floor level, (discussed in the previous section), it appears that there is no consistent linear effect on $PM_{2.5}$ concentrations as a function of distance from the woodstove. This is further demonstrated by examining how $PM_{2.5}$ shifts over time during each of the six burn cycle tests, shown in Figure 5.2.
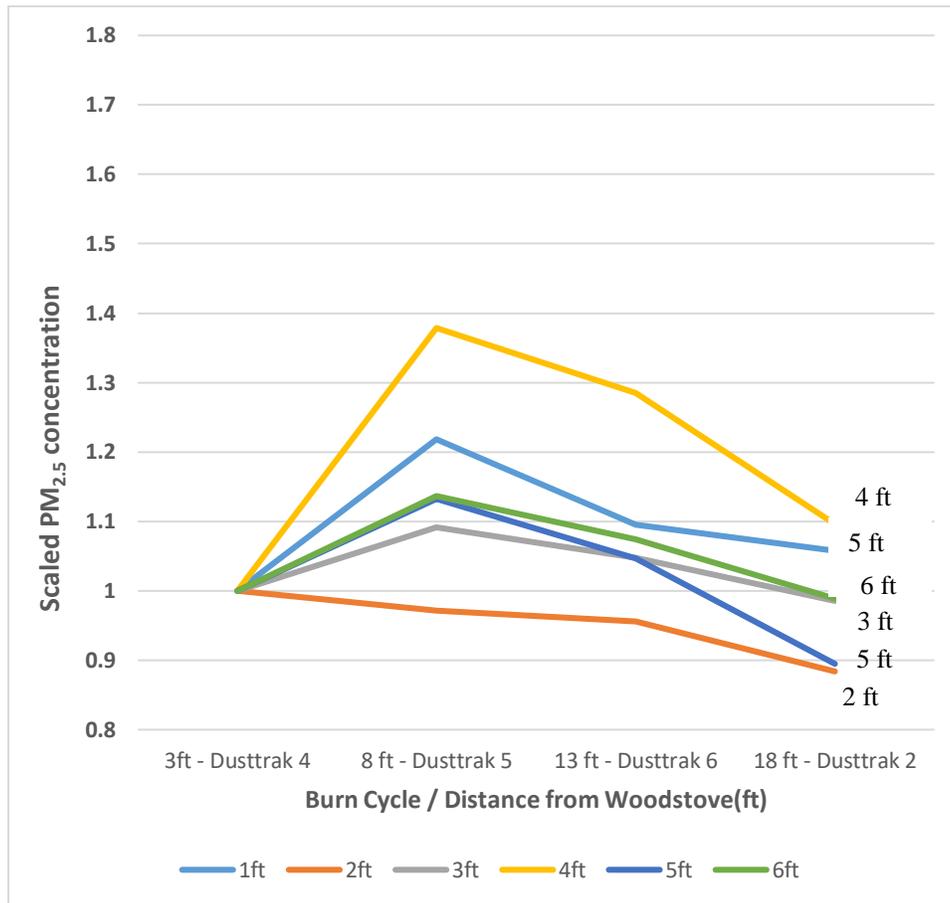
Figure 5.2. Average PM$_{2.5}$ Concentrations Measured at Several Distances from the Woodstove
During 90-min Burn Cycles Normalized to PM$_{2.5}$
Recorded by Instruments Located at a Distance of 3' from the Woodstove

(Each curve represents height above floor level. Note that the actual instrument nozzle was 5.5"
higher than the distances shown in this figure.)

On average, higher concentration levels were recorded at a distance of 8' from the woodstove,
with the exception of the concentrations recorded at 2' above floor level. At a distance of 8' from
the woodstove, a slightly higher PM$_{2.5}$ average of 15.5% was detected than that at 3'. The highest
level was recorded at 4' above floor level which was, approximately, 38% higher than that at 3'
from the woodstove. However, on average, the concentration levels were lower at distances

further away from the woodstove with the largest reduction, approximately 28%, recorded by instruments located at 4'-5.5" above floor level. As the wood smoke rises and travels along the walls and ceiling, it will curl back towards the center of the room from all sides. This could explain the slight increase in average $PM_{2.5}$.

Because there is no clear consistent pattern on how $PM_{2.5}$ concentrations change as a function of distance from the woodstove, no clear strategy can be recommended for residents to avoid high concentrations of $PM_{2.5}$ solely based on distance from the woodstove alone. Expecting a change in behaviour from residents with regards to altering their proximity from the woodstove on a typical day is unrealistic and even with any hypothetical proposed changes in behaviour, the effort would be insignificant, at best. It is clear that the person responsible for refueling the fire is at a potential higher risk than others if particulate matter is directly inhaled as it permeates the air. However, this is not as critical as one might think as the smoke rises directly up from the woodstove door towards the ceiling. Only once it reaches the ceiling will it begin to travel back across the room away from the woodstove. The $PM_{2.5}$ that reached the instruments located at a 3ft distance mark from the woodstove had, for the most part, already traversed the room. With regards to average $PM_{2.5}$ concentrations, there is little or no difference between the readings from the various instruments, indicating particulate matter had well saturated the room over the period of testing and had settled in the room in a pattern that varies according to height, and not so much distance.

5.3 PM$_{2.5}$ Concentration as a Function of Woodstove Door Opening for Refueling

It should be noted that each burning cycle constitutes a test period. Thus, results from different burn cycles cannot be directly compared (this issue is discussed in more detail in Appendix A). For the positional tests, more accurate comparisons between data collected by the Dusttrak II instruments could be made because they were running simultaneously during a single burn cycle. In contrast, when comparing data from different burn cycles listed earlier, parameters which are outside the scope of study, could in fact introduce significant error, such that a direct comparison would yield ambiguous results. Thus, in discussing the behaviour surrounding the woodstove door open time, the analysis was done within each period's data individually, and only patterns between periods are discussed qualitatively.

In the following discussion, the terms R$_1$, R$_2$, R$_3$, and R$_4$ are used to define concentrations during various stages of a burn cycle so patterns between concentrations from various burn tests can be examined. These terms are defined below.

R_1: The woodstove door was initially open to load the wood and light the fire. It was kept open for enough time to ensure that the kindling and wood have both visibly caught fire. The lowest measured PM$_{2.5}$ concentration of particulate due to biomass burning, initially detected by the Dusttrak instruments prior to the woodstove door being re-opened for refueling, is identified as R$_1$. This is not to be confused with ambient levels, which are negligible in comparison. R_1 can

be best described as an estimate of indoor $PM_{2.5}$ concentration during a burn cycle if the door is never opened for refueling.

R_2: This refers to the lowest $PM_{2.5}$ concentration detected after the woodstove door was re-opened for the first time in order to refuel the fire, but prior to the $PM_{2.5}$ spike from the woodstove door being refueled for a second time.

R_3: This is the lowest $PM_{2.5}$ concentration detected after the woodstove door was re-opened for a second time in order to refuel the fire, but prior to the $PM_{2.5}$ spike from the woodstove door refueled for the third time.

R_4: This refers to the lowest concentration after the woodstove was reopened for the third time in order to refuel the woodstove for the fourth time. This only happened in Burn Cycle D3-N which lasted 90 min.

Comparing R_1 to $R_2,$ R_3, and R_4 provides a rough estimation of the particulate accumulation after each refueling stage. More specifically, these R values represent the lowest possible $PM_{2.5}$ concentrations measured after each spike in $PM_{2.5}$ but prior to the next spike caused from reopening and refueling the woodstove door.

When the concentration R is consistently increasing from R_1 to R_4, it is said to be cumulative. This is because the value of $PM_{2.5}$ will keep climbing each time the woodstove door is opened for refueling as there is not enough time between each refueling cycle for $PM_{2.5}$ levels to drop below the previous stage. When there is evidence that $PM_{2.5}$ levels drop between successive values of R, then the concentration is said to be dissipating.

5.3.1 Effect of Duration of Time the Woodstove Remained Open During Refueling Without the Use of an Air Purifier

The results from Burn Cycle D2-N are shown in Figure 5.3 along with the designated concentrations R_1, R_2, and R_3, as defined in the previous section.

Comparing $PM_{2.5}$ concentration levels immediately prior to the first spike, which is attributed to the first refueling and is labeled as R_1, with the concentration level immediately prior to the second spike, which is attributed to the to the second refueling and is labeled R_2, will yield a rough estimation of the indoor particulate sustainability. If a third spike occurs in the data, a concentration level, R_3, can be determined. These points can be described as R-$PM_{2.5}$ concentration values and will act as snapshots to help establish the system's $PM_{2.5}$ sustainability, or trend. Additionally, by utilizing this method, exact concentrations of $PM_{2.5}$ are not required between data sets from different days for analysis. Instead, each burn cycle data set will be individually characterized by how R-values of $PM_{2.5}$ change over time within that burn cycle. Comparing the nature of each burn cycle, in this sense, will eliminate any possible error

introduced upon direct PM$_{2.5}$ magnitude comparisons between data sets. A more detailed

discussion of this is provided in Appendix A.
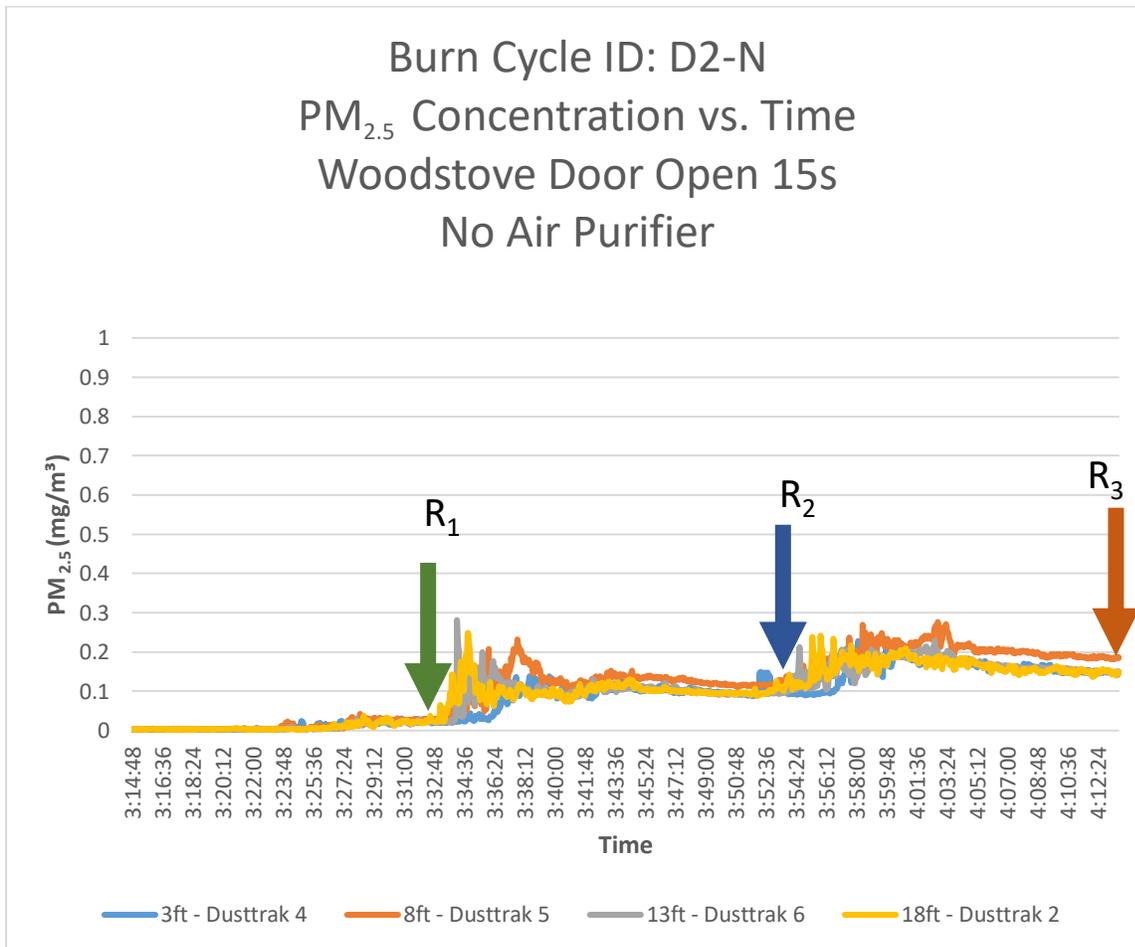
Figure 5.3. PM$_{2.5}$ Concentrations Over Time for Burn Cycle D2-N
with a 15s Woodstove Door-Open Period

(Dusttrak II instruments were positioned at several distances from the woodstove. No air purifier
was used. Arrows indicate concentrations R$_1$, R$_2$, and R$_3$ recorded just before refueling.)

The PM$_{2.5}$ concentrations R$_1$, R$_2$ and R$_3$ were normalized and plotted for each of the eight burn

cycles. Measurements were taken at several different distances, at a fixed height of 3'- 5.5". The

R-PM$_{2.5}$ values at each of the different distances of the instruments from the woodstove were

averaged in order to determine overall R_1, R_2 and R_3 concentrations. The results are shown in

Tables 5.5 to 5.8.

Table 5.5. Average and Normalized PM$_{2.5}$ Concentrations Prior to Refueling Stages
Corresponding to Burn Cycle D1-N with <u>No</u> Air Purifier
(Woodstove Open 5s Every 20 min for Refueling)

PM$_{2.5}$ Prior to Refueling (mg/m³)	Average	$\overline{R_N}$ Normalized to Average R_1
R_1	0.0410	1.0000
R_2	0.1106	2.6920
R_3	0.1164	2.8342

Table 5.6. Average and Normalized PM$_{2.5}$ Concentrations Prior to Refueling Stages
Corresponding to Burn Cycle D2-N with <u>No</u> Air Purifier
(Woodstove Open 15s Every 20 min for Refueling)

PM$_{2.5}$ Prior to Refueling (mg/m³)	Average	$\overline{R_N}$ Normalized to Average R_1
R_1	0.0201	1.0000
R_2	0.0958	4.7603
R_3	0.1528	7.5905

Burn Cycle D3-N lasted 90 min, compared to other burn cycles which lasted 60 min. As a result,

for Burn Cycle D3-N, values of R_4 are given.

Table 5.7. Average and Normalized PM$_{2.5}$ Concentrations Prior to Refueling Stages
Corresponding to Burn Cycle D3-N with <u>No</u> Air Purifier.
(Woodstove Open 30s Every 20 min for Refueling)

PM$_{2.5}$ Prior to Refueling (mg/m³)	Average	$\overline{R_N}$ Normalized to Average R_1
R_1	0.1422	1.0000
R_2	0.1960	1.3786
R_3	0.4622	3.2506
R_4	0.5959	4.1904

Table: 5.8. Average and Normalized $PM_{2.5}$ Concentrations Prior to Refueling Stages Corresponding to Burn Cycle D4-N with <u>No</u> Air Purifier.
(Woodstove Open 45s Every 20 min for Refueling)

$PM_{2.5}$ Prior to Refueling (mg/m^3)	Average	$\overline{R_N}$ Normalized to Average R_1
R_1	0.0740	1.0000
R_2	1.3867	18.7466
R_3	1.4176	19.1640

As the results shown in Figure 5.3 and listed in Tables 5.5 to 5.8 demonstrate, there is no indication that average $PM_{2.5}$ levels are dropping over successive fueling/refueling stages. In fact, the $PM_{2.5}$ concentrations tend to be cumulative, increasing with each refueling stages. This indicates that indoor $PM_{2.5}$ concentrations are not sustainable. In other words, concentration levels show no indication of dropping towards safe levels.

The implications of these results are that if $PM_{2.5}$ concentration levels rise above typically recommended safe levels and not enough time is allowed to elapse between refueling stages, then concentrations will never fall below safe levels, assuming the woodstove burns continuously and no external method for accelerating particulate dissipation is utilized.

Testing whether $PM_{2.5}$ levels are sustainable in the home while varying the period of time the woodstove door is left open during refueling will indicate whether further counter measures, such as the use of air purifiers, must be used in order to establish safe best practice and to create a healthier environment for residents.

5.3.2 Effect of Duration of Time the Woodstove Remained Open During Refueling with the Use of an Air Purifier

An example of data from Burn Cycle D4-W with a 45s woodstove door-open period is shown in Figure 5.4. The woodstove door was opened 10 min into the burn cycle for the first refueling stage. Arrows indicate concentrations recorded just before each refueling stage. These are labeled R_1, R_2, R_3, and R_4, in Figure 4.7. These concentrations were averaged for each distance away from the woodstove and normalized, as shown in Tables 5.9 to 5.12.
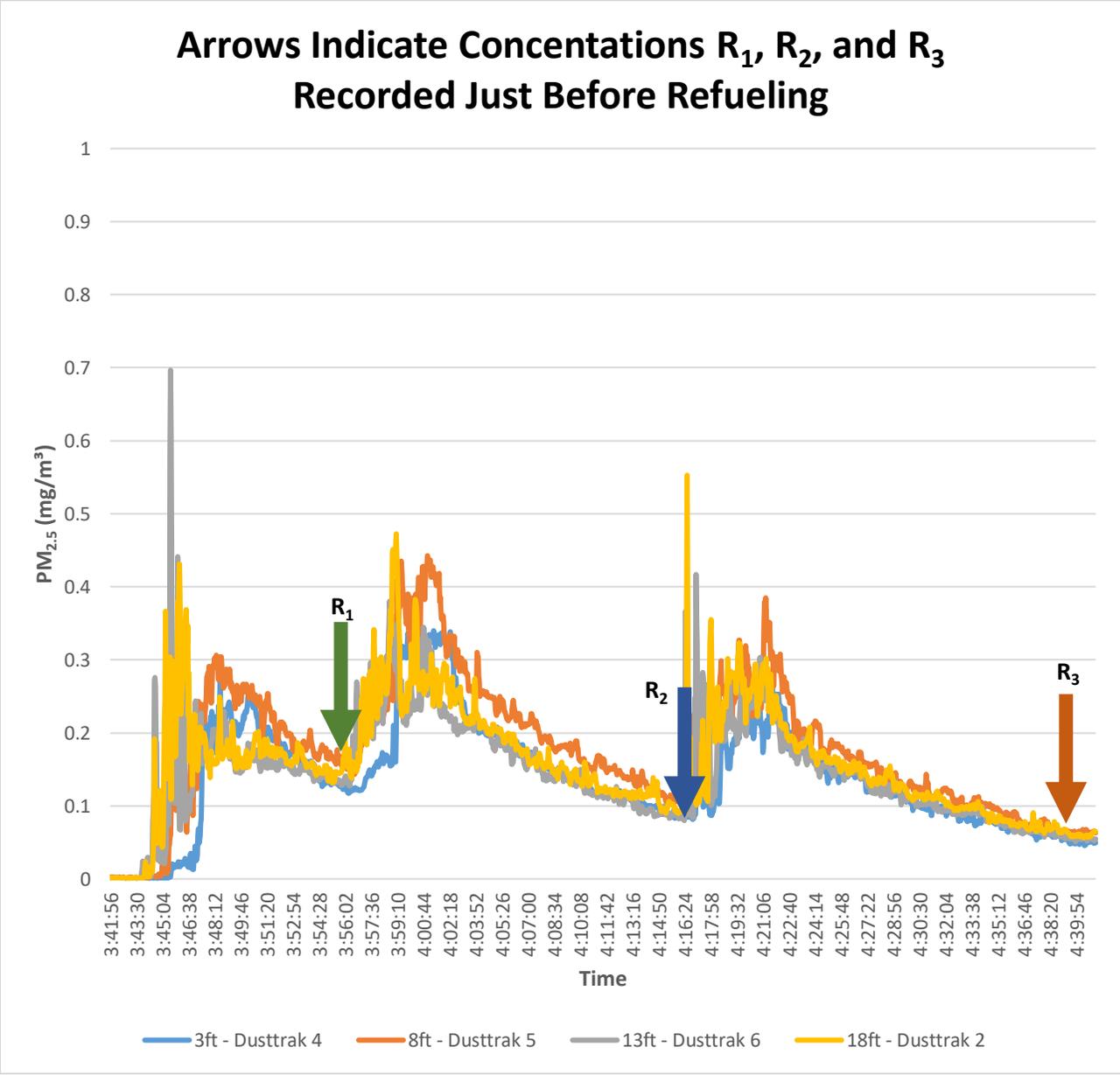
Figure 5.4. PM2.5 Concentrations Over Time for Burn Cycle D4-W
with a 45s Woodstove Door-Open Period

(Dusttrak II instruments were positioned at various distances from the woodstove.
The time interval between refueling stages was 20 min. An air purifier was used.
Arrows indicate concentrations R_1, R_2, and R_3 recorded just before refueling.)

Table 5.9. Average and Normalized PM$_{2.5}$ Concentrations Prior to Refueling Stages Corresponding to Burn Cycle D1-W <u>with</u> an Air Purifier.
(Woodstove Open 5s Every 20 min for Refueling)

PM$_{2.5}$ Prior to Refueling (mg/m³)	Average	$\overline{R_N}$ Normalized to Average R$_1$
R$_1$	0.0441	1.0000
R$_2$	0.0437	0.9909
R$_3$	0.0265	0.6014

Table: 5.10. Average and Normalized PM$_{2.5}$ Concentrations Prior to Refueling Stages Corresponding to Burn Cycle D2-W <u>with</u> an Air Purifier.
(Woodstove Open 15s Every 20 min for Refueling)

PM$_{2.5}$ Prior to Refueling (mg/m³)	Average	$\overline{R_N}$ Normalized to Average R$_1$
R$_1$	0.0665	1.0000
R$_2$	0.0915	1.3772
R$_3$	0.0391	0.5879

Table: 5.11. Average and Normalized PM$_{2.5}$ Concentrations Prior to Refueling Stages Corresponding to Burn Cycle D3-W <u>with</u> an Air Purifier
(Woodstove Open 30s Every 20 min for Refueling)

PM$_{2.5}$ Prior to Refueling (mg/m³)	Average	$\overline{R_N}$ Normalized to Average R$_1$
R$_1$	0.0314	1.0000
R$_2$	0.0328	1.0434
R$_3$	0.0786	2.5038

Table: 5.12. Average and Normalized PM$_{2.5}$ Concentrations Prior to Refueling Stages Corresponding to Burn Cycle D4-W <u>with</u> an Air Purifier
(Woodstove Open 45s Every 20 min for Refueling)

PM$_{2.5}$ Prior to Refueling (mg/m³)	Average	$\overline{R_N}$ Normalized to Average R$_1$
R$_1$	0.1301	1.0000
R$_2$	0.0899	0.6914
R$_3$	0.0537	0.4126

5.3.3 PM$_{2.5}$ Concentration as a Function of Woodstove Door Opening for Refueling: Discussion of the Results

The *R* values depend on the number of times the woodstove door is reopened during each burn cycle. In this study, at least three *R* values were recorded for each burn cycle. In those tests that the burn cycle ended prior to a third door opening, R_3 was estimated to be the lowest PM$_{2.5}$ detected at least 20 min after R$_2$, as this was the exact duration of time between refueling stages.

R_1 was used here as the base line with which other values of R concentrations were normalized within each burn cycle in order to make comparisons of PM$_{2.5}$ concentration patterns between burn cycles. The rationale behind this is that there are far too many unmeasured and uncontrolled parameters differentiating data from various burn cycles on different days. Instead of direct comparisons of PM$_{2.5}$ values, patterns in concentration change were identified between refueling stages during each burn cycle. While discussing the effect of the time that the woodstove door was left open, and also the effect of using an air purifier, on the values of *R*, the formation and dissipation of PM$_{2.5}$ particulate levels can be best be described using a normalized parameter $\overline{R_N}$. This normalized parameter is defined as the ratio of the average concentration R$_i$ (i = 1, 2, 3, 4) to the average concentration R$_1$ from all instruments.

By using a baseline, the change in the value of the concentration parameter R during each successive woodstove door opening may be characterized and compared to particulate progression during a different burn cycle. Exact levels of PM$_{2.5}$ concentrations are lost once data

are normalized. However, this is not that important when comparing particulate dispersion patterns from different test days.

Normalized and averaged $PM_{2.5}$ concentrations $\overline{R_N}$ listed in Tables 5.5 to 5.12 for Burn Cycles D1-N through D4-N as well as D1-W through D4-W, are shown in Figure 5-5. Each trend is color-coded, where solid lines represent R data for burn cycles without the use of an air purifier and dotted lines represent R data for burn cycles conducted with the use of an air purifier running.
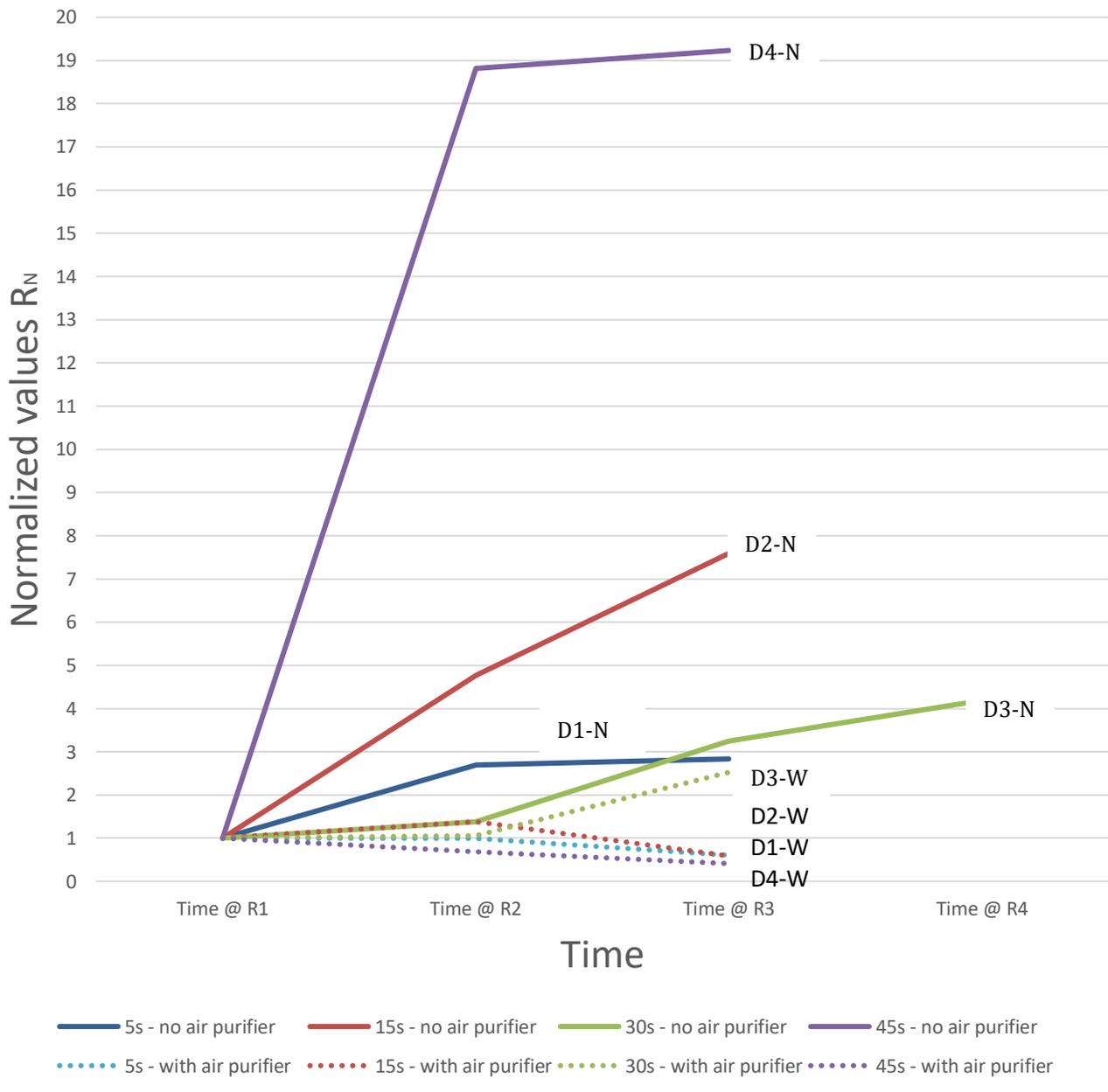
Figure 5.5. PM2.5 Average Concentrations $\overline{R_N}$ Normalized to Average Initial Concentration R_1 as a Function of Time for all Burn Cycles

(The woodstove door opened approximately every 20 min for each Burn Cycle. Refer to Tables 3.4 and 3.5 for Test Identification.)

As shown in Figure 5.5, the introduction of an air purifier allows for a significant reduction in PM$_{2.5}$ levels during the 20-min periods between refueling stages, indicating a trend towards safer levels.

All burn tests conducted with the use of an air purifier, except Burn Cycle D3-W, with a 30s refueling period, showed dissipating PM$_{2.5}$ levels throughout the duration of testing. All tests conducted without the use of an air purifier, showed continuous cumulative PM$_{2.5}$ concentration levels after each refueling stage. It appears that the longer the woodstove door remained open, the higher the accumulation was. The most cumulative effect was observed in Burn Cycle D4-N where the woodstove door was left open the longest (45s) for refueling.

From the results shown in Figure 5-5, the significant effect of the air purifier in dissipating the PM$_{2.5}$ concentrations is also clear. This is especially evident in the case of Burn Cycle D4-W where a 45s refueling stage was used. Without the use of the air purifier, PM$_{2.5}$ concentration levels increased over 20 times their initial state, R$_1$, and continued to cumulate in successive stages. In contrast, when an air purifier was used, low particulate levels were recorded from the start and continued to dissipate with time.

The only case that resulted in low particulate concentrations without the use of an air purifier was in Burn Cycle D1-N where the woodstove door was opened for just 5s to refuel. This,

however, proved to be an unrealistically short period of time because refueling involves physical work, loading more wood and/or kindling, and rearranging the wood in the woodstove via poking. The results, nevertheless, are useful. They provide a bottom-end benchmark to be used to compare against the results from burn cycles with the use of an air purifier for the various durations that the woodstove door is left open (5s, 15s, 30s, and 45s). This comparison will help determine the effectiveness of utilizing an air purifier.

The level of particulate accumulation is not the only important factor when discussing health effects due to smoke. Extremely high $PM_{2.5}$ concentrations over an extensive period of time may pose a serious health threat. If, however, $PM_{2.5}$ concentration levels continue to increase by an order of magnitude from fueling to refueling, then it will still take a long period of time before $PM_{2.5}$ levels reach safe levels in the home. Examining particulate matter concentration while varying the periods of time the woodstove door is left opened during refueling, both with and without utilizing an air purifier, provides one way of determining the potential benefit of utilizing wood smoke filtration units, under a range of woodstove-refueling behaviours

It should be noted that, in this study, an arbitrary but very liberal 20-min interval was chosen between woodstove refueling stages. This is a worst-case scenario since, in some cases, the time between refueling could be much longer – a situation which could yield significantly different results.

Whether this could help residents establish particulate sustainability within the home will, of course, heavily depend on the time scale of $PM_{2.5}$ dissipation rates, which air purifiers have

shown to affect. The scale of $PM_{2.5}$ dissipation over extended periods of time, 12+ hrs, is discussed in the next section of this thesis.

5.4   Effect of Various Types of Air Purifiers on Concentration Levels of $PM_{2.5}$

The T-Drop, defined in Section 4.4 as the time it took for the $PM_{2.5}$ concentration to drop from 0.5 mg/m^3 to the acceptable level of 0.035 mg/m^3, for the various burn tests, with and without an air purifier, is listed in Table 4.9. The T-Drop, as recorded by Dusttrak #4 located 20' from the air purifier, is plotted in Figure 5.6.

The results shown in Table 4.12 and plotted in Figure 5.6, indicate that the use of an air purifier can potentially lead to a significant reduction in particulate matter due to wood smoke. Of the air purifiers tested, those that employed HEPA technology proved to reduce $PM_{2.5}$ levels significantly. HEPA filters, or High Efficiency Particle Air Filters, essentially are designed to trap 99.9% particles of 0.3 µm or larger. In the case of air purifier D, where no HEPA filter was present, its effectiveness in reducing $PM_{2.5}$ is by far worse than any of the other solutions. The T-drop in this burn test was, on average, over 10 hrs. The much less expensive HEPA air purifier, B, took only 2 hrs and 31 min to bring the $PM_{2.5}$ levels from 0.5 mg/m³ to acceptable safe levels of 0.035 mg/m³.
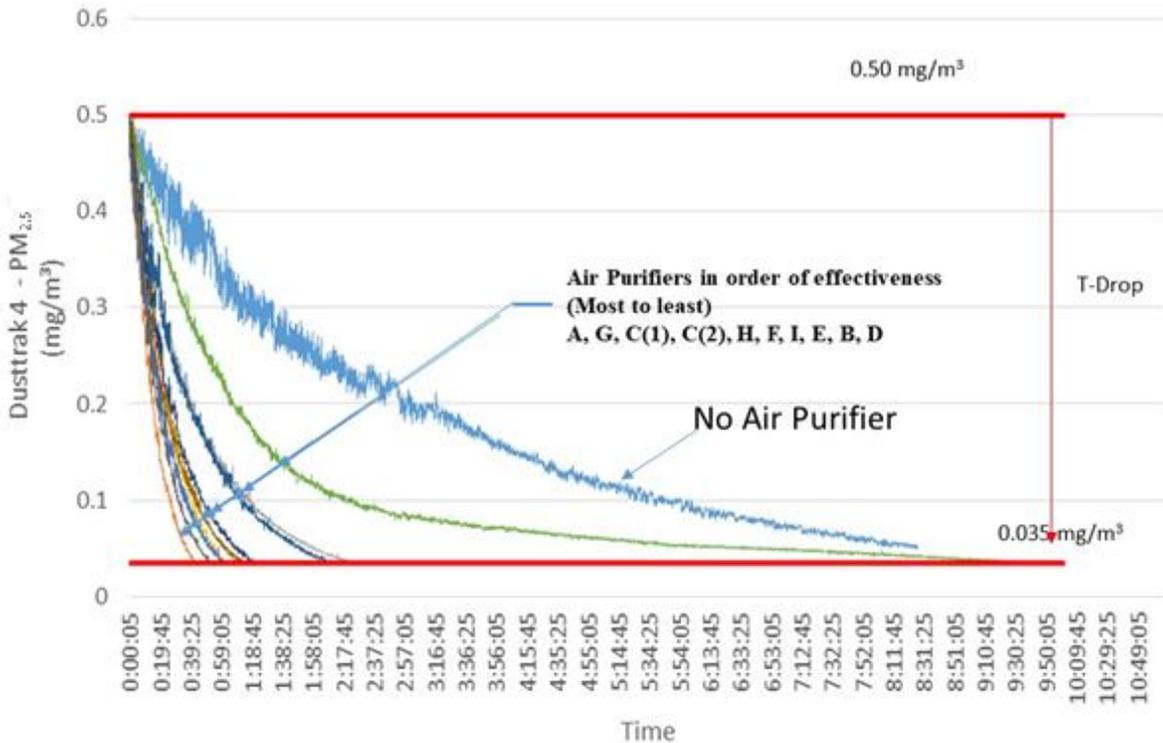
Figure 5.6. PM$_{2.5}$ Dissipation Due to the Use of Air Purifiers
as Recorded by Dusttrak II #4 Located 20' from the Air Purifier.

(Refer to Table 3-6 for Air Purifier Identification.)

The error between data from different burn cycles can be estimated by comparing the T-drops between the two tests. In this research part of the investigation, two tests were conducted using Air Purifier C, shown in Table 4.9. Tests #1 and #2 were conducted with the same air purifier on different days, and characterized by radically different PM$_{2.5}$ peaks. The difference between the two T-drops is 1.7%. This difference is within TSI's instrumental error of 5%, as discussed in Appendix A.

It is interesting to note that the more expensive Air Purifier A ($700 CAD) was the most effective in achieving the T-drop in approximately 42 min compared to the T-drop of 2 hrs, 31 min achieved by Purifier B ($70 CAD), emphasizing that there is a clear scale of effectiveness when comparing different HEPA air purifiers. However, it is evident from the results that even the less expensive HEPA air purifiers significantly decreased $PM_{2.5}$ levels in the test home.

HEPA technology allows the fibers in the air purifier to help trap smoke particulate. Because all but one of the air purifiers employed in this experiment use this technology, the limiting factor would be the rate at which the internal fan can pull air through the HEPA filter. Carbon filters are employed in Air Purifiers A and C which, unlike Air Purifier B, both employ multi-stage filtration.

The ozone generating Air Purifier D used in this study does not use HEPA filtration, but the manufacturer still claims that it can be used for smoke filtration and the air purifier is promoted as a healthy option. This air purifier also uses dual negative ionization, which the manufacturer claims it helps eliminate smoke particles in the air even when they do not pass directly through the air purifier. In the tests conducted in this study, it was found that ionization and ozone creation, despite claims made by the manufacturer, have little to no effect on $PM_{2.5}$ levels. The T-drop for this air purifier is just under 10 hrs, which is closer to the case of no air purifier. The main difference with regards to $PM_{2.5}$ dissipation between this ozone generating air purifier and those that utilize HEPA is the rate of $PM_{2.5}$ dissipation. Whereas, initially, Air Purifier D showed

a significant rate of dissipation in $PM_{2.5}$, this dissipation rate seems to stagnate noticeably once a threshold is met. At the end, the T-drop in the case of the ozone Air Purifier D was over 10.5 hrs, indicating that this air purifier was not effective in reducing the indoor $PM_{2.5}$ concentrations to safe levels.

As mentioned earlier, the EPA standards for healthy living put a 24-hr maximum value for indoor $PM_{2.5}$ levels at 0.035 mg/m^3. The woodstove used in the test home is obviously a health hazard and if levels are expected to be in the healthy range defined by the EPA, this particular woodstove simply cannot be used. Air purification is only attempting to lower the already dangerously high levels, which it does effectively in comparison to a scenario where no air purifiers are used. It does not, however, target the root of the problem, which is the stove itself being either improperly installed or improperly maintained. $PM_{2.5}$ levels in this study did not drop below the EPA safe limit in sufficient time, in any of the tests conducted. The sufficient time described here was determined to be roughly 2 hrs from the first part of the gravimetric test, which is the maximum allowed time between opening the woodstove door to ensure the fire is burning.

During testing, $PM_{2.5}$ concentrations at several distances from the air purifier were measured in order to determine differences in air purification around the air purifiers. Dusttrak II instruments were placed at distances of 5', 10', 15' and 20' and at a fixed height of 3'- 5.5'. In the distance testing reported earlier, it was shown that there is little difference in average $PM_{2.5}$ concentrations at different distances from the woodstove. This time, the effectiveness of each air purifier at

several distances was examined. This was accomplished by examining differences in T-drop at each location of the instruments (see Table 4.9). The results are shown in Table 5.13. These showed that there is no systematic shift in T-drop due to a change in distances.

Table 5-13. Normalized $PM_{2.5}$ Concentrations at Various Distances from the Air Purifier
(Normalized to $PM_{2.5}$ Concentration Recorded at 20' from the Air Purifier)

Air Purifier (Refer to Table 3-6)	Average $PM_{2.5}$ (mg/m³) over a 24-hr period at Different Distances from the Air Purifier in the Direction of the Woodstove			
	20' Dusttrak II #4*	15' Dusttrak II #5	10' Dusttrak II #6	5' Dusttrak II #2**
A	1	1.241572	1.097678	1.13858
B	1	1.326395	1.10576	1.185414
C(1)	1	1.227665	1.018802	1.080086
C(2)	1	1.248713	1.030767	1.180291
D	1	1.37638	1.171961	1.244345
E	1	1.072788	0.999996	1.026705
F	1	1.191337	1.090621	1.112854
G	1	1.054203	1.018239	1.096745
H	1	1.048865	0.978681	0.979216
I	1	1.154757	0.954919	1.032885
No Air Purifier	1	1.174931	1.090697	1.066412

* Instrument closest to the woodstove
** Instrument closest to the air purifier

It appears there is no consistent pattern characterizing the effect of distance on average $PM_{2.5}$ concentration data collected. Although one might think that the space closest to the air purifier would be the least impacted by $PM_{2.5}$, there is no indication in the results to support this argument. The average normalized values appear to be random. The results, however, show that, in general, $PM_{2.5}$ concentrations decreased at distances further away from the woodstove and close to the air purifier. The reductions, however, were rather small (less than 12%).

5.4.1 Effect of Various Types of Air Purifiers on Concentration Levels of PM$_{2.5}$: Concluding Remarks

Given that many northern communities across Canada are remote, options for alternative heating sources are limited. Electricity in these remote areas is expensive or nonexistent. As a result, residents rely on woodstoves as a primary source of heating and wood used for fuel is readily available.

One of the main goals of this study was to determine whether using air purifiers can be an effective means of reducing PM$_{2.5}$ levels due to wood smoke. The test home that was selected is located on a First Nations community and was, for many years, the main residence of a family of four. The woodstove in the home had been used for over a decade as the regular heating source. In this part of the study, two Dusstrak II devises were used to measure PM$_{2.5}$ concentrations in mg/m^3. The results show that the use of air purifiers incorporating HEPA filters proved beneficial in reducing PM$_{2.5}$ levels faster than if no air purifiers were used. Spikes in PM$_{2.5}$ recorded immediately after the woodstove door was opened quickly dissipated with the use of the air purifier. The drop in PM$_{2.5}$ concentrations would have been significantly smaller and would have taken much longer to drop to acceptable levels if no air purifiers were used. It took over 12 hrs for levels to drop from 0.5 to 0.035 mg/m^3 without an air purifier.

The test house used in the study proved to have extremely high levels of PM$_{2.5}$ when the woodstove was in use. Even with the use of a HEPA air purifier, which in this study proved to be

effective in reducing $PM_{2.5}$ levels, it would have been impossible for the $PM_{2.5}$ concentration to drop to an EPA acceptable level. The best solution, in this case, would be to replace the woodstove with a more efficient EPA-certified stove. However, the cost of such a replacement may be outside the affordability of many residents.

Wood is not a perfect fuel. According to Jones (2014), all energy producing fuels impact the environment, including wood burning. He lists three major environmental issues associated with wood burning: nuisance smoke (caused by neighbors inefficiently heating their homes); air-shed contamination (caused by too much smoke produced in areas with a depressed topography, such as a river valley, which is prone to temperature inversions in the winter that trap smoke close to the ground); and indoor air pollution (caused by leaky or inefficient in-house wood-burning appliances). As the literature cited in the present study points out and the results from the research study confirm, burning wood produces smoke, which can be harmful to children, the elderly, and people with lung diseases or allergies. Smoke from wood burning stoves is a public health risk. The high cost associated with health issues due to wood smoke is not sustainable. Furthermore, over the last few years, the death toll from fires in First Nations communities caused by woodstoves has risen dramatically. A CBC news report aired on December 21, 2015, stated that "…fire incidence rates for First Nations are 2.4 times higher than for the rest of Canada. First Nations residents are also 10 times more likely to die in a house fire." (CBC, 2015). According to the CBC Report, the victims are often young children.

CHAPTER 6. CONCLUSIONS AND RECOMMENDATIONS

This research study examined the degree to which residents are exposed to $PM_{2.5}$ concentration levels due to wood smoke in a single-room house located in a First Nation community in Manitoba, Canada.   The parametric variations in this study included distance from the woodstove, height above floor level, duration of time the woodstove door remained open for refueling, time between refueling stages, and the effectiveness of various types of air purifiers in reducing the concentration levels. An extensive instrument calibration and error analysis were also conducted as a way of evaluating the accuracy of the data obtained through different instruments.

The main objective in the research study was to assess the degree to which residents are exposed to $PM_{2.5}$ concentration levels in a home. More specifically, answers to the following questions were sought:

1. How does the $PM_{2.5}$ concentration vary with **distance** from the woodstove, measured normal to the woodstove and at fixed heights from the floor?

2. How does the $PM_{2.5}$ concentration vary with **height** above floor level and at fixed distances from the woodstove?

3. How does varying the amount of time the woodstove door remains open during the refueling process affect indoor $PM_{2.5}$ concentration levels?

4. How might the duration of time the woodstove door remains closed between refueling sessions affect $PM_{2.5}$ concentration levels?

5. How effective are air purifiers in reducing $PM_{2.5}$ concentration levels?

6. How accurate are the instruments used in monitoring $PM_{2.5}$ concentration levels?

7. Can the cost associated with monitoring $PM_{2.5}$ concentration levels be reduced without compromising accuracy?

The following section summarizes the results from the study by addressing each of the above questions.

6.1   Summary of Findings

How does the $PM_{2.5}$ concentration vary with height above floor level and at <u>fixed</u> distances from the woodstove?

The estimated 24-hr $PM_{2.5}$ concentrations, normalized to concentration levels recorded by instruments located at 2'-5.5" above floor level, increased with height above floor level. The largest changes in concentration were recorded at distances 3' and 18' from the woodstove (see Table 5.1 and Figure 5.1). On average, for every foot increase in height, there is an 11-15% increase in $PM_{2.5}$ concentrations.

How do $PM_{2.5}$ concentrations vary with distance from the woodstove, measured normal to the woodstove and at <u>fixed</u> heights from the floor?

Unlike the results related to $PM_{2.5}$ concentration as a function of height above floor level, it appears that there is no consistent linear effect on $PM_{2.5}$ concentration as a function of distance from the woodstove. On average, however, concentration levels increased with distance away from the woodstove peaking at a distance of 8' and began to dissipate at distances further away (see Figure 5.2). Only a slight increase in average $PM_{2.5}$ concentrations (~15.5%) was recorded

towards the center of the room, at 8' from the woodstove, when compared to the position closest to the woodstove, 3'. The highest increase in concentration at 8' (38% higher than that at 3') was recorded by the Dasttrak II instrument located at 4'-5.5" from the floor level. At 18' from the woodstove all values had dropped considerably from their peak values at 8' (ranging approximately between -11% and +10% of the values recorded at 3').

Over the course of a burn cycle, it appears as if the particulate matter rises and travels along the ceiling and then curls back down upon reaching the back of the room. Instruments in the back of the room detected rising concentration levels earlier than instruments located closer to the fire.

Because there is no clear consistent pattern on how $PM_{2.5}$ concentrations change as a function of distance from the woodstove, no clear strategy can be recommended for residents to avoid high concentrations of $PM_{2.5}$ solely based on distance from the woodstove alone.

How does varying the amount of time the woodstove door remains open during the refueling process affect indoor $PM_{2.5}$ concentration levels?

All tests conducted without the use of an air purifier showed continuous cumulative $PM_{2.5}$ concentration levels after each refueling stage. It appears that the longer the woodstove door remained opened for refueling, the higher was the $PM_{2.5}$ concentration.

The following observations were made using the average value of $PM_{2.5}$ across all instruments located at various distances from the woodstove and at a fixed height of 3'-5.5" above the floor level, just prior to refueling as a benchmark (defined in this study as concentration R):

a) Tests Without an Air Purifier:

Prior to the first refueling, the concentration level R_1 varied from 0.041 mg/m^3 to 0.1422 mg/m^3 for all four, different door-opening durations of 5s, 15s, 30s, and 45s. These values are between 1.2 and 4.1 times the acceptable EPA limit of 0.035 mg/m^3.

With each subsequent refueling stage, however, the PM$_{2.5}$ concentration values R_1, R_2, R_3, and R_4 increased dramatically with the highest increases recorded during the 45s door opening refueling stage with PM$_{2.5}$ concentration values over 19 times their initial values.

b) Tests with an Air Purifier:

The average concentration levels R_1 prior to the first refueling when an air purifier was used varied from 0.0314 mg/m^3 to 0.1301 mg/m^3. These values range from 0.9 to 3.7 times the acceptable EPA limit of 0.035 mg/m^3.

With each subsequent refueling stage, the concentration values continue to dissipate, with the exception of one case (one corresponding to the 30s refueling period) with the values reaching as low as 41% of their initial value for the case of the 45s refueling stage (see Table 4.9). These concentrations are only 1.5 times the acceptable EPA limit of 0.035 mg/m^3.

How effective are air purifiers in reducing PM$_{2.5}$ concentration levels?

The results show that the use of air purifiers incorporating HEPA filters proved beneficial in reducing PM$_{2.5}$ levels significantly faster than if no air purifiers were used. Spikes in PM$_{2.5}$,

recorded immediately after the woodstove door was opened, quickly dissipated by the air purifier. The drop in PM$_{2.5}$ concentrations between refuelling stages would have been insignificant had no air purifiers been used. It took over 12 hrs for levels to drop from 0.5 mg/m^3 to 0.035 mg/m^3 without an air purifier.

The test house used in the study proved to have extremely high levels of PM$_{2.5}$ when the woodstove was in use. Even with the use of a HEPA air purifier, which in this study proved to be effective in reducing PM$_{2.5}$ levels, it would have been impossible for the PM$_{2.5}$ concentration to drop to EPA acceptable levels.

How accurate are the instruments used in monitoring PM$_{2.5}$ concentration levels?

To ensure reliability among the various instruments used to monitor PM$_{2.5}$, a number of calibration tests were conducted following strict procedures, as recommended by the instrument manufacturer. These tests, discussed in detail in Appendix A, included:

a) Tests to determine the Instrument Synchronization Factor (ISF),

b) Tests to determine the Flow Calibration Factor (FCF), using the Flowmeter – TSI 4140, and

c) Tests required to conduct the Gravimetric Analysis and to determine the Size Corrections Factor (SCF).

The results showed that, after calibrating the flow rate of the instruments and using the FCF, the difference in the data between instruments narrowed from 20% to less than 10% — well within the manufacturer's specification of 20%.

The SCF of 0.1867 obtained through the gravimetric analysis was used to scale all data obtained through the Dusttrak instruments in this study to accurately represent $PM_{2.5}$ due to wood smoke. The SCF is most accurate for particulate concentration of less than approximately 12 mg/m^3, which is the maximum measured concentration in the particulate 24-hr gravimetric experiment.

An error analysis of the data collected using various Dusttrak instruments revealed that the ISF error of -3.70% (and a standard deviation of 0.88%) lies within the instrument manufacturer's average error specification of approximately 20%. The ISF for each Dusttrak instrument was adjusted prior to each burn cycle by using the TSI Flowmeter 4140 to limit the error to approximately 3%.

Can the cost associated with monitoring $PM_{2.5}$ concentration levels be reduced without compromising accuracy?

In this study, the ability of inexpensive instruments to accurately measure $PM_{2.5}$ was assessed through comparison of the data captured during an identical burn cycle by the various instruments. More specifically, data collected by two inexpensive instruments, the Dylos DC1700 (approximate cost $500 CAD), and a UNI-T (approximate cost $200 CAD) were compared to data from a Dusttrak II 8530 (approximate cost $10,000 CAD). A detailed discussion of these comparisons is provided in Appendix B.

While the Dusttrak II instrument has strong internal logging capability, the Dylos DC1700 has limited such capability and UNI-T has none. The Dylos DC1700 proved to be inadequate in determining Dasttrak II equivalent $PM_{2.5}$ concentrations. However, the UNI-T data were well-correlated with that of the Dasttrak II data.

Using character recognition software, freely available on the Internet, data collected by the UNI-T instrument were analyzed and compared to data obtained through the Dusttrak II instrument. The results showed a strong positive linear correlation between the data from the two instruments. Based on this linear correlation, a calibration curve was developed in order to describe their relationship. When comparing the data from the Dusttrak II instrument to the modified UNI-T 938C data, the error was 5.56%.

6.2 Impact of the Present Study on Practice

Literature shows that exposure to wood smoke poses a major health problem to children (Torres-Duque, Maldonado, Pe´rez-Padilla, Ezzati, & Viegi, 2008). Safe limits for particulates have been recommended by both EPA and CSA. One such limit relates to $PM_{2.5}$ due to wood smoke which is set at 0.035 mg/m^3 over a 24-hr period.

Resident comfort, however, will always be a primary factor in determining how the woodstove door is used which will affect indoor $PM_{2.5}$ concentration levels. Additionally, residents will

decide where they will reside within their home. Because $PM_{2.5}$ is typically undetectable, residents' behavior will ultimately be swayed by their personal thermal comfort requirements in cold weather. Smoke tolerance will also be a factor. Although opening a window might help to aerate the room, thermal comfort will be affected negatively in colder climates. Ultimately, a better understanding of the health risks associated with $PM_{2.5}$ might help residents adopt a healthier woodstove practice without further affecting their comfort.

Establishing **particulate sustainability** is a new way of characterizing risk due to woodstove use. Achieving particulate sustainability should be set as a minimum requirement for reaching an acceptable $PM_{2.5}$ concentration due to wood smoke. Increasing the time between openings of the woodstove door for refueling will greatly affect the system's particulate sustainability only when HEPA air purifiers are used. If high $PM_{2.5}$ concentrations are allowed to enter the home during refueling, there is virtually no chance that these concentrations will dissipate prior to the next refueling without the use of an air filtration system or natural ventilation. One way to eliminate this problem is to use woodstoves that do not require opening their door for refueling, such as pellet woodstoves.

In this research study, a better understanding of how average $PM_{2.5}$ concentrations vary with height above floor level and distance away from the woodstove was formed. Residents, however, cannot be expected to change their day-to-day behavior in terms of their physical location in their home. A better solution would be to use a HEPA air purifier. Determining the optimum location of such an air purifier, especially given the effect height has on $PM_{2.5}$ concentrations, is left to future research.

The great majority of the published literature related to indoor levels of $PM_{2.5}$ deals primarily with ambient/outdoor concentrations entering homes, as this poses unique challenges in polluted cities. The primary strategy for mitigation in this case is to ensure that the windows and doors are properly sealed and not allow $PM_{2.5}$ to enter the home.

High indoor levels of $PM_{2.5}$ concentration can also be created through the use of woodstoves. Ensuring proper window and door sealants, in this case, would actually result in higher levels of $PM_{2.5}$ within the home. Identifying ways of controlling high concentration of indoor PM2.5 is essential in reducing the risks associated with respiratory illness, especially for those individuals most at risk, such as the elderly and young children.

Research studies examining $PM_{2.5}$ particulates created from indoor sources are limited. The majority of these studies explore the use of air purifiers as a way of reducing indoor $PM_{2.5}$ concentrations.

The present study is unique in a number of ways:

- It explores PM2.5 concentrations in a single room house as a function of **distance** from the source and **height** above floor level.
- It examines the amount of $PM_{2.5}$ released as a function of the **time the woodstove door remains** open during refueling.

- It examines the dissipation of $PM_{2.5}$ concentration using several different types of **air purifiers**, purchased in the open market.

- It explores a new way of characterizing indoor air quality conditions, when a woodstove is used, by determining the system's particulate sustainability.

- It demonstrates the effectiveness of a data logging method, SSOCR, when used together with an inexpensive $PM_{2.5}$ monitoring device. The SSOCR method, described in this thesis, provides a significantly inexpensive data acquisition system with potential uses in other applications where data are displayed digitally.

- It prescribes an error analysis and an instrument comparison method which can be used by future researchers to identify discrepancies in data obtained using different $PM_{2.5}$ measuring instruments.

This study also provides an extensive **critical review of the accuracy of the instruments** used in monitoring $PM_{2.5}$ concentration levels and examines ways of **reducing the cost** associated with monitoring concentrations without compromising accuracy.

To this end, this research study adds to the body of knowledge and has practical implications in the monitoring and mitigation of indoor levels of $PM_{2.5}$ due to wood smoke.

Assessing possible risks associated with such exposure is always beneficial. While the results from this study can lay the foundation for developing best practice guidelines for the use of

woodstoves in homes, additional future work is required to assess wood smoke dispersion in multi-room homes.

6.3   Limitations of the Present Study and Recommendations for Future Research

Although a number of parametric variations were included in this research study, there are certain limitations which must be considered in any future work. These include:

- **The time between refuelling stages.** Although the time elapsed between refuelling remained fixed at approximately 20 min, it would be valuable to test for $PM_{2.5}$ concentrations at longer periods and determine the particulate sustainability with or without the use of an air purifier.

- **The number of refueling stages.** The burn cycles conducted in this study were limited to three or four refuelling stages. It would be interesting to examine particulate sustainability during extended periods of time, with and without an air purifier, especially over a 24-hr period.

- **$PM_{2.5}$ sustainability using air purifiers over a 24-hr period.** It has yet to be determined whether air purifiers will maintain particulate sustainability for longer than a few hours, and more interestingly over a 24-hr burn test. Because the testing in this research study was limited to short burn durations, particulate sustainability can only be estimated.

- **The use of natural ventilation**. In this research study, all doors and windows were closed during testing and any reduction in $PM_{2.5}$ was accomplished through air purifiers.

It would be informative, in future work, to incorporate natural ventilation as a way of reducing $PM_{2.5}$ levels. This has to be balanced with thermal concerns in colder climates.

- **Effect of ambient $PM_{2.5}$.**   In this research study, the test house was purposely chosen to be located in a remote community where ambient levels of $PM_{2.5}$ are low. Future research should consider location as a parameter in determining the effect of $PM_{2.5}$.

- **Position and type of woodstove.** The woodstove in the test home used for this research study was positioned at the back of the room. The woodstove was also old and not well maintained. The $PM_{2.5}$ levels might be quite different if the woodstove was located in the centre of the room or if the woodstove was a new model or had been properly maintained. These parameters should be considered in future research.

- **Wood moisture content.** The moisture content in the firewood and kindling used in the burn tests, which were commercially available, was not measured. However, these were advertised as well seasoned. Identifying the variance in heating properties and particulate generation of firewood with varying moisture contents is left to future research.

- **Linkage between $PM_{2.5}$ and health.** While, in this research study, a number of issues related to distribution and monitoring of $PM_{2.5}$ in a home due to wood smoke were examined, the linkage between high concentrations of $PM_{2.5}$ and health was not. Future work should examine this issue using surveys and an examination of medical records of residents in homes with woodstoves who have reported respiratory problems associated with wood smoke.

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APPENDIX A: INSTRUMENT CALIBRATION AND ERROR DETERMINATION AND ANALYSIS

A.1 Air Flow Calibration and Gravimetric Analysis. Instrument Synchronization Factor (ISF): Comparing Data Obtained by the Various Dusttrak II Instruments

During testing, it became apparent that the Dusttrak II units displayed different values when logging data in the same environment at the same time during burn cycles. One unit, Dusttrak II #3, for example, could not be properly zeroed, providing erroneous results. It was decided to exclude this instrument from data collection and use only Dusttrak II #4, #5, #6, and #2.

In this Appendix, an extensive discussion is presented of the procedures used to calibrate the instruments employed for monitoring $PM_{2.5}$ concentration levels. These procedures were necessary in order to ensure the accuracy of the data obtained and, thus, the relevancy of discussion of the results that are presented in the body of the thesis. The calibration process followed strict procedures, as recommended by the manufacturer of the instruments, TSI, and as explained by its technical staff during numerous personal telephone communications.

The calibration procedures, which are discussed in this section, include:

a) Determining the Instrument Synchronization Factor (ISF)

b) Using the Flow Meter TSI 4140 to determine the Flow Calibration Factor (FCF); and,

c) Conducting a Gravimetric Analysis to determine the Size Corrections Factor (SCF)

These are discussed in detail in the following sections.

(a) Determining the Instrument Synchronization Factor (ISF)

This factor was used to describe the differences in data collected by concurrently running Dusttrak II instruments in an identical environment during a burn test. The ISF is defined as the ratio of the total $PM_{2.5}$ (mass) recorded by one Dusttrak II instrument to the total $PM_{2.5}$ (mass) recorded by another Dusttrak II instrument at the end of an entire test period.

In order to accurately determine the ISF, all Dusttrak II instruments being compared must be observing data that represent identical environments. That is, all external factors that may affect the results, such as the distance between instruments, are virtually eliminated and what remains is the intrinsic instrument differences described by the ISF. This way any comparison made can be characterized by the equipment itself and not by external factors.

To make a proper comparison between the results obtained from different Dusttrak II instruments the following procedure was followed.

a) The Dusttrak II instruments were placed as close together as possible and at the same height level.

b) An interval of data logging of at least 1 min was used to ensure accuracy of the results. If a smaller interval than the time it takes for the smoke to travel between the Dusttrak II instruments is used, then the data from that instrument may not be accurate.

c) The same test period for all the Dusttrak II instruments being compared was used.

The instrument manufacturer, TSI, recommends T-ing the in-take connections of the instruments for comparing their results. This means hooking up the nozzles of two instruments with a tube that is conjoined in the middle, like a T, as shown in Figure A.1.

Figure A.1. T-ing of Instruments

T-ing is supposed to allow the two Dusttrak II instruments being compared to draw from the same position in the environment, thus, negating the effect of distance between the two as a factor. At the end, this procedure did not prove useful as the ISF between the T-eed instruments increased when compared to the ISF when no T-ing was used.

(b) Using the Flowmeter TSI 4140 to determine the Flow Calibration Factor (FCF)

A flowmeter, Model TSI 4140, was used to check the flow rate of the Dusttrak II instruments, as shown in Figure A.2. All Dusttrak II instruments are supposed to have been calibrated annually by TSI to have a flow rate of 3.0 L/min. It was quickly realized, however, that air flow rate in the Dusttrak II instruments was actually not 3.0 L/min.

Figure A.2. Using a Flowmeter, Model TSI 4140,
to Check the Flow Rate of the Dusttrak II Instruments

The TSI 4140 flowmeter was used to calibrate the flow rate prior to and after every burn test in this study. This was done in order to determine how the internal flow rate of the Dusttrak IIs drifted over time. The Dusttrak II instruments have a built-in flow regulation system that can

maintain the set flow (internal FCF) to within ±5%. The period of time it takes for the flow meter readings to level off as the flow meter measures the flow rate was determined by logging flow rate data at 5s intervals over a 10-min period. When the flow meter was used, the Dusttrak II instruments were left to run for 3 min in "Flow Cal" to display more accurately the flow (L/min) of the internal Dusttrak II pump. The instrument error on the TSI 4140 is ± 3% or 0.06 L/min at a flow of 3.0 L/min (as specified in the TSI 4140 manual). The internal factory default setting on Dusttrak II instruments for the Flow Calibration Factor (FCF) is 1.00. This default setting is supposed to correspond to a flow rate of 3.0L/min. However, the FCF will have to be manually adjusted to ensure that the TSI Dusttrak II flowrate, as measured by the TSI 4140, is as close to 3.00L / min as possible.

During testing, the Flow Calibration Factor (FCF) on the Dusttrak II instruments was set between 0.50 (1.7 L/min) and 1.50 (4.0 L/min) at 0.01 intervals. Then, using the TSI 4140 flowmeter, the FCF was adjusted in the instruments to obtain the desired flow rate of 3.00 L/min.

It should be noted that the TSI Manual states that, if data collected by the TSI instruments in the same environment are within 20% of each other, then these data are within their specification. Upon calibrating the flow rate of the instruments, it was noticed that the difference in the data narrowed from around 20% to less than 10%. This error gap can be explained using the Instrument Synchronization Factor (ISF). Ultimately, a low ISF will be important when characterizing changes in $PM_{2.5}$ levels using several instruments running concurrently and

located in close proximity to each other and at different heights above floor level and distances from the woodstove.

Dusttrak II instruments #4, #5, #6 and #2 were primarily used in this study as they displayed similar values when measuring an identical environment while close to one another. More details on the ISF and implications of error are given in Section A.2.

(c) Conducting a Gravimetric Analysis to Determine the Size Corrections Factor (SCF)

A gravimetric analysis must be performed to ensure that the displayed values for $PM_{2.5}$ concentrations (mg/m^3) on the Dusttrak II instruments are accurate. The Dusttrak II instruments are originally calibrated using $PM_{2.5}$ from Arizona road dust. In this research study, the focus was on $PM_{2.5}$ concentration due to wood smoke particulate. In order to ensure that the $PM_{2.5}$ data are representative of those due to wood smoke, a gravimetric analysis is required. This analysis involved the following procedure:

Step 1: A filter is weighed without the particulate and then reweighed after exposed to the particulate. The difference represents the weight of the $PM_{2.5}$ wood smoke particulate. The Dusttrak II instruments have a built-in gravimetric filter cassette chamber which is used for the gravimetric test. The instruments used for the gravimetric analysis were Dusttrak II #4, #6, and #2. In this case, matched weight Zefon PVC 37 mm filter cassettes were used. These have a sufficient fiber pore size to trap particulate in the $PM_{2.5}$ range (5 μm). This filter was recommended by Zefon for use with $PM_{2.5}$ due to wood smoke. The filters were then sent to ALS Environmental, a commercial lab

located in Calgary, Alberta, to accurately weigh the particulate following a proper protocol.

Step 2: The Dusttrak II instruments display a total mass value over a fixed period of the test, which, in this case, was 24 hours. It is critical that the period chosen for the gravimetric test is equal to the period of time the Dusttrak II instruments are running while the filter cassettes are in the instrument. This is because the total mass of particulate matter captured by the internal filter cassette will be associated with a fixed period of time. If the cassette is removed prior to the completion of the test period, then the total mass of particulate will not be representative of the total value indicated on the Dusttrak II display. The Dusttrak II will continue to run with or without the presence of the filter cassette. It is, therefore, up to the user to ensure that the filter cassette is inside the Dusttrak II instrument for the entire duration of the test. Finding a ratio between data that do not have matched test periods will result in a widely skewed and inaccurate CSF.

Comparing the ratio between the total particulate mass obtained in Step 1 to the total mass of the particulate obtained in Step 2 yields the Size Correlation Factor (CSF). The CSF is simply a multiplicative scaling that is applied to all data that is collected by any Dusttrak II instrument with CSF set to 1.00. Alternatively, a user may input a CSF into the instrument to display accurate values on screen and in the logged data.

A.2   Error Determination and Analysis

The testing conducted in this thesis, involved four Dusttrak II 8530 model instruments. The accuracy of the data collected by these four instruments, either during the same burn test or during different burn tests, must be assessed. Analyzing the $PM_{2.5}$ data from each of the four Dusttrak II instruments will help determine the accuracy of these instruments. More importantly, any systematic error between the instruments must be established prior to any comparisons in order to better understand the effect of the parameters being examined from any systematic error between the instruments. In the present experimental investigation, $PM_{2.5}$ concentration in a single-room house was measured at various heights above floor level as well as at various distances from the woodstove. In addition, the effect of the time that the woodstove door was left open during refueling on the $PM_{2.5}$ levels was also examined.

Ensuring the Dusttrak II instruments are recording $PM_{2.5}$ concentrations accurately is essential in evaluating the effect of wood smoke on residents. Recommendations from TSI, the supplier of the Dusdttrak II instruments, include the following conditions designed to ensure instrument accuracy: zeroing prior to testing, gravimetric calibration, flow calibration, and cleaning the impactors at the start of every burn test. These conditions help ensure that data collected by different instruments are comparable. However, these recommendations are not sufficient to account for all the potential errors in the data values. The effect of systematic as well as random % error on $PM_{2.5}$ data collected under different circumstances will be estimated in the next section. These include errors associated with a shift in the Flow Calibration Factor (FCF) and the Instrument Synchronization Factor (ISF). These are also discussed in detail below.

Flow Calibration Factor (FCF)

Prior to each burn cycle test, the Dusttrak II instruments were zeroed (initial readings were set to zero) and were then calibrated with the use of the TSI 4140 Flowmeter to ensure that a 3.00 L/min airflow was set, as required by TSI. The TSI 4140 Flowmeter was used both at the start of each test as well as the end of each test. This was to determine how the internal flow of the Dusttrak II instruments drifted over time which could partially explain any systematic error in data recorded by different instruments.

The Flow Calibration Factor (FCF) is set in the Dusstrak II's User Interface (UI) control. Factory defaults have a value of 1.00, which is supposed to be representative of a 3.00 L/min flow under TSI's Lab conditions, where road dust is the particulate of study. The Flow Calibration Factor (FCF) can be set between 0.5 and 1.5 at 0.01 intervals.

In this research study, the FCF was set according to the TDI Flowmeter 4140 to a recommended value of 3.00 L/min. During calibration, the Dusttrak II instruments were left to run for 3 minutes in "Flow Cal" to display more accurately the flow (L/min) of the internal Dusttrak II pump. The instrument error on the TSI 4140 is ± 3% or ± 0.06 L/min at a flow of 3.0 L/min, as indicated in the TSI 4140 manual. The Dusttrak II built-in flow regulation system can maintain the set flow (internal FCF) to an accuracy of ± 5%. It should be noted that the TSI Manual states that the results are within TSI specifications if the data collected by the instruments in an identical environment are within 20% of each other.

Upon calibrating each of the four instruments for the proper FCF from their default settings, to reflect a consistent 3.00 L/min flow, the error between the readings from each of the instruments was reduced from around 20% to less than 10%.

Systematic errors, however, cannot be attributed to differences in the initial flow rate, which was set at a fixed 3.00 L/min for all the instruments. Differences in how the flow rate drifts during a test period between the various instruments can in fact contribute to additional systematic error in the data. Because the flow rates were measured prior to and also after each test conducted, the effect of the flow rate drift can be measured. An example of this flow rate drift is shown in Figure A.3 for Dusttrak II #6

As shown in Figure A.3, the flow rate for Dusttrak II #6 drifted between 2.95 and 3.1 during a 60-90-min test period and between 2.875 and 3.055 during a 12-hr test. On average, the drift between pre-flow and post-flow for this instrument was 1.25% and 2.34 %, respectively, for the two test periods.
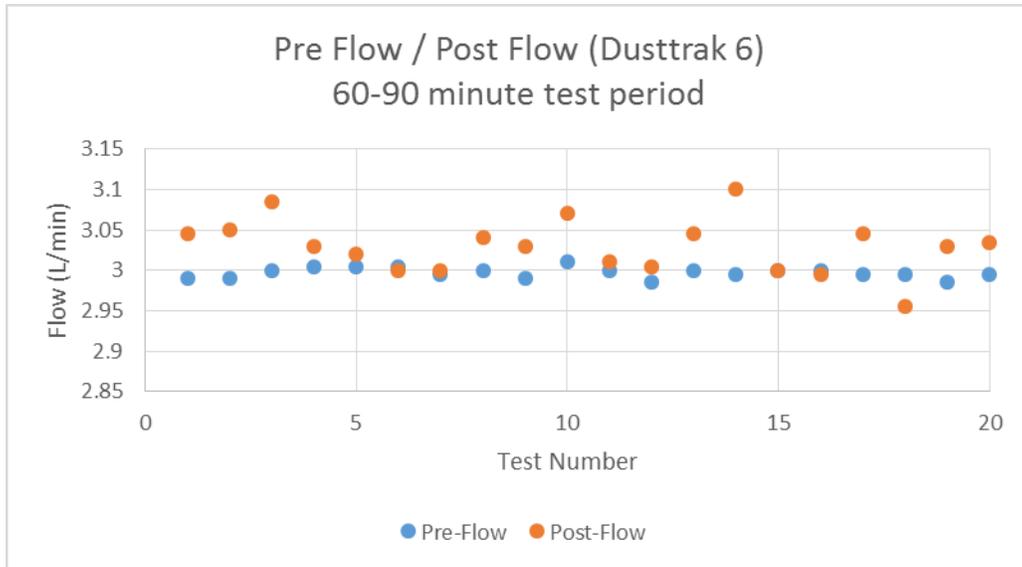
(a) Pre-flow/Post-flow for Dusttrak II #6 Over a 60-90-min Test Period
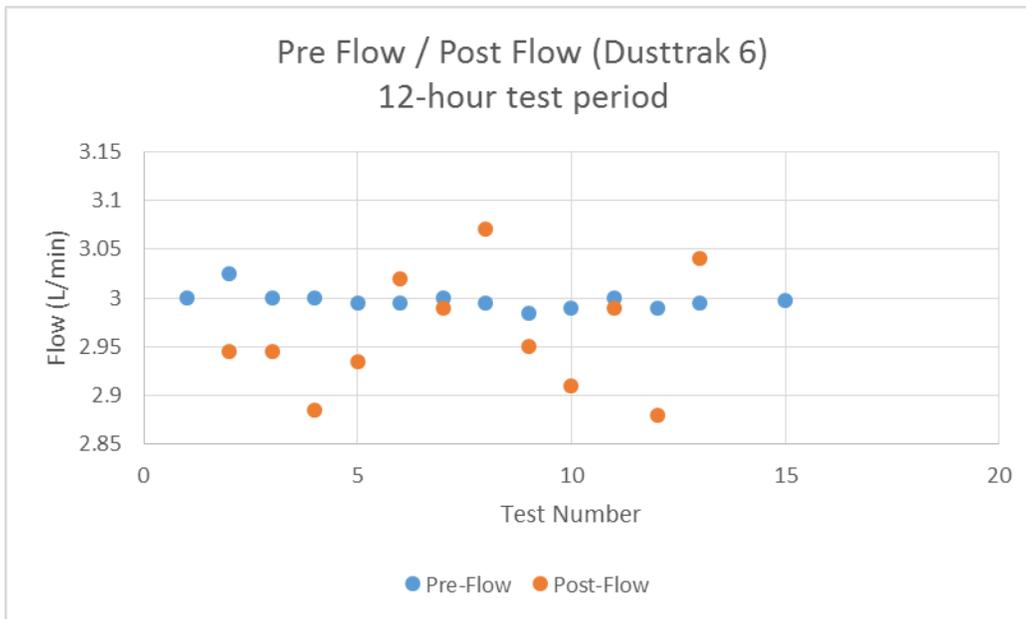
(b) Pre-flow/Post-flow for Dusttrack II # 6 Over a 12-hr Test Period

Figure A.3. Pre/Post Flow Results, as Detected by the TSI Flowmeter 4140.

(The results show that the drift in the flowrate is more significant for longer test periods. This translates into a larger systematic error.)

Although the FCF is fixed prior to a test burn cycle, by the end of the test cycle, slightly different

flow readings were recorded on the Flowmeter 4140. The errors, introduced due to flow drifting

over a single test period, are summarized in Table A.1.

Table A.1. Percent Error Due to Average Flow Drift Introduced Over Two Test Periods,
60-90-min and 12 Hrs, where Flows Measured after the Test Period Have Drifted
from their Initially Calibrated Value of 3.00 L/min .*

| | % Error in FCF Introduced over Two Test Periods due to Flow Drift | |
Dusttrak II #	60-90-min Test Period	12-hr Test Period
4	0.63	1.00
5	1.17	1.84
6	1.25	2.34
2	1.23	1.74

*As shown, as the testing period increases, the error introduced due to flow drift also increases.

Despite the drift in the flow rate, from the beginning to the end of a specific test, the error is still

within the range of the instrumental error of the TSI Flowmeter 4140 of 3% (or 0.06 L/min) or

the Dusttrak II Flow Rate error of 5%. When comparing data from one test day to another, it is

important to note that the Flow Calibration Factor (FCF) itself will drift from one test date to

another. This, in part, can be due to human error, such as not waiting sufficient time for the

Flowmeter to stabilize before setting the FCF on each instrument. A systematic FCF drift over a

3-month period may be attributed to environmental factors, which is outside of the scope of

present study. Thus, although some error can be introduced during testing, calibration of the

instruments prior to testing reduces the magnitude of this error that is within acceptable limits.

Since every Dusttrak II instrument was gravimetrically calibrated using a single burn-cycle test and since a fixed FCF per instrument was used, there could also be some error introduced due to environmental effects. This would be impossible to control if testing is continuous over an extended period of time. An example of the FCF drift over a 3-month period is shown in Figure A.4.
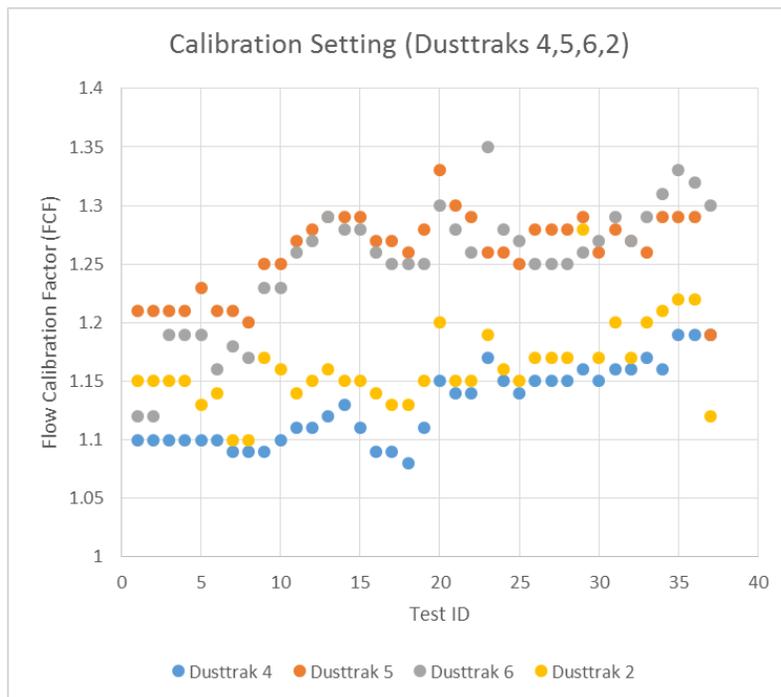
Figure A.4. Flow Calibration Factor (FCF) Shift over a Duration of 3-Months
(The First Test Conducted is Test #1 and the Last is Test #37)

A clear trend is noticeable in Figure A.4. It appears that, early in the experiment, the FCF remained relatively constant and then increased over time. The FCF shift over time was translated into an average % error for each Dusttrak II instrument in Table A.2.

Table A.2. Percent Error Due to Average Drift Flow Over a 3-Month Period

Dusttrak II #	% Error Over a 3-Month Period Due to FCF Drift
4	2.87
5	2.64
6	4.26
2	2.97

The average % errors due to drift in FCF over time are lower than the recommended value of 5% for the Dusttrak II instruments. The % error introduced due to FCF drift will only be significant when comparisons are made between results from instruments used in different burn-cycle tests, especially when these tests are conducted months apart. Despite the assurance by the TSI Flowmeter 4140 that the flow rate in each instrument is fixed at 3.00 L/min during the 3-month period, the FCF will drift over this time. If this drift is not accounted for, comparison between data from different burn cycles becomes less accurate. The values in Table A.2 can be used as a guide to establish the maximum effect due to this drift.

The errors due to drift in the air flow alone cannot explain systematic differences in results between Dusttrak II instruments during an identical burn cycle, which appear to be consistent throughout the testing. This remaining difference in the recorded data between instruments can be explained using the Instrument Synchronization Factor (ISF).

In examining $PM_{2.5}$ data collected simultaneously by four closely positioned Dusttrak II instruments, it was observed that there is a systematic deviation in results from one Dusttrak II

instrument to another. This was most predominantly evident in the data collected by Dusttrak II #4 which recorded lower $PM_{2.5}$ values than the other instruments whose results were closer to each other. The deviation can be characterized by a single multiplicative factor, the ISF.

If the ISF is not used to modify the $PM_{2.5}$ data, the intrinsic deviation in the data collected by each Dusttrak II can still be described through the introduction of an additional systematic error. According to the manufacturer of the instruments, TSI, when observing data over a long period of time, with at least a 1-min data logging interval, deviation in $PM_{2.5}$ data of 20% from each other is within specifications.

Data recorded by Dusttrak II instruments #4, #5, #6 and #2 for the purpose of determining the ISF or a systematic % error introduced due to instrument differences are shown in Figure A.5.

For the sake of the analysis, the data collected over a 12-hr period, shown in Figure A5, were split into two periods. The first period lasted 15 min and 30s and is characterized by a single spike. The other period is the time after the spike and is characterized by a steady drop in $PM_{2.5}$ concentration. This period is broken into five time segments. The % error for these segments are shown in Figure A.6. These include: one 8-hr period of the 12-hr burn cycle, Figure A.6(a); the first 15 min of the burn cycle, Figure A.6(b); approximately one 8-hr period following the spike in concentration, Figure A.6(c); approximately one 6-hr period following the spike in concentration, Figure A.6(d); and one 3-hr period following the spike in concentration, Figure A.6(e).
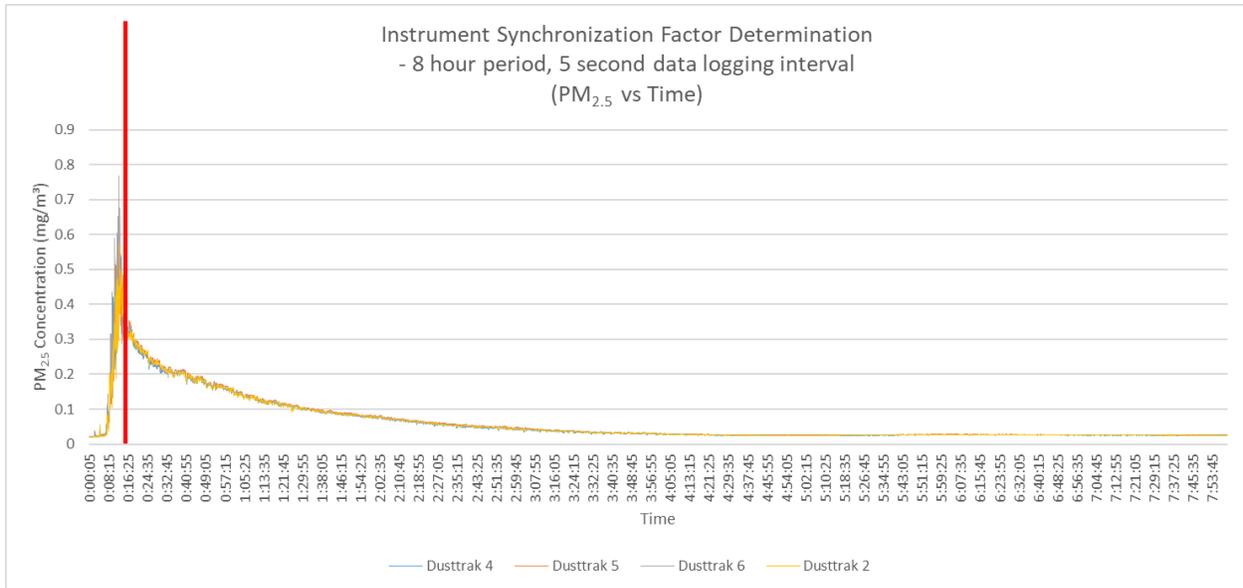
Figure A.5.   PM$_{2.5}$ Concentration Recorded by Dusttrak II #4
Over a 12-hr Period with a 5sData Logging Interval
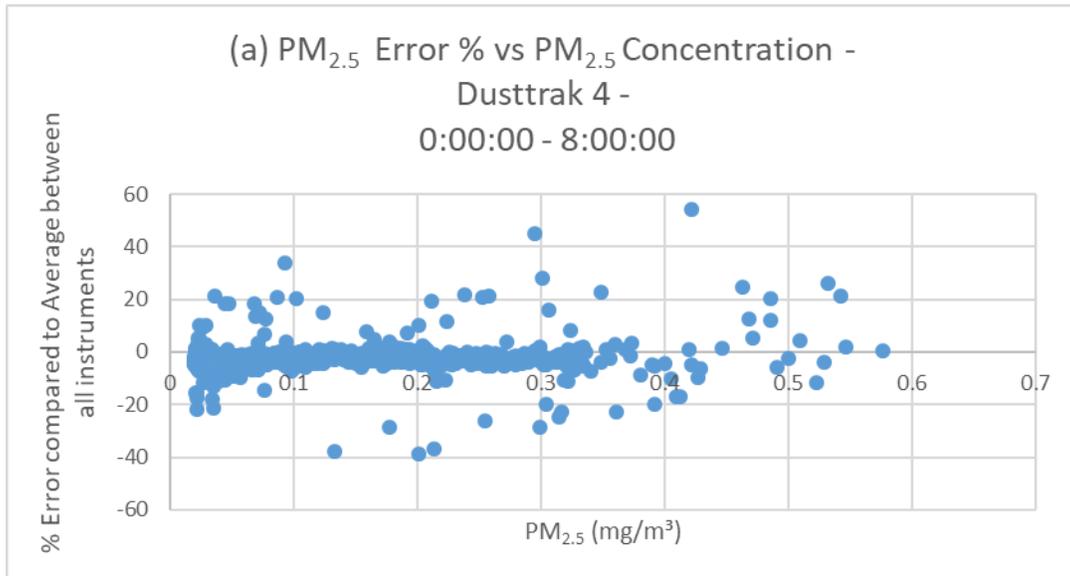
(a) Percent Error vs. PM$_{2.5}$ Concentration Detected by Dusttrak II #4
when Compared to the Average Concentration Detected Simultaneously by All
Instruments During the First 8 hours of a 12-hr Burn Cycle
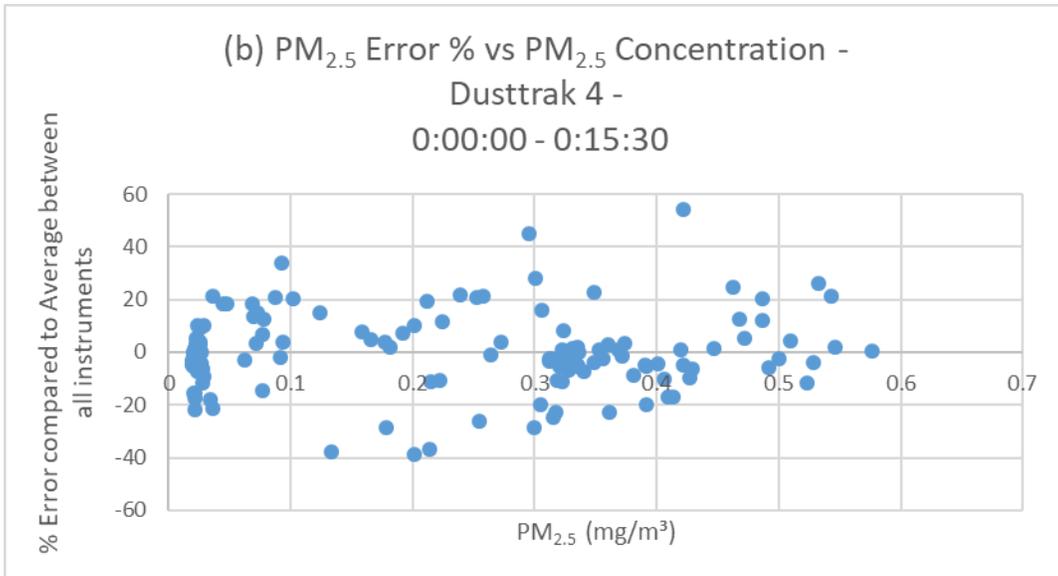
(b) Percent error vs. PM$_{2.5}$ Concentration Detected by Dusttrak II #4
During the First 15 min and 30s of a 12-hr Burn Cycle
and Before the Concentration of PM$_{2.5}$ Spikes.
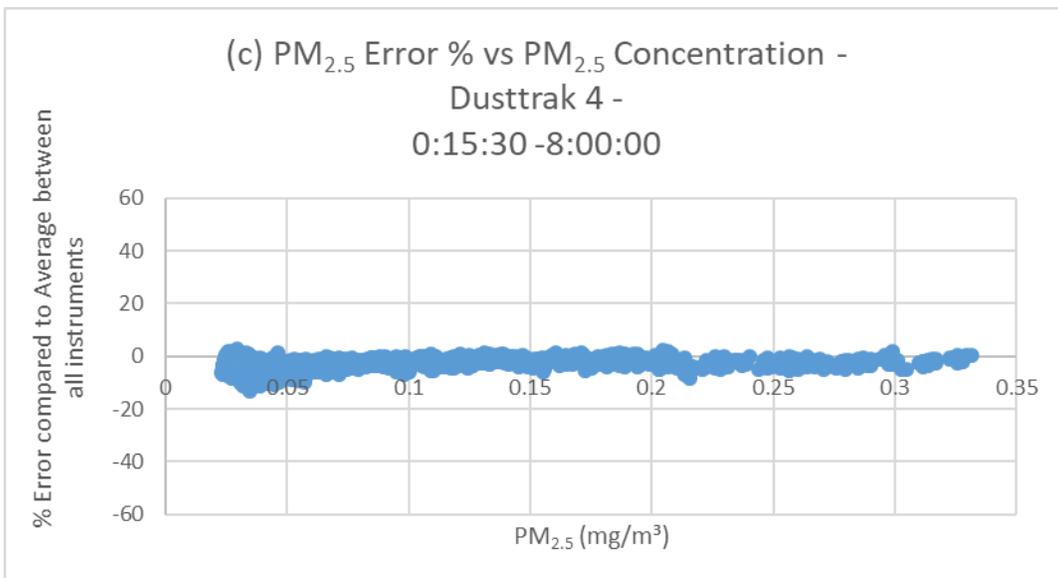
(c) Percent Error vs. PM$_{2.5}$ Concentration Detected by Dusttrak II #4
After the Spike in PM$_{2.5}$ Concentration Over Approximately an 8-hr Period
Starting at 15-min 30s into the Burn Test.
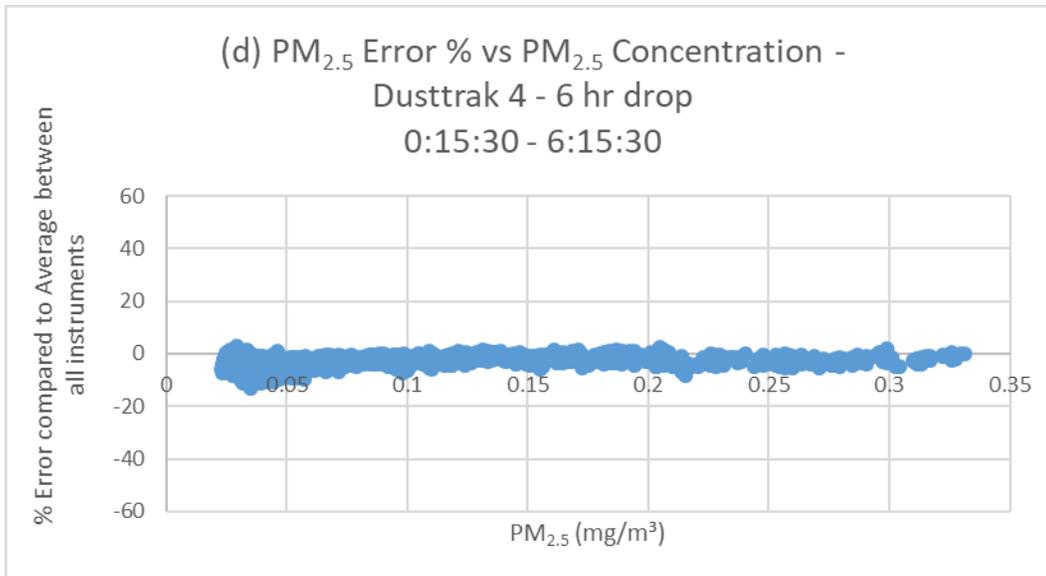
(d) Percent Error vs PM$_{2.5}$ Concentration Detected by Dusttrak II #4
Over a 6-hr Period After the Spike in PM$_{2.5}$ Concentration
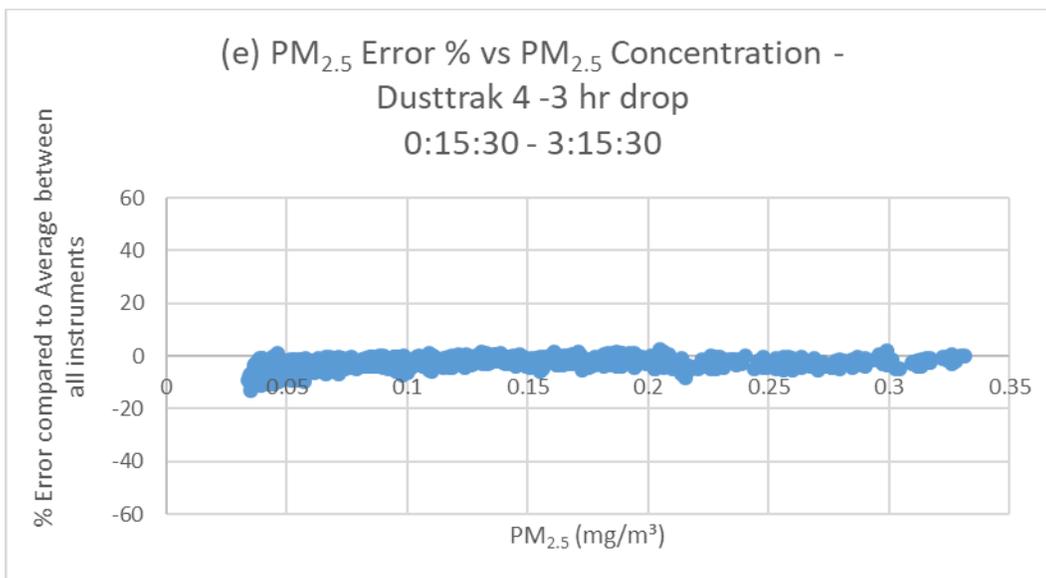
(e) Percent Error vs PM$_{2.5}$ Concentration Detected by Dusttrak II #4
Over a 3-hr Period After the Spike in PM$_{2.5}$ Concentration

Figure A.6.   Percent Error vs. PM$_{2.5}$ Concentration Detected by Dusttrak II #4 when Compared
to the Average Concentration Detected Simultaneously by all Instruments
During Various Time Periods in a Single-Burn Test

As shown in Figure A.6(b), the error varies between -40% and +55%. The range is quite large and appears to be random. This implies that the absolute maxima in PM$_{2.5}$ concentrations detected by the various Dusttrak II instruments cannot be directly compared to each other.

As shown in Figure A.6(c), for approximately 8 hrs after the spike in PM$_{2.5}$ and the drop in concentration, the % error between the Dusttrak II #4 instrument and the average of all instruments becomes well defined, varying between -5% and +5%. The range in the % error trend represents the random error, which itself can systematically increase as PM$_{2.5}$ increases. The position of the trend with respect to the y-axis is indicative of a systematic difference in readings by Dusttrak II #4 when compared to the readings by the other instruments.

The % error due to Instrument Synchronization along with the % error in the standard deviation for Dusttrak II #4 over several periods during a burn-cycle test from ISF Data (Figures A.6(a) to A.6(e)) based on a 5-second data logging interval are listed in Table A.3.

The Instrument Synchronization Factor (ISF) can be estimated as a calibration factor derived from simple division of the total averages detected by each instrument. However, it is best reflected as an average % error in order to compare it with the instrumentation error.

Table A.3. Percent Error Due to Instrument Synchronization Over Several Periods
During a Burn Cycle Test from ISF Data (Figures A.6(a) to A.6(e))
Based on a 5s Data Logging Interval

Burn-Cycle Period	Test-Period Section (hr: min: s)	Average % Error in $PM_{2.5}$ Concentration Recorded by Dusttrak II #4 with Respect to the Average $PM_{2.5}$ Concentration Recorded by of all Dusttrak II Instruments (systematic error – trend position)	Error % Standard Deviation Dusttrak II #4 vs. Dusttrak II Average (random error – trend range)
a) Entire Burn Cycle Test	00:00:00 – 24:00:00	-2.81	2.82
b) Pre-Spike	00:00:00 – 00:15:30	-0.68	12.75
c) 8-hr Post-Spike	00:15:30 – 08:00:00	-2.88	1.63
d) 6-hr Post-Spike (ISF % error)	00:15:30 - 06:15:30	-3.05	1.71
e) 3-hr Post-Spike	00:15:30 - 03:15:30	-2.99	1.91

As shown in Table A.3, the ISF % error, for the test case examined, between 00:15:30 –

08:00:00 (hr: min: s) was estimated to be -2.88 % with a standard deviation of 1.63 %.

On average, the % error is well within TSI's instrument error when steadily decaying data are

being compared. Being that data in this thesis are not characterized only by a single large spike,

the results can be compared using this % error.

The results prior to the concentration spike at 00:15:30 (hr: min: s) show a radically increased %

error. This indicates that with a 5-s internal data logging interval $PM_{2.5}$ spike values measured by

the Dusttrak II instruments cannot be directly compared at any one time without a significant error being introduced. This is true even between closely positioned instruments running simultaneously. This is not to say that each Dusttrak II instrument is recording inaccurate concentrations of $PM_{2.5}$. Instead, this is an indication that the data logging interval should be increased. This is especially important when comparison is made between different types of instruments, such as with the Dylos DC1700 or UNI-T.

Because the data logging interval in this test was 5s, the % error introduced is higher than it should be when making comparisons between the various Dusttrak II instruments. It is clear that 5 seconds is not long enough to eliminate spatial differences between the Dusttrak II instruments, despite the fact that they were positioned as close to each other as possible during the burn test. When instruments are separated by significant distance, such as in the height and distance testing, it cannot be assumed they are each experiencing near identical environments. This is described through the variation in 2s-interval data with distance or height. As the data logging interval increases, the separation of the instruments plays less and less of a role. This effect was confirmed when examining the gravimetric data which, in contrast, have a 1-min data-logging interval, as shown in Figure A.7. The gravimetric data shown in Figure A.7 were obtained during a 24-hr burn test using Air Purifier I. By comparing the gravimetric data to the ISF data, the effect of an increase data logging interval becomes more evident.
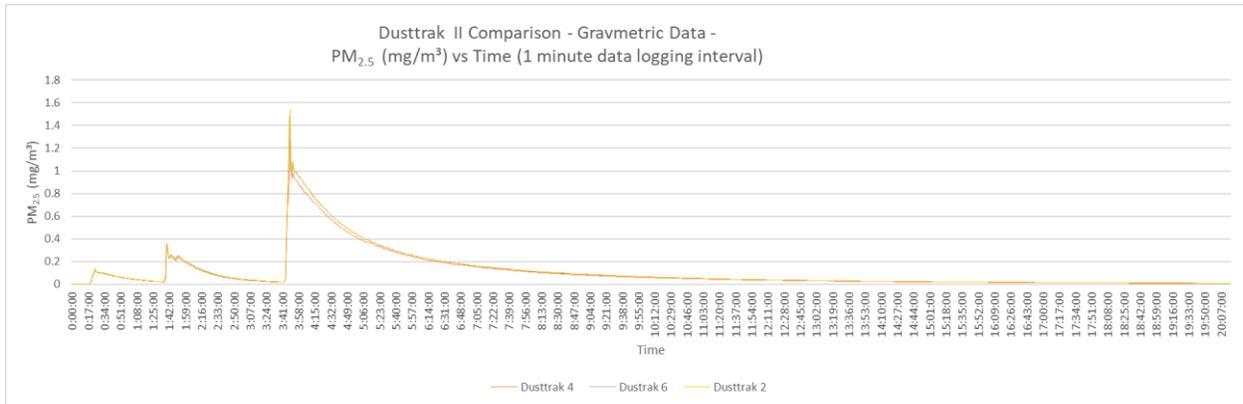
Figure A.7. PM$_{2.5}$ Concentrations from Dusttrak II #4, #6, and #2 as a Function of Time
During the Gravimetric Test

(Dusttrak II #5 did not properly work during the gravimetric test and its
data are not shown in this figure. The area of interest begins at the 00:03:15 mark
when the final PM$_{2.5}$ spike takes place.)

The % error between PM$_{2.5}$ concentration recorded by Dusttrak II #4, shown in Figure A.7, and

the average PM$_{2.5}$ detected by all Dusttrak II instruments during the gravimetric test is shown in

Figure A.8 for various test periods over the 24-hr burn test.
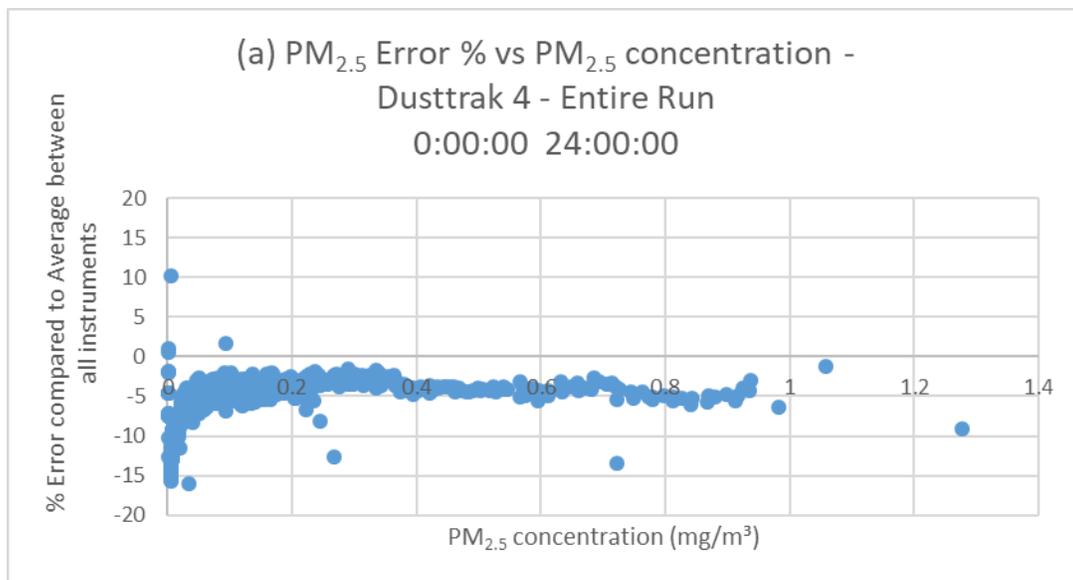
(a) Percent Error as a Function of PM$_{2.5}$ Concentration for Dusttrak II #4 over a 24-hr Period

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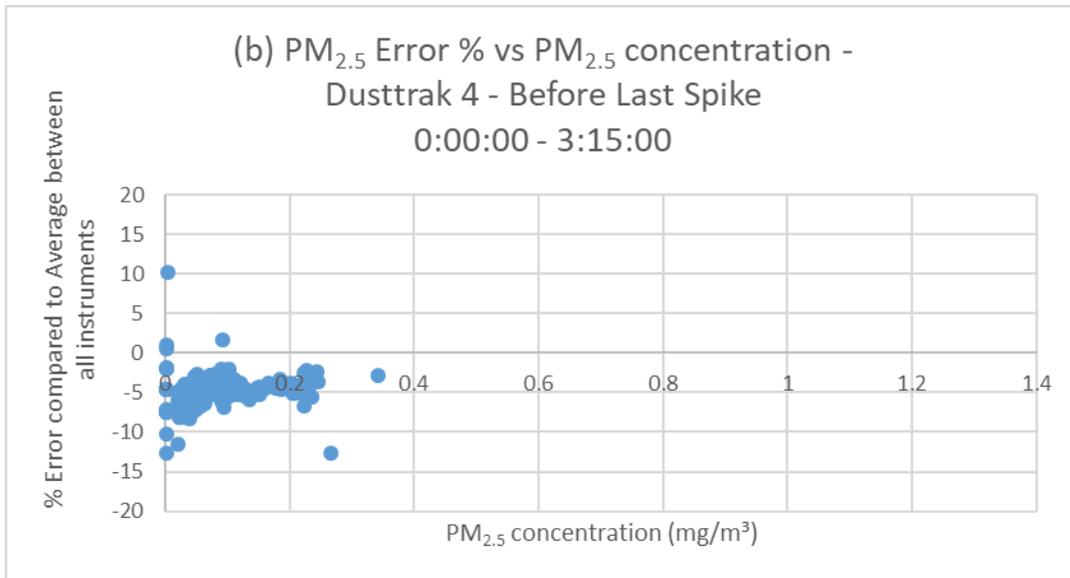
(b) Percent Error as a Function of PM$_{2.5}$ Concentration for Dusttrak II # 4 Over a 3-hr Period
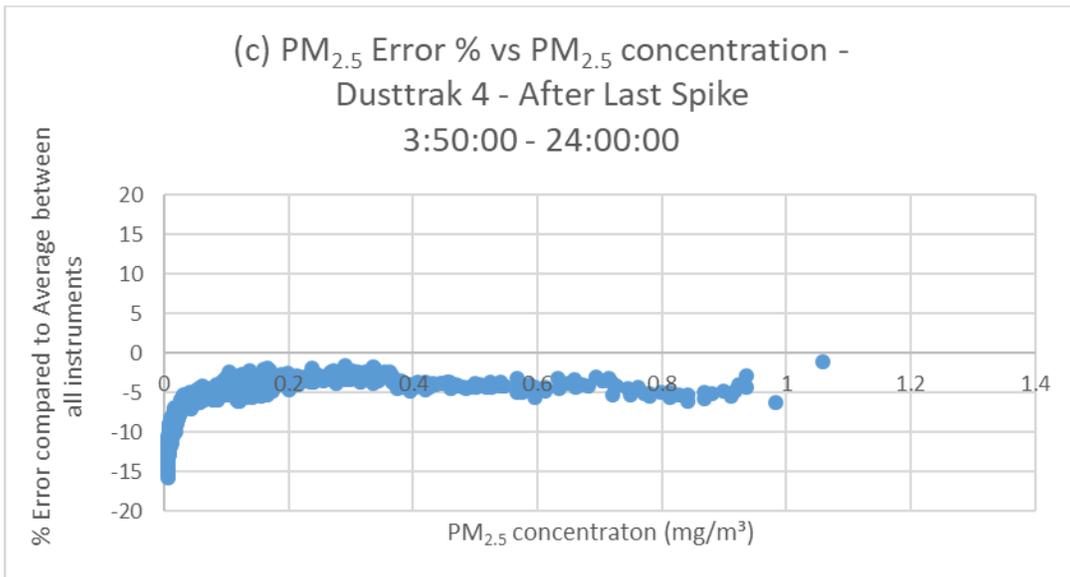
(c) Percent Error as a Function of PM$_{2.5}$ Concentration for Dusttrak II #4 over a 20-hr Period After the Last Spike in Figure A.7

.
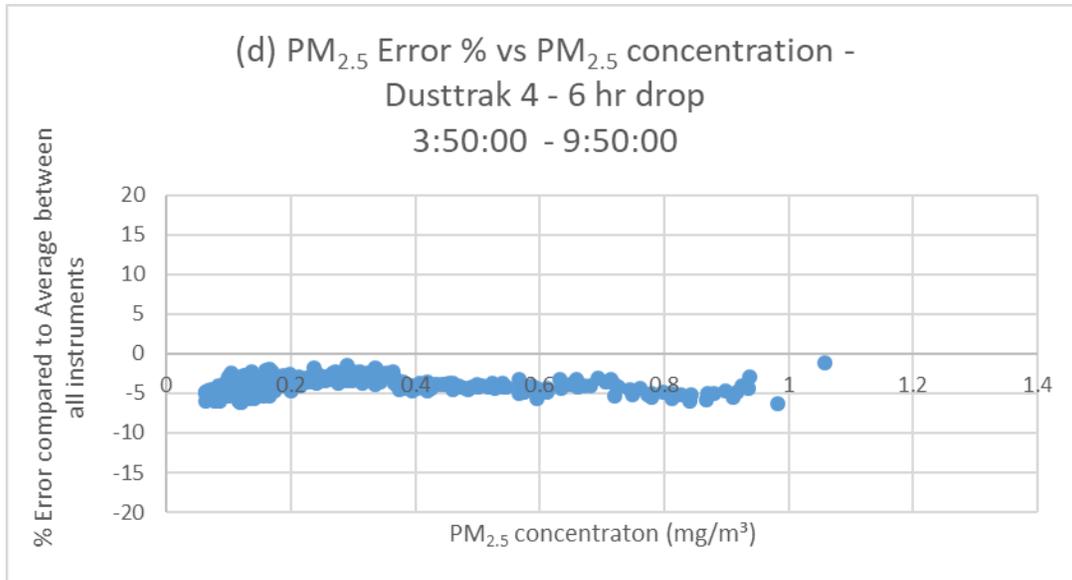
(d) Percent Error as a Function of PM$_{2.5}$ Concentration for Dusttrak II #4 over a 6-hr Period
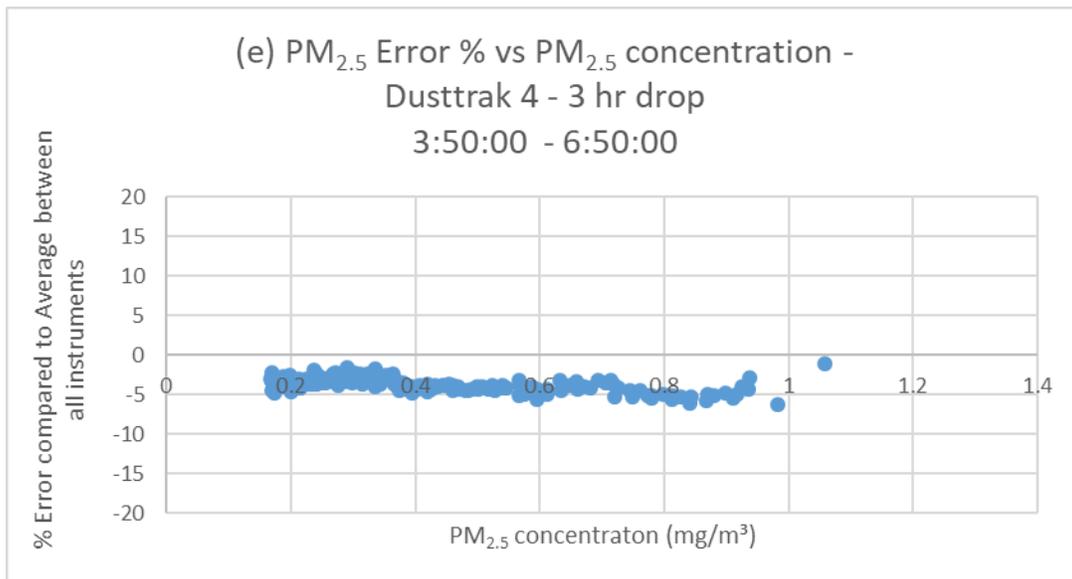
(e) Percent Error as a Function of PM$_{2.5}$ Concentration for Dusttrak II # 4 over a 3-hr Period.

Figure A.8. Percent Error Between the PM$_{2.5}$ Concentration Recorded by Dusstrak II #4
and the Average Gravimetric Concentration Recorded by all Instruments
for Various Test Periods Within a 24-hr Burn Test

Including the spikes in the data of $PM_{2.5}$ during the gravimetric test, as was done with the ISF data, created outlier points. However, when only data after the final drop in $PM_{2.5}$ were used, a clearer picture emerges of the % error. In this case, after the spike in concentration at 00:03:50, there are no more spikes in $PM_{2.5}$. Instead, a steady drop in concentration was observed. The post-spike data in Figure A.8(c) display a clear trend in comparison to the pre-spike data shown in Figure A.8(b). This behaviour is similar to what was observed in the ISF data Figure A.6 (a, b, c, d, and e).

Identifying the ISF % error is not sufficient to establish a total error in the results, as the weight of the systematic error must first be considered. A decrease in random % error, evidenced by the decreased in the deviation range from the average, follows a rather well-defined curve. Increasing the internal data logging interval allows for less randomness in the particulate present in the air around each instrument. Thus, the 5-s internal data logging interval used in the ISF data is not sufficient to assume that $PM_{2.5}$ concentrations detected simultaneously by the various instruments is representative of an identical environment. This is indicative of a slow permeation of the particulate in the room. Even with a well-defined curve, the results are different when compared to the ISF data set. This is probably due to shifting of the flow rate in the Dusttrak II instruments during testing, as explained earlier.

Prior to testing, each Dusttrak II instrument was zeroed. This means that the value of $PM_{2.5}$ was set to zero prior to testing. This was accomplished with the use of a zeroing tool supplied by the manufacturer of the Dusttrak II instruments TSI. It involved blocking the flow of particulate into

the Dusttrak II instrument completely. However, it was observed that, after testing, the readings no longer returned to zero, indicating a residual reading other than zero. This was evident in some of the tests involving air purifiers, where negative $PM_{2.5}$ concentrations were detected at the end of the burn-cycle test. However, $PM_{2.5}$ levels are supposed to approach zero as the air had been sufficiently purified with the use of HEPA filtration. Any shifting in readings from zero will have a greater effect in % error when $PM_{2.5}$ concentration values are small.

As shown in Figure A.8(c), the % error is high at low $PM_{2.5}$ ranges, regardless of the presence/absence of $PM_{2.5}$ spike in the data. This is due to initial zeroing shifting in each instrument, which could create substantial systematic errors when the total detected $PM_{2.5}$ is within an order of magnitude. This explains the trend shown in Figure A.4 near the low $PM_{2.5}$ concentrations. Removing these data by limiting the data range to either 6 or 3 hrs, which will exclude the low $PM_{2.5}$ concentrations near the end of the data cycle, results in a more logical % error due to the fact that the systematic error introduced by a drifting zero shift is virtually eliminated.

Because zeroing the instruments was only done at the beginning of each test, a process that was not repeated during testing, any shifting of the initial setting affects data with a long test period where a large number of low $PM_{2.5}$ concentrations are recorded. Also, in long period tests with single $PM_{2.5}$ spikes and subsequent dropping off of the concentration, a significant range of the data will be of sufficiently low $PM_{2.5}$ values. In this case, the data would be adversely affected by any zeroing calibration accuracy. Removing these low $PM_{2.5}$ values, such as those shown in

Figure A.8(d) and (e), leads to an increasingly well-defined % error between the Dusttrak instruments. This % error in turn can be uniquely attributed to instrumental differences.

The overall % errors in $PM_{2.5}$ concentration during the gravimetric tests are shown in Table A.4. When compared to the data shown in Figure A.3(b), the results in this table show that an increase in internal data logging interval, from 5s to 1 min, leads to a noticeable decrease in the systematic as well as the random error.

Table A.4. Overall Percent Error in $PM_{2.5}$ Concentration During the Gravimetric Test Based on a 1-min Logging Interval

Test Period Gravimetric Data Based on a 1-min Data Logging Interval	Test Period Section (hr: min: s)	Average Error % Dusttrak II #4 vs. Dusttrak II Average (True) (systematic error – trend position)	Error % Standard Deviation Dusttrak II #4 vs. Dusttrak II Average (True) (random error – trend thickness)
a) Entire Test	00:00:00 – 24:00:00	-7.26	3.29
b) Pre-Spike	00:00:00 – 03:50:00	-5.47	3.13
c) Post-Spike - until End	03:50:00 – 24:00:00	-7.59	3.22
d) $PM_{2.5}$ Dropping for 6 hrs after Spike	03:50:00 - 0 9:50:00	-5.53	1.77
e) $PM_{2.5}$ Dropping for 3 hrs (ISF % Error)	03:50:00 - 06:50:00	-3.70	0.88

The results in Table A.4 show that the predominant source of error is due to the zero-shifting effect. If near-zero values are removed from the data set in the post-spike range (after 03:50:00 - hr: min: s), the % error and its standard deviation are both reduced significantly.

Comparing the results for Test Periods (a), (b), (c), (d), and (e), shown in Table A.3, with the test results for Periods (a), (b), (c), (d), and (e) in Table A.5, it is observed that when a 1-min data logging interval is used instead of a 5-s data logging interval, the effect of excluding the spikes in $PM_{2.5}$ concentration has no effect on the error. In fact, in this case, the pre-spike % error from the gravimetric data set, Test Period (b) in Table A.5, is actually smaller than the post-spike % error in Test-Period (c), as is the standard deviation of the % error. This again shows the importance of setting a longer data logging interval for more accurate results. The huge increase in % error at low $PM_{2.5}$ concentration values in this data set is predominantly due to zero-shifting at the low $PM_{2.5}$ range near the end of the test.

When this zero-shifting behaviour is accounted for by removing low $PM_{2.5}$ values, there remains a systematic and intrinsic instrumental difference. This is what is defined as the ISF % error, as discussed earlier. In the case of the gravimetric data, Dusttrak II #4 had an ISF % error of -3.70 with a % standard deviation error of ±0.88 %. This error, seen in Test Period (e) in Table A.4 can represent the same effect that multiplicative calibration ISF can, but now it can be compared with the instrumentation error.

Because of the zero-shifting behaviour, there is no linear trend in % error, except when the near zero $PM_{2.5}$ concentration values are excluded from the error analysis. In this sense, the ISF % error of -3.70 with a % standard deviation error of ±0.88 % can accurately describe the Dusttrak II #4 data differences due to the ISF alone when comparing its data to that of other Dusttrak II instruments.

Through this error analysis, it was determined that the increase in % error as PM$_{2.5}$ concentration decreases (Figure A.8(c)), is attributed to thezero-shifting behaviour of the instruments. For PM$_{2.5}$ data that do not approach near zero values, such as those typically found in a home with a woodstove on, and with at least a 1-min data logging interval, any systematic differences between the instruments can be attributed to instrumental differences, as described by the ISF. Once the % error is determined for each Dusttrak II instrument, then its effect can be taken into account when comparisons in data need to be made between various Dusttrak II instruments.

ISF data based on a 5-s data logging interval are not consistent among the various instruments due to the fact that particulate cannot evenly permeate the space between the devices within 5 s. Even if the Dusttrak II instruments are positioned as closely together as possible, they can still be detecting different PM$_{2.5}$ concentrations. This difference can be minimized if the data logging interval is increased to 1 min in the gravimetric test. This, in turn, allows for a more accurate determination of the % error due to the ISF alone by helping to distinguish it from other % error sources, such as a zero-shifting behaviour which occurs concurrently between the instruments. Regardless of the data logging interval, a zero-shifting effect will always lead to an increasing % error as PM$_{2.5}$ values approach near-zero values.

In the current study, the ISF test period was only 8 hours (see Figure A.5). This, however, was not long enough of a time to observe any significant zero-shifting effect as PM$_{2.5}$ concentrations did not drop as low as when a 24-hr test duration was chosen. Increasing or decreasing the data logging interval, however, has no effect on the % error due to zero-shifting of the Dusttrak II instruments.

Through the analysis of the gravimetric and ISF data sets it was observed that any comparisons made between the data obtained by Dusttrak II #4 and the other Dusstrak II instruments is assumed to be accurate within an average error that is described as the ISF % error. Thus, for Dusttrak II #4, the ISF % error was estimated to be -2.88 % with a standard deviation error of 1.63 % (see Table A.3). This was chosen because no $PM_{2.5}$ concentrations, throughout the test periods, were low enough for the zero-shifting effect to play any significant role.

For the gravimetric test, the ISF % error was estimated to be -3.70 % with a standard deviation error of 0.88 % (Test Period (e) in Tale A.5). This was chosen from the approximate 3-hr drop-off as it best avoids any zero-shifting effects.

The ISF % error can be estimated by carefully selecting the proper period range from either the gravimetric or ISF data sets. However, this alone cannot explain instrument differences, potentially affected by exterior parameters such as: flow rate drifting over a test period, environmental differences affecting FCF calibration, or zero-shifting behaviors. Instead, it acts as a guide to establish and quantify intrinsic differences in detected $PM_{2.5}$ by the different Dusttrak instruments in this study. The ISF % error will be a factor regardless of the situation or environment, most likely because of instrumentation limitations.

The estimated % error (-2.88% and standard deviation of 1.63%) from the 1-min logging interval during the gravimetric test, and the % error (-3.70% and standard deviation of 0.88%) from the 5-second logging interval ISF data set, both lie well within TSI's maximum average error

specification of, approximately, 20% when comparing $PM_{2.5}$ data between Dusttrak II instruments. This large percentage is most likely to be observed if all instruments, when left at the default FCF value of 1.00, will have different flow rates and, depending on when they were last calibrated by TSI, will bias the results. Additionally, this 20% error covers an extensive range of particulate matter sizes and types, not specific to wood smoke $PM_{2.5}$. The TSI Flowmeter 4140 has an instrumentation error of 0.06 L/min or 3%, and the Dusttrak II can only maintain its flowrate within an error of 5%.

The Instrument Synchronization Factor (ISF) % error can help explain any intrinsic differences in the Dusttrak II instruments when compared to the average $PM_{2.5}$ value from all instruments. This error can in fact be a direct result of differences in each Dusttrak II's Flow Calibration Factor (FCF), which is adjusted through measuring each instrument's air flow with the TSI Flowmeter 4140. The FCF value is initialized in each instrument prior to any testing for $PM_{2.5}$. Adjusting each Dusttrak II instrument's FCF increases overall accuracy as it allows for consistency between instruments. This consistency is established by using the TSI Flowmeter 4140, as each Dusttrak II is assured to have a 3.00 L/min flow within instrument error.

A faulty flowmeter, however, will result in systematic error in $PM_{2.5}$ concentration values, as each FCF is set up initially only once, and remains fixed during the entirety of the test. This could partially be responsible for the ISF % error between the various instruments. This is corroborated by the fact that the ISF % error is consistent with instrument error of the TSI Flowmeter 4140. Additionally, flow rate drifting can vary across each instrument over the duration of a test. This, in addition to any error introduced upon manually setting the FCF, will

together help explain some errors due to ISF alone. Additional intrinsic parameters differentiating the Dusttrak II $PM_{2.5}$ data can also help explain the ISF error. It should be noted that the ISF % error for each instrument is assumed to remain constant throughout the 3-month testing despite ICF being adjusted prior to each burn cycle to ensure a flow rate of 3.00 L/min. In future work, it is recommended that more than one ISF test be conducted using, at least, a 1-min data-logging interval. The ISF % error in this study can only be used as a guide for understanding how much error can be expected when comparing $PM_{2.5}$ data from different Dusttrak II instruments used to observe an identical environment.

Adjusting the ISF for each Dusttrak II prior to each burn test by using the TSI Flowmeter 4140, in theory, will limit the instrumental error of the TSI Flowmeter 4140 (to approximately 3%). Inaccuracies in initializing ISF prior to a test, in part, will affect the ISF. Additionally, flow rate drift differences between instruments will also affect the ISF. Because the ISF in this study was within range of the Flowmeter 4140 instrumental error, an assumption of consistency is made between these error values. More importantly, their magnitude remains comparably low when compared to TSI's recommended maximum ISF % error of approximately 20%, which would hold true even when a Flowmeter 4140 is not utilized.

When discussing flow rate drifting over a single burn test period, and inaccuracies in setting the ISF prior to each burn test, the % error introduced should not be confused with that due to ISF drift over the 3-month experiment. The ISF itself, as discussed earlier, will change over a 3-month period, despite consistently representing a 3.0L/min Dusttrak II flowrate as measured by

the Flowmeter 4140 each day of testing. This ISF drift is due to environmental parameters outside the scope of present study.

Summary of Error Analysis Results

In this research study, errors in data recorded during testing, under different circumstances, by various Dusttrak II instruments are compared. These errors and the conditions governing these are summarized in Table A.5. Because the total % errors are kept within instrument specifications, differences in $PM_{2.5}$ concentrations due to human behavior can be measured and accurately characterized. Understanding the nature of the ISF for Dusttrak II #4, for example, helps explain why it typically detects lower $PM_{2.5}$ values than the other Dustrak II instruments across all tests. It is not solely due to its position during testing. It is also due to intrinsic instrumental differences described through its ISF % error. If the percent error between the instrument data, due to the ISF, drifts over time due to air flow drifting during testing, such a drift will be reflected in its standard deviation. This phenomenon was tested over the duration of a single test in both the gravimetric burn cycle (see Figure A.7) and the ISF burn cycle tests (See Figure A.5). To check if the percent error due to the ISF, also called the ISF % error, drifts significantly from test to test, data from both cycles are compared. Comparing both ISF % errors, determined from the gravimetric data set and the ISF data set, yields results that lie within range of each other, and therefore, are consistent with each other, as demonstrated by the low values of % error and standard deviation of the error (see Figures A.3 and A.4). Thus, for this research study, the drift in ISF % error from test to test was assumed to be negligible. Thus, the ISF % error will be treated as a fixed error and applied when comparisons are made between data collected by different Dusttrak II instruments from a single burn test.

Table A.5. Summary of Errors and Conditions Leading to these Errors

Comparing data from:	Example	Tests where comparison is required for analysis	Estimated % Error (PM$_{2.5}$ Concentration)
Identical Dusttrak II Instruments During Different Tests	Data from Dusttrak II # 4 (Test 1, Day 1 vs Dusttrak II #4 Test 2, Day 2)	- Woodstove Door Open Every 5, 15, 30, and 45 s - Use of Air Purifier. Data with Single PM$_{2.5}$ Spike and Subsequent Drop off in PM$_{2.5}$ Concentration	**For Testing Period of 24 hr:** Total % Error = FCF Drift % **For Testing a Single Parameter's Effect (Resident Behavior):** Total % error cannot be measured. Possibly too high. Difficult to attribute PM$_{2.5}$ to differences in human behaviour. This could be attributed to external parameters which are outside of the scope of the present study. A sharp increase in PM$_{2.5}$ concentration cannot be attributed solely to an increase in time duration the woodstove door is opened. In such cases, patterns or trends in PM$_{2.5}$ concentration or PM$_{2.5}$ dissipation should be examined within each test, as opposed to direct PM$_{2.5}$ comparisons between tests.
Different Dusttrak II Instruments During the Same Test	Dusttrak II # 4 (Test 1, Day 1 vs Dusttrak 5 Test 1, Day 1)	- Distance tests (Instruments at 3, 8, 13, and 18 ft from the Woodstove) - Air Purifier open Duration Testing: Comparing PM$_{2.5}$ at Each Distance (3,8,13,18 ft) - Height tests (Instruments at 2, 3, 4, and 5ft +5.5 in)	**Testing PM$_{2.5}$ Concentration Over 24 hrs:** Total % error = ISF % Error (affected by inaccuracies in initial FCF setting, differences in flow rate drifting between instruments, and other intrinsic differences between Dusttrak II instruments) **Testing for the Effect of a Single Parameter Related to Resident Behaviour:** ISF % Error + Zero shifting (only significant at near 0 PM$_{2.5}$ concentrations) Zero shifting does not play a significant role in real world woodstove data over a 24-hr period as the magnitudes affected are insignificant.
Different Dusttrak II Instruments During Different Tests		- No such tests were conducted in this study. However, this would be typical for testing PM$_{2.5}$ concentration in a home where different Dusttrak II instruments may be used on different days.	**For testing Period of 24 hr:** If only the average of PM$_{2.5}$ over a 24-hr period is of interest, it is reasonable to assume: Total % error = ISF % error + FCF Drift % **Testing for the Effect of a Single Parameter Related to Resident Behaviour:** This is not useful when the effect of varying a single parameter is being tested. Error will be simply too big and the results will be ambiguous.

APPENDIX B: INSTRUMENT PERFORMANCE CALIBRATION AND COMPARISON

B.1 Introduction

The overall goal of this section of the thesis is to compare the effectiveness of measuring $PM_{2.5}$ by various devices, one expensive device: the Dusttrak II, and two comparably less expensive devices: the UNI-T and the Dylos DC1700. The question that this part of the thesis intends to answer is: Are the measurements sufficiently accurate to use significantly less expensive instruments for recording indoor $PM_{2.5}$ concentration levels due to smoke from a woodstove?

Testing the feasibility of using cheaper equipment as a possible alternative to the more expensive Dusttrak II is an important task, as it will open the door for much broader $PM_{2.5}$ studies where multiple simultaneous data acquisition setups may be required.

In this research study, direct comparisons were made between two cheaper instruments that measure $PM_{2.5}$, the UNI-T and the Dylos DC1700, with the more expensive Dusttrak II 8530. Some important information about these instruments is provided in Table B.1.

Table B.1. General Instrument Comparison Information

Instrument	Built-in Data-Logging Capabilities	Price (CAD $)
Dusttrak II 8530	Yes	~10,000
Dylos DC1700	Yes	~500
UNI-T	No	~200

In this research study, less expensive instruments with limited or no data logging capabilities were compared to a more advanced and expensive equipment with built-in data logging capabilities. In order to directly compare any two instruments' ability to collect data, several conditions must be considered: data logging intervals must be identical, instruments must be located at the same location and data must be collected at virtually at the same time, and data must be logged and tabulated in digital form.

The Dylos DC1700 does have internal data logging capabilities. However, it's limited to a minimum of a 1-min interval setting, and it measures particulate as a count per cubic foot, compared to other instruments that measure particulate in microgram per cubic meter. The only possible effect the difference in units may have is dependent on how the mass of the airborne wood-smoke particulate will change during the testing period due to environmental changes in the room. This will be discussed later in this section.

The UNI-T instrument does not have any built-in data logging capabilities. Therefore, a character recognition method was implemented in order to digitize the data for the sake of directly comparing it to the gravimetrically calibrated Dusttrak II 8530 data. The character recognition method used is described below.

Seven Segment Optical Character Recognition (SSOCR) Procedure:

In this research study, three software packages were used to recognize numerical characters from a periodically captured series of webcam images of instruments' displays, and subsequently convert them into useful data in the form of an excel file. The three software packages used in succession in this research study were:

a) **SkyStudioPro**: a freemium windows software used to capture the raw images with a webcam at a fixed period automatically.

b) **ImageMagick**: a well-recognized open-source image editing software used to adequately transform the raw webcam images into a form that is ideal for character recognition. More specifically, the following transformations were implemented in ImageMagick: image rotation, colorspace manipulation, threshold fixing, and image cropping.

c) **SSOCR (**Seven Segment Optical Character Recognition): the main software used for character recognition developed by Eric Auerswald (2018), a member of the github open source community.

The method used is summarized in Figure 6.1, where the instrument display being observed is the UNI-T PM$_{2.5}$ meter. It should be noted that changing the 2D image orientation with ImageMagick (rotation) was not required for the webcam images of the UNI-T meter, as its 3D orientation was sufficient for SSOCR.
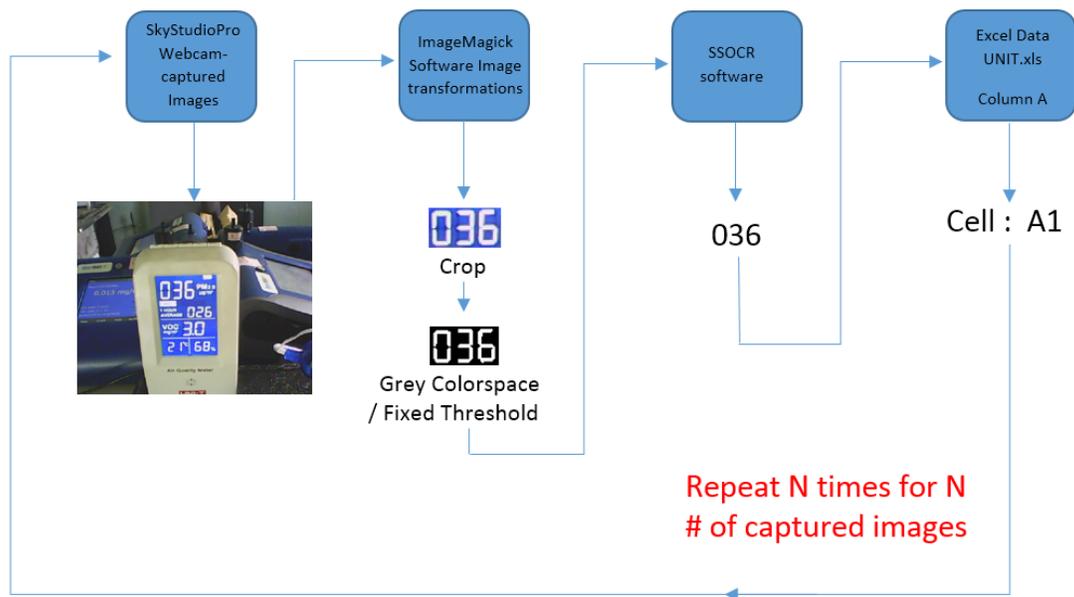
Figure B.1. Visual Summary of the SSOCR Method

This strategy was also used for tabulating flowmeter data from a TSI 4140 which, although not used in the present comparison of instruments, helped form a better understanding of the limitations of the SSOCR method.

Detailed Method of Character Recognition:

First, in SkyStudioPro, the user interface will appear as shown in Figure B.2.
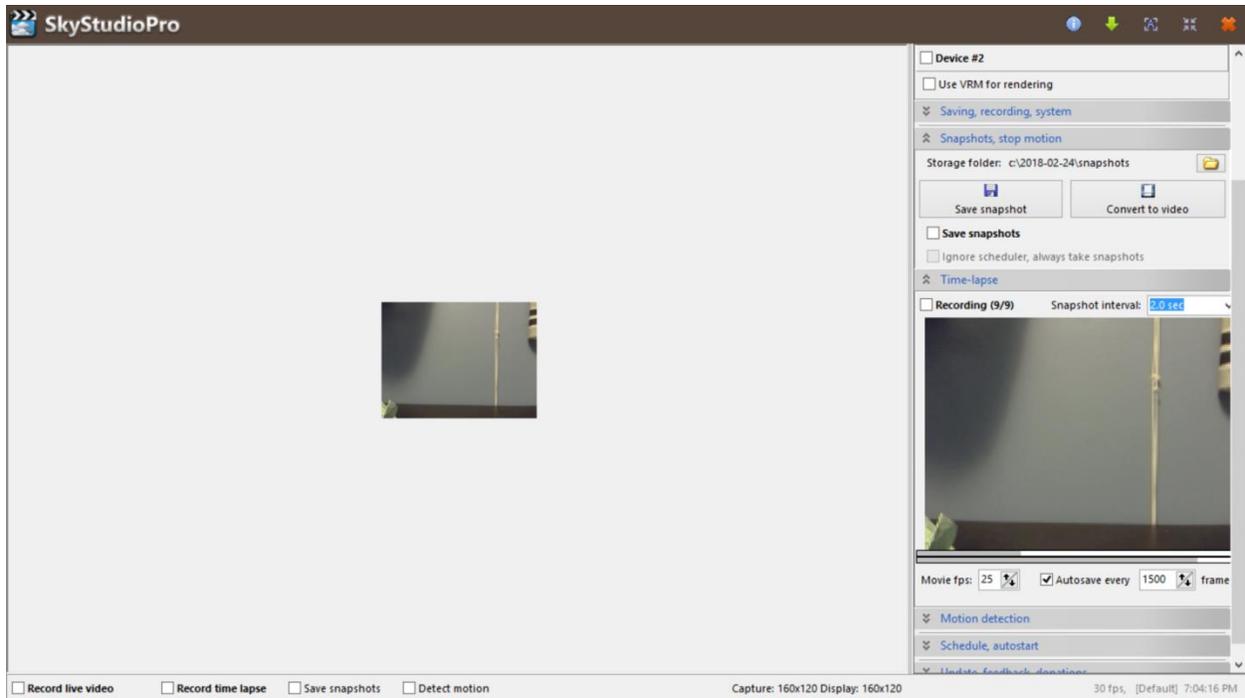
Figure B.2. SkyStudioPro User Interface in Windows

To capture an image every N seconds, under time lapse, the interval time is set to N, which will vary depending on the test being conducted. Pressing "recording" will begin the process of saving images to a given storage location shown in the UI (User Interface).

In hindsight, this was not the optimal webcam software choice, as both the SSOCR and ImageMagick software packages can perform this same task, but with a more optimal configuration potential.

The results from three individual tests were analyzed through the implementation of the three software packages in succession: SkyStudioPro, ImageMagick, and finally SSOCR. Two

different instrument displays were observed and captured: TSI 4140 FlowMeter, observed for two tests, and the UNI-T PM$_{2.5}$, observed for one. Images were captured by pointing a webcam at the display and using SkyStudioPro software to take images every one or five seconds. The directory in which they were saved is displayed in the UI.

An example of an image captured for each instrument is given below.

Figure B.3. UNI-T: Images Captured Every 5s

Figure B.4. Image Showing Flowmeter TSI 4140
Measuring Air Flow of the Dusttrak II 8530 in L/min

The number of images for each test are summarized in Table B.2.

Table B.2. General Information of Various Tests Conducted for the Use of the SSOCR Method

Test ID	Instrument Used	Image Capture Interval	Test Period	Number of Images Captured
1A	UNI-T	5s	~3 hr 25 min	2464
2A (second test)	TSI 4140 Flowmeter	5s	~10 min 10s	122
2B (first test)	TSI 4140 Flowmeter	1s	~10 min 1s	601

Instead of manually going through each image and logging each value by hand, it is much more efficient to utilize a software to recognize the characters, especially when thousands or even tens of thousands of images are captured. This numerical recognition technique could be used to digitize data where seven segment characters are shown on an instrument displays, but internal data logging capabilities are not present. Usually, when data logging is necessary, a lot more money must be spent for more advanced instrumentation. If there is an instrument that can display digital values on a well-lit screen, but lacks internal data logging capabilities, and there is

no risk of the instrument being moved, perhaps then data logging truly may be possible without spending any additional funds.

This concept was tested in this research study for the three tests listed in Table 6.2. Because all laptops are now equipped with webcams, and the hardware used is extremely lightweight, this method is a very attractive alternative as a data logging system for a wide range of instruments in science and engineering.

In this particular study, a USB webcam was used for easier maneuverability. However, its low resolution had eventually hindered the software implementation from accurately determining the numerical value for the images analyzed. A better alternative would have been to use a smartphone with an HD camera.

There are several critical factors in the SSOCR software package's ability to accurately recognize numerical values from the images it reads. These factors include: 2D image orientation, 3D image orientation, border locations, resolution, color space, and noise. These are discussed below.

2D Image Orientation:

The 3D images captured are displayed on a computer screen as a 2D image. This is in effect a projection onto a 2D plane, which is the screen. Images can be rotated along the XY plane by any degree amount. An example would be a rotation of 180 degrees seen as follows.
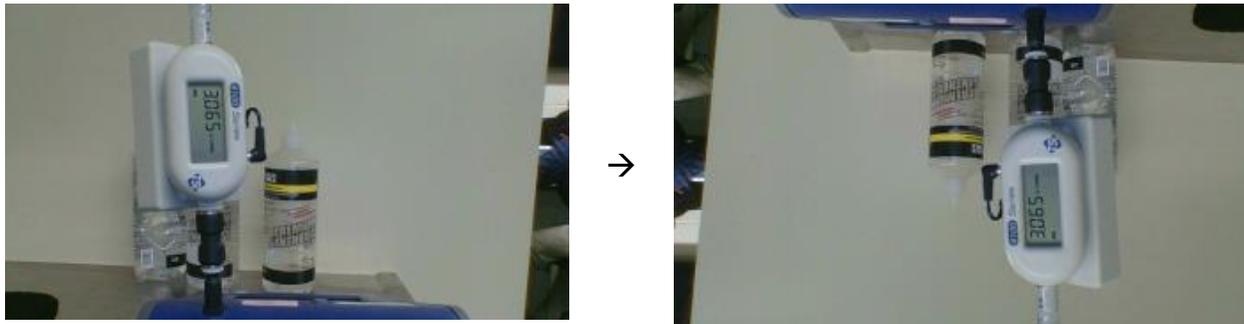
(a) Original Image   (b) Rotated Image

Figure B.5. Rotation of a 2D-Image of a Flowmeter TSI 4140

(Rotating the 2D images will prove useful in aiding the SSOCR software to accurately recognize and effectively decipher the observed image and subsequently recognizing the proper depiction of a numerical value.)

The SSOCR software used in this study requires an upright image to accurately decipher the pixels that make up the number's image space into a digital numerical value. The reason for this is in the nature of the software. Each number is broken into parts and each part is analyzed and compared to an internally pre-trained idea of what makes up a seven-segment digital number. This idea is shown in the following example:

Figure B.6. Example of a Captured Instrument Display

(This would be read accurately by SSOCR as: 4 3 1 4 3 2.
Image source: Auerswald, 2018)
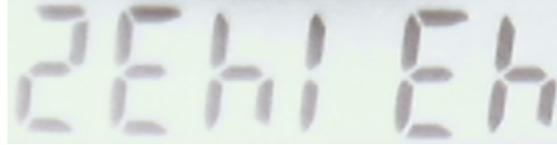
Figure B.7. A 180° Rotation of the Image in Figure B.6

(This image would be inaccurately read by SSOCR as: 2 * * 1 * *.
The * indicates no known number recognized.
Image source: Auerswald, 2018)

The software, which lacks human intuition, has to be told what to look for. Simply changing the 2D image orientation can result in many inaccurate and blank values, which only hinders the entire process. Ultimately, if a data set had over a million images, there would be no absolute way to ensure that each value is properly depicted by SSOCR. In this sense, it is critical to ensure the images are rotated once to reflect an upright position. However, each time the image is altered in any way, the image quality degrades. All image manipulation should be attempted with as little degradation in the image quality as possible. This is especially critical when the image resolution of the original raw image is low. In some cases, the decimal place is not picked up by the software, as it blends in with the numbers it sits in between. A simple "replace all" in excel can easily fix this problem.

Image Space and Boundary:

The image boundary is important as it defines what exactly the image is. If the SSOCR software, which is looking specifically for seven segment digits, is given the entire image including the flow meter casing and the background, it will come up with blank results. This is due to the fact that the numbers that lie in the original image boundary are lost in all the other pixels. Despite

the fact that the human eye can easily detect the image in the original image, regardless of its surroundings, the SSOCR software doesn't realize what it's looking at and fails. This is why it is important to clearly define the boundaries of the numbers on the display that ultimately need to be digitized.

Cropping the image is a requirement for the SSOCR software to properly identify that it is observing numbers at all. This must be done as close to the boundary of the numbers on the display as possible, without clipping them even the slightest. The entirety of the number on the display is crucial for the software to properly match and determine a numerical value. It is important to remember that, depending on the location of a specific digit in the whole number being observed, it could be cut off. This is due to minute differences in the 3D display orientation as well as in cropping coordinates. It should be noted that differences in cropping coordinates and sizing will be limited by a minimum difference of single pixel. One must also be aware that any possible border irregularities created by the cropping, even down to a single pixel width, could fool the SSOCR software into thinking there is a "1" either at the beginning or the end of the number in question. The user must make sure that the cropping allows for a uniform and clear boundary around the digits that make up the number on the display being recorded.

Image Resolution:

Image resolution is an important factor in ensuring accurate number depiction from the image data. During the initial planning for establishing a method for determining digital numeric values for numbers found in a series of images captured by a webcam, **image resolution** was not

hypothesized to be as big a factor in the accuracy of the resulting data as it was later discovered to be. Because the manipulation of the image orientation results in lossful compression, it is important to ensure the highest possible initial image resolution to preventing the image from becoming inaccurately deciphered into digital number values.

Both the type of the webcam used and the software employed may limit the image resolution for all initial captured set of pictures. Another critical factor that may affect image resolution of the resulting images, which is to be fed into the SSOCR software, is the position of the webcam with respect to the display of the device being captured. The closer the camera, the clearer the numbers become in the images of the display, and consequentially, the final image resolution, after cropping each image, will be higher. In addition, a higher final image resolution will counter the effect of noise on the accuracy of the numeric recognition. Another limitation of image resolution is the size of the hard drive used to store the images. If a raw uncompressed image is over 50 megabytes large, and ten thousand images are captured, over 500 gigabytes of space would be required. This would scale past any feasible hard drive capacity if significantly more images were required to be digitized without any form of compression.

3D Display Orientation:

When an image is captured by the webcam, the orientation of the display setup being captured, will affect the resulting image in a way that cannot solely be explained through a diminished image resolution. Ideally, the direction of the webcam should be normal to the plane of the digital display being recorded, as shown in Figure B8. In this case, the 2D projection of the

digits, which make up the image captured, will not appear distorted and be optimally legible. Any shift from this ideal 3D-display orientation condition may cause a distortion that could lead to further issues, once the SSOCR software package is used to decipher the images.
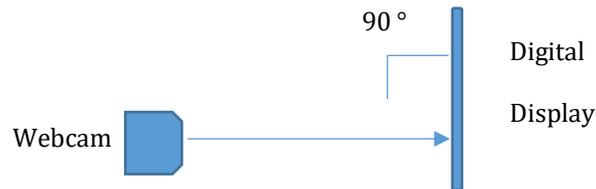
Figure B.8. Ideal Line of Sight of the WebCam

(In an ideal scenario, the line of sight is normal to the plane of the digital display being recorded, and the entire display is visible in the image being captured.)

Minute differences in the display orientation can affect the ability of the SSOCR method used in this experiment in ways that can't simply be understood by looking at the image being analyzed. The human brain does a remarkable job at determining what it's looking at when it comes to character recognition, whereas a software is limited to the scope in which it is programmed. This is why, for example, web pages prevent bots from accessing their sites through the use of CAPTCHA technology, which requests users to type characters they see in an image. These images are typically low resolution, on an angle, and with variable contrasts.

When observing the display of the TSI 4140 Flowmeter, the original image resolution was 320 x 180. This was limited by the setting in the software, as the webcam used had a 1280x720 resolution setting. Because such a low resolution was chosen by mistake, the final cropped images of the digital displays themselves had an image resolution of just 27 x 12. At this point,

even a single pixel difference threw off the software's ability to accurately determine numerical values from the images. It was at this low resolution that the effect of 3D-image orientation was noticed. In Test 2A, the TSI 4140 Flowmeter, which rests on top of the Dusttrak's nozzle (see Figure B.5) was rotated ever so slightly along its vertical axis away from the camera viewing direction. This resulted in bad data for the majority of the images. In contrast, in Test 2B, the TSI 4140 Flowmeter was more precisely perpendicular to the webcam's viewing direction, and the SSOCR software accurately recognized over 99% of the display data. Some examples for the final images from both flow meter tests are shown below.

Test 2B: 2988 2992 3022

Test 2A: 2989 3058 3070

Figure B.9. The SSOCR Output Examples from Tests 2A and 2B

(The SSOCR software had trouble recognizing the digits in Test 2A perfectly when compared to Test 2B. Notice how in the '0' digit in Test 2A appears differently depending on the location in the number string. This is due to a difference in 3D-image orientation of the display in either test.)

The SSOCR software used in this study does not necessarily require the 3D display orientation to be in its ideal state, but in cases where the final image resolution is extremely low, slight variations in the display angle can fool the software, as noise affects the clarity of each digit in the number string slightly differently. The SSOCR software may recognize identical digits at different positions in the number being observed as inherently different. Although this is not simply a resolution problem, a higher initial resolution would have negated this issue completely. This was noticed in the UNI-T display data which, in comparison to the TSI 4140 Flowmeter

data, had a significantly higher resolution. The initial images had a resolution of 352 x 288, and the final cropped and rotated images had a resolution of 56 x 31. At this scale, a single pixel of noise, cannot throw off the SSOCR's ability to accurately decipher the numeric values from the image.

Color Space and Threshold:

The color space is an important factor in defining the boundary that lies between the digits in the image, and the background. Ideally, the numbers that are to be recognized by the SSOCR software should be well-defined and separated as to allow them to be identified individually and accurately. This can be accomplished by converting the image to have only two distinct colors or shades. In this study, the color space was set to grey. This means that black was chosen for the foreground and white for the background. The boundary between these two elements is defined by the threshold, which is adjusted in ImageMagick prior to applying the image manipulation.
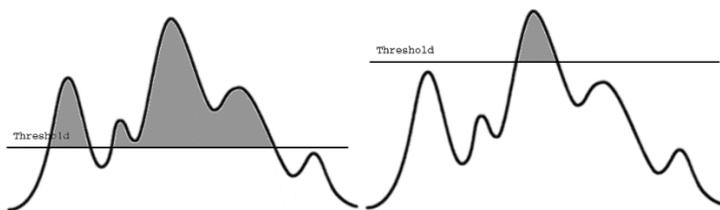
Figure B.10. Image Threshold Visualization
(Source: http://scikit-
image.org/docs/0.13.x/auto_examples/xx_applications/plot_thresholding.html)

(If a color image is converted to greyscale, then the y axis can describe the different shades from white to black for the image. The threshold will define a boundary in all these shades, and the resulting image will be composed solely of pixels either black or white. This will define the boundaries between the digits being observed and their backgrounds. In the extreme cases, when the threshold is 100%, the image will be completely white, and when it is 0% the image will be completely black.)

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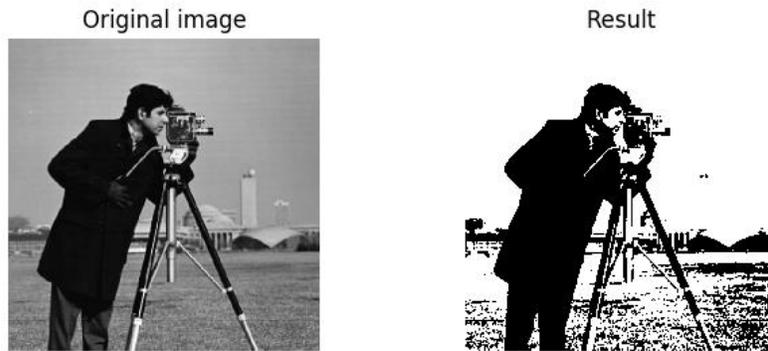
Figure B.11. Example of a Greyscale Image Converted to One with a Fixed Threshold .
(Source: https://svi.nl/SeedAndThreshold)

In order to minimize the effects of noise and to clearly define the boundaries of each digit in the number string, the threshold was manually adjusted. Sometimes, the SSOCR software will witness two merged digits when the boundary isn't well defined, and will not recognize the character as a seven-segment digit. The decimal point in the flow data was especially cumbersome in that it sometimes disallowed the first two digits from being recognized distinctly.

In the end, a threshold of 80% was used for the UNI-T data, 23% for test 2A, and 19% for test 2B.

Image Noise:

Noise will occur when image resolution is low, when a threshold is improperly set, when 2D image orientation transformations degrade the image quality, and when changes occur in environmental effects over the duration of the test being conducted. Examples are shown in Figure B.12.

(a) Image Captured Early on in the
Burn Cycle

(b) Image Captured Near the end
of the Burn Cycle

Figure B.12. Example Showing Differences Between
Two Captured Images with Identical Number Values

The images in Figure B.12 are solely cropped versions of the original captured images and no further changes have yet been made with ImageMagick. Image (a) in Figure B.12 was captured early on in the burn cycle, while image (b) was captured near its end. Although these images represent the same digital number values, the shades of blue are not identical. This is most likely due to either a change in lighting or the effects of smoke perturbing the webcam's sight. Furthermore, this shows how a fixed threshold in ImageMagick may pose some recognition issues in SSOCR somewhere within the entire period of a given test. It is, therefore, important that environmental parameters, such as lighting, be held roughly fixed throughout the testing period.

As threshold changes from its ideal value, boundaries that define where the number is and where the background lies will be poorly defined. Noise will occur between the numbers, breaking their boundaries. As was described earlier, when the image resolution is low, even a single pixel in the wrong place can effectively disallow the SSOCR software from identifying a number. Environmental effects such as smoke being present in the air can also lead to noise. In this study, a single threshold value was chosen for each set of images, but it is clear that once the room

becomes smoky, the numbers become more illegible at a fixed threshold. This issue can be alleviated by the use of multiple thresholds for each set of data (Tests 1, 2A, or 2B). Changing the threshold, however, was not effective in helping SSOCR determine numeric values in Test 2A, as the 3D-image orientation remained consistently different than that of Test 2B. In addition, attempting to rotate the image ever so slightly led to further degradation of the image due to lossful compression. The resulting noise became too overwhelming to distinguish from the digits themselves, and adjusting threshold had little effect.

Attempting lossless compression algorithms on the images could help in this scenario. However, there is a much easier fix to maximize **the final image resolution.** This will ensure that any image rotations due to lossful compression will have only negligible effects on the SSOCR software's ability to discern numerical characters. This can be accomplished by ensuring the **initial image resolution** is set to the highest setting allowed by the webcam in the software. Noise, 3D-image orientation (to an extent), environmental effects, and threshold limitations can all be negated with a sufficiently **high final image resolution**.

In order to properly utilize both the SSOCR and ImageMagick software suites in a Windows 10 environment, CygWin was used to unpack and compile each software, as well as to utilize their tools, which would normally require a Linux operating system.

Detailed Method:

Once both ImageMagick and SSOCR are installed through CygWin, the following commands were used:

- Using Ctrl-Right Click, open command prompt in the folder with the images labelled "snapshots".

- Prior to the use of either ImageMagick or SSOCR, the time stamps for each image captured by SkyStudioPro must be recorded. Because SkyStudioPro labels each image with a time stamp in the title itself, this can be extracted and then tabulated. This was done with the help of the regular windows command prompt.

- In the Windows 10 command prompt, type: dir > dir.xls

- Open "CygWin" as administrator. It is a type of command prompt which gives access to a Linux environment in Windows 10

- Type: cd /cygdrive/c/Users/polyzoi/Downloads/SSOCR/ImageMagick-6.8.8/bin/snapshots

- This will point the CygWin terminal to the folder where the snapshots being analyzed lie.

- mogrify -format png -crop 56x31+122+117 -colorspace gray -auto-level -threshold 80% *bmp

  This replaces each file in the folder "snapshots" with an altered version. The new images are: cropped to show images of only the digits to be determined by SSOCR, have their color space converted to grey, have their thresholds changed (in this case to 80%), and the file format converted into bmp (which will be compatible with the SSOCR software), as shown in Figure B.13.

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Figure B.13. Example of Resulting Image from Test 1A

Once the files are transformed, they are then moved into a new folder named "converted snapshots".

- Then, SSOCR is used, pointing to this new folder, to determine digital values for the characters in each image file.

```
$ FILES=./converted_snapshots/*.png ; for f in $FILES; do
/cygdrive/c/Users/polyzoi/Downloads/SSOCR/ssocr-2.16.4/ssocr -f
white -b black -d 3 -T "$f"; done
>> text.xls
```

With the use of a simple 'for loop,' each image in the 'converted snapshots' folder is individually analyzed by the SSOCR software, with the background (-b) set to black, foreground (-f) set to white, and the number of digits (-d) set to three.

Each result will be outputted into an excel file titled 'text.xls' in the same directory.

During this process, it is important to remember that:

- Image Crop dimensions and position are manually determined, and were assumed to be fixed across all images for each Test: 1A, 2A, and 2B.

- 3D-orientation, although assumed to be fixed as well, in the end, it was determined to have slight variations between Tests 2A and 2B. This was due to the fact that the Flowmeter, which is free to rotate along its axis, was repositioned between tests. Assuming nothing is touched during the duration of a test, 3D-orientation will remain fixed for the duration of the test. If tests are to be conducted outdoors, for instance, wind could have an effect on this.

- In this project, threshold values are assumed to be fixed throughout each test. Environmental changes, although evident during the duration of Test 1A, were not significant until late in the test. If smoke is expected to billow heavily throughout the entire duration of a test, then image threshold must not be fixed throughout the test for the best results for all data. Identifying threshold values across a test in this hypothetical scenario will add one more layer of difficulty.

- The validity of the character recognition method used in this experiment was confirmed by manually checking values, but only once the SSOCR software indicates it is detecting the right number of numerical values. If any evidence during the calculation process suggests a hindrance in adequate completion, such as a failure to fully complete or the wrong number of digits in each image detected, then manual adjustments in the ImageMagick or SSCOR code were made to better allow the software to detect the right values. Once the right number of digits is detected and the images run through the SSOCR software smoothly, the resulting values in the xls file generated are confirmed through random selection of 20 images from the beginning, middle, and end of the test, and then manually checking the values by visually comparing them to the original images.

Strengths of the SSOCR Method for Data Logging and its Potential Use in Future Research*:*

The SSOCR techniques used in this study can be implemented in a wide range of research projects. In this study, an open-source method was used capturing data that are only displayed on screen and converting these to digital values which were subsequently tabulated and analyzed.

Using this method, less expensive instruments that detect data that have a strong correlation with data collected by a much more expensive instrument with built-in data logging capability can be used in Linux. This would be ideal in any study that requires a large number of data logging instruments that not only may be too expensive but also unavailable for immediate use. The following is a rough comparison of the cost estimate of the equipment required using the SSOCR method and Linux (as opposed to Windows) and the Dusttrak II 8530 method:

Table B.3. Cost Comparison Between Two Methods of Data Collection

Cost of SSOCR Method (CAD $)		Cost of Dusttrak II 8530 Method (CAD$)	
Frame/Light:	30	Windows PC:	400
Rasp Pi :	40	OS:	100
OS:	0	TrakPro Software:	0
webcam/software:	30	Dusttrak II 8530:	10000
ImageMagick:	0	**Total:**	**10500**
SSOCR software:	0		
+ UNI-T:	200		
Total:	**300**		

The potential for the SSOCR method is quite extensive when it comes to studies involving data collection from a large number of homes concurrently. The benefit of measuring data concurrently is in reducing the error generated from differences in environmental conditions over time. If the ultimate goal is to compare data from different homes, the environmental differences

that arise over time must be eliminated. If data are taken in April from one home and in July from another, no useful information can come from a comparison of the two data sets, as constant environmental conditions between the two would be implied. This effect could in fact lead to a larger systematic error in data than that introduced by the use of cheaper instruments such as the UNI-T or Dylos DC1700, but this will be further analyzed in the Instrument Comparison section later in this study.

Weakness of the SOCCR Method:

Initially, upon deciding to use software for character recognition in this study, the extent of the challenges was not accurately predicted. First, determining digits with the use of traditional character recognition software, known as OCRs or Optical Character Recognition software, failed to even recognize any digits, despite the use of many image manipulation techniques. Understanding the differences between how SSOCR and how a traditional OCR determine numerical values is key in dealing with the challenges.

Typical traditional OCRs generally assume that a printed digital document is being scanned and saved as a PDF. From this PDF, a digital copy can be estimated when run through an OCR software, by determining each character and placing them in the right order. This would be beneficial if one would want to edit scanned documents and be able to reprint them. Acrobat Reader is an example of a software that can implement OCR techniques in this fashion.

In contrast, the SSOCR method used in this study will only be beneficial when the software system is already told specifically that it will be used to recognize only seven segment characters. Because the software now is expecting seven segment characters, it can study each segment, and understand its positioning with relation to other segments, and ultimately determine numerical values. Because the software is told what the total number of digits is in each picture beforehand, if a numeric character is recognized, it will move over to the next set of segments of roughly equal spacing, as font sizing is assumed to be constant across the number string. If another numeric character is recognized, it will again move over and so on. When segments within the same digit are partially or fully merged with each other, the software does not falter. As long as there are no pixels conjoining the individual digits, each character can be separately recognized. When conjoining occurs, the software will fail to recognize accurately what is being displayed in the image.

When a number with fewer characters than expected is listed in the output, one can assume the software did not accurately recognize the complete number. Reasons for this include, but are not limited to: partial to complete merging of segments between different digits in the number due to noise caused by low image resolution, cropping the image improperly (even a single pixel can make a significant difference), or incorrect threshold chosen in ImageMagick,. Other reasons include: low image resolution or image quality degradation due to rotation transformation, 3D-orientation of instruments that are improperly positioned, or the number of digits recognized visually does not actually match what is typed in the SSOCR code (-d).

The SSOCR software must be pre-tuned to know how many digits are in the number it is identifying. This could be a limiting factor when instruments with varying digit sizes are used without numerical placeholders.

Another potential pitfall of this method could be an inaccuracy issue that cannot easily be perceived. If thousands of images are being processed through ImageMagick and SSOCR, and the software detects the correct number of digits for each image, the accuracy of each digit in each image is not completely guaranteed. In this study, random images were always selected after they were run through the software without any hiccups to develop a confidence in its ability to accurately depict them. If there were any signs of mischaracterization of the digits, despite the software not halting, it was often, if not always, detected early by visual inspection. Of course, this is extremely limiting if the environmental conditions randomly change throughout the testing period without knowing exactly when, such as with the use of a woodstove in a real-world environment. In this case, the SSOCR software may not be able to accurately recognize the numbers. To counter this, it is important to select the images and digital values to be visually inspected and checked at random so that the number of images selected is an adequate representation of the complete set of images.

Ultimately, if the final image resolution of the digital display being observed and captured is too low, there is no image manipulation or SSOCR tuning that will compensate, and the software will either fail or the resulting recognized digits will not accurately represent what is actually being displayed on the instrument.

In dimly lit rooms, there will be an issue if the digital display being recorded does not have a back light. In these situations, ImageMagick will fail to precisely depict the foreground/background boundary when any threshold is chosen. Under these lighting circumstances the SSOCR software will not identify any digits in the images. 3D-orientation of the digital display could also be altered, or the display even covered accidentally in a real-world environment, and this would in turn prevent any SSOCR techniques from being effective. To counter these issues, an open mesh, partially lit but rigid SSOCR cage setup can be developed, where the webcam is fixed pointing normal to the display and nothing can block or hinder the camera's view of the display.

Instrument Comparisons and Detailed Analysis of the Results:

The purpose of this section of the study is to compare $PM_{2.5}$ data collected by various instruments: Dusttrak II 8530, Dylos DC1700, and UNI-T, and to determine any possible calibration curves which can then be used to create Dusttrak II equivalent data. Data were collected simultaneously during a 3-hr woodstove burning session in the test home and the instruments had their internal data logging intervals matched to 1-min time lengths. The smoke in the room moved quite slowly; however, it would take very few seconds for it to travel between any of the instruments. By using a 1-min data logging interval, less error will be introduced due to these spatial differences (see Tables A.3 and A.4). The instruments were placed on a table as close to each other as possible, 7-10 feet in front of the woodstove. During this test, data were collected from three Dusttrak II 8530 instruments, one Dylos DC1700, and one UNI-T. It should

be noted that the Dylos DC1700 does not measure $PM_{2.5}$ as a mass concentration, like the Dusttrak II and the UNI-T (mg/m^3). Instead, it detects a particle count (number of particles / ft^3).

Once $PM_{2.5}$ data from the UNI-T Test 2A were digitally tabulated with the use of the SSOCR method described earlier, it was characterized by a 5-s equivalent data logging interval, as that is the rate in which images were captured. Although the UNI-T's display refreshes every second, the pictures captured at 5s intervals act as snapshots from which min-interval data can be estimated. The 5s interval data were then converted to equivalent data with a 1-min interval through averaging, in order to directly compare them to the Dusttrak II 8530 which was also set to measure particulate at a 1-min interval. If pictures were captured at 1-s intervals, the minute equivalent intervals would have been slightly more accurate.

The reason a 1-min interval was chosen in this case, was to also allow for comparison data with the Dylos DC1700, which has a minimum built-in logging interval of 1 min.

Before any comparison analysis is made between the instruments, a good benchmark for a perfect linear correlation must be established. Earlier it was decided that a 1-min interval would be sufficient to eliminate any spatial effects between the instruments. By measuring the correlation between identical instruments at different, yet closely positioned, locations, the magnitude of this potential spatial effect can be understood. It is hypothesized that an ideal value for R^2 when comparing different instruments would be within range of what would be found when comparing identical instruments.

In the following discussion, data from three separate gravimetrically calibrated Dusttrak II instruments will be averaged, and the average will represent what will be discussed as the Dusttrak II data. Because three Dusttrak II instruments were used, there can be three separate comparisons between them. Their R^2 values are summarized in the following table.

Table B.4. R^2 Values Determined by Comparisons of Datasets
from Closely Positioned Pairs of Dusttrak II Instruments*

Comparison	Dusttrak Number	Dusttrak Number	Correlation Coefficient (R^2)
A	4	2	0.9983
B	4	6	0.9929
C	6	2	0.9962

*Comparing closely positioned Dusttrak II Instruments which were previously gravimetrically calibrated. R^2 is virtually equivalent to a value of 1 when identical instruments are being compared. As seen, spatial effects are minimal as a 1-min interval was chosen.

Comparison Between Dusttrak II and the UNI-T:

The UNI-T, which has an internally fixed data logging interval of one second, had its display captured at 5 second intervals. These images were digitally recognized with the use of the seven-segment character recognition method described earlier. Once the data were digitized and tabulated, they were then averaged in Excel and converted into a data set with an equivalent 1-min averaged logging interval. Despite only capturing one fifth of all possible data points, as previously explained, this snapshot of information is sufficient to describe a 1-min equivalent averaged data logging interval.

With identical data logging intervals of 1 min, the two data sets: the average $PM_{2.5}$ concentration detected by closely positioned Dusttrak II instruments and that obtained from the UNI-T, were compared. Initial ambient values of approximately 0 mg/m³ (during the first 17 min) are ignored to allow for a more accurate comparison of the instruments' ability to detect $PM_{2.5}$ concentrations typically found in a home with a woodstove in use.
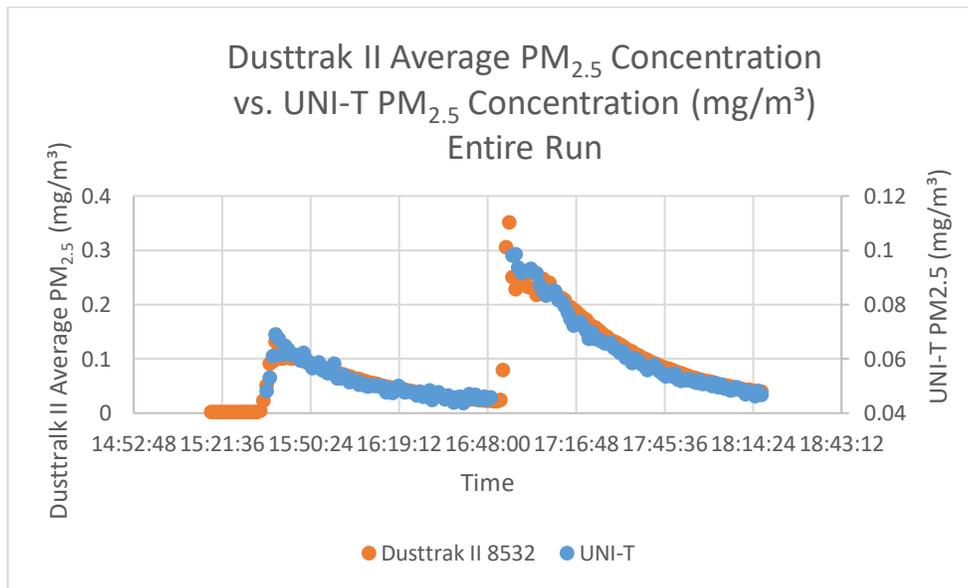

Figure B.14. Comparison Between Dusttrak II $PM_{2.5}$ Concentration Data and UNI-T Concentration Data, Over an Identical Period of Time
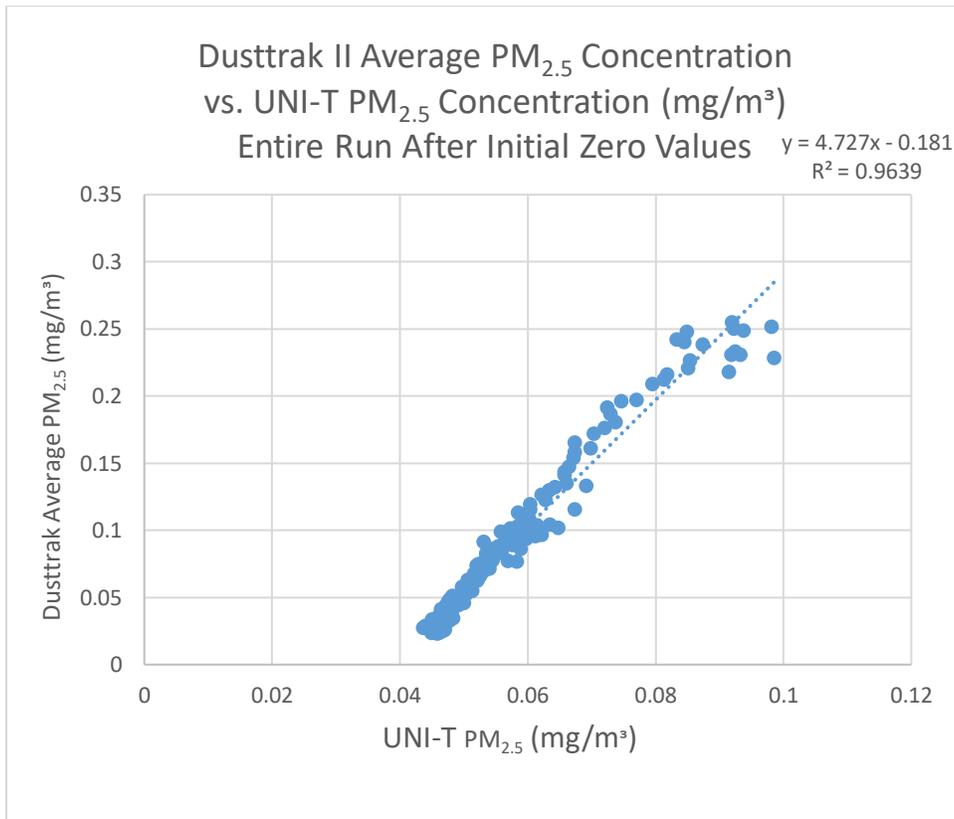
Figure B.15. Correlation Plots Comparing Dusttrak II and UNI-T for an Identical Testing Period

As shown in Figure B.15, there is a high correlation ($R^2 = 0.9639$) between the two data sets, despite three separate occasions in which the woodstove door was opened and large spikes in particulate observed. Initially, little was expected using a UNI-T device, especially with a price as low as it was. Digital values being displayed were not corrected or calibrated for wood smoke and the quality of the instrument itself was not up to a standard expected for a worksite. Interestingly, the results generally approximate those found with a Dusstrak II when a calibration factor is applied. This comparison calibration factor (CCF) is defined by the x-y relationship and the intercept. The accuracy of the determined CCF will be maximized when $PM_{2.5}$ concentrations

are within the acceptable range defined by the test ($0.035 - 0.3$ mg/m^3 as measured by the Dusttrak II, or $0.04 - 0.1$ mg/m^3 as measured by the UNI-T).

It is important that an updated CCF is determined in future studies that may require detection of higher PM$_{2.5}$ concentrations, to allow for a more accurate representation of Dusttrak II equivalent data by the less expensive and more portable UNI-T. This range was chosen to represent a typical range expected in a home with woodstove in use.

Dylos DC1700 vs. Dusttrak II Detailed Comparison:

In contrast to the strong correlation found between data collected by the Dusttrak II and the UNI-T, the comparison with the more expensive Dylos DC1700 displayed some significant inconsistencies. It should be noted that the Dylos data describe both a large and a small particle count per volume, as opposed to mass concentrations measured by the Dusttrak II or the UNI-T. The Dylos DC1700 detects both small and large particle counts which, according to the manufacturer, represent PM$_{0.5}$ and PM$_{2.5}$.

In a study conducted by the World Air Quality Index Team (WAQIT), data collected by both Dylos DC1700 and Dusttrak II instruments, simultaneously, at the Beijing Chaoyang Agricultural Exhibition Hall, Beta Attenuation Monitor (BAM) stations and the Beijing Sanlitun Dylos monitoring station, were compared (AQIT, 2014). The study, which is called: Air Quality Experiments on Alternative Sensors and Prediction Model, describes the large particle counts

detected by the Dylos DC1700 to most accurately be described as PM_{10} as opposed to $PM_{2.5}$ as directed by the manufacturer. Additionally, the study found that the small particle counts detected and displayed by the Dylos DC1700 can be more accurately described as $PM_{2.5}$ as opposed to $PM_{0.5}$. The data compared were taken over a 12-hr period.

The following compare data sets 1 to 4 from the study described above through a multidimensional correlation graph. Data set 1: Dylos Small Particles >0.5, Data set 2: BAM $PM_{2.5}$, Data set 3: Dylos Large Particles >2.5, and Data set 4: BAM PM_{10}.

Figure B.16 clearly shows that in the case of the Beijing study, a good correlation between Data Sets 1 and 2, as well as between Data Sets 3 and 4 is evident. Despite Dylos describing the large particles detected by the DC1700 (Data Set 3) as >2.5, only a very weak correlation can be seen when comparing this data set with $PM_{2.5}$ detected by the BAM setup (Data Set 2).
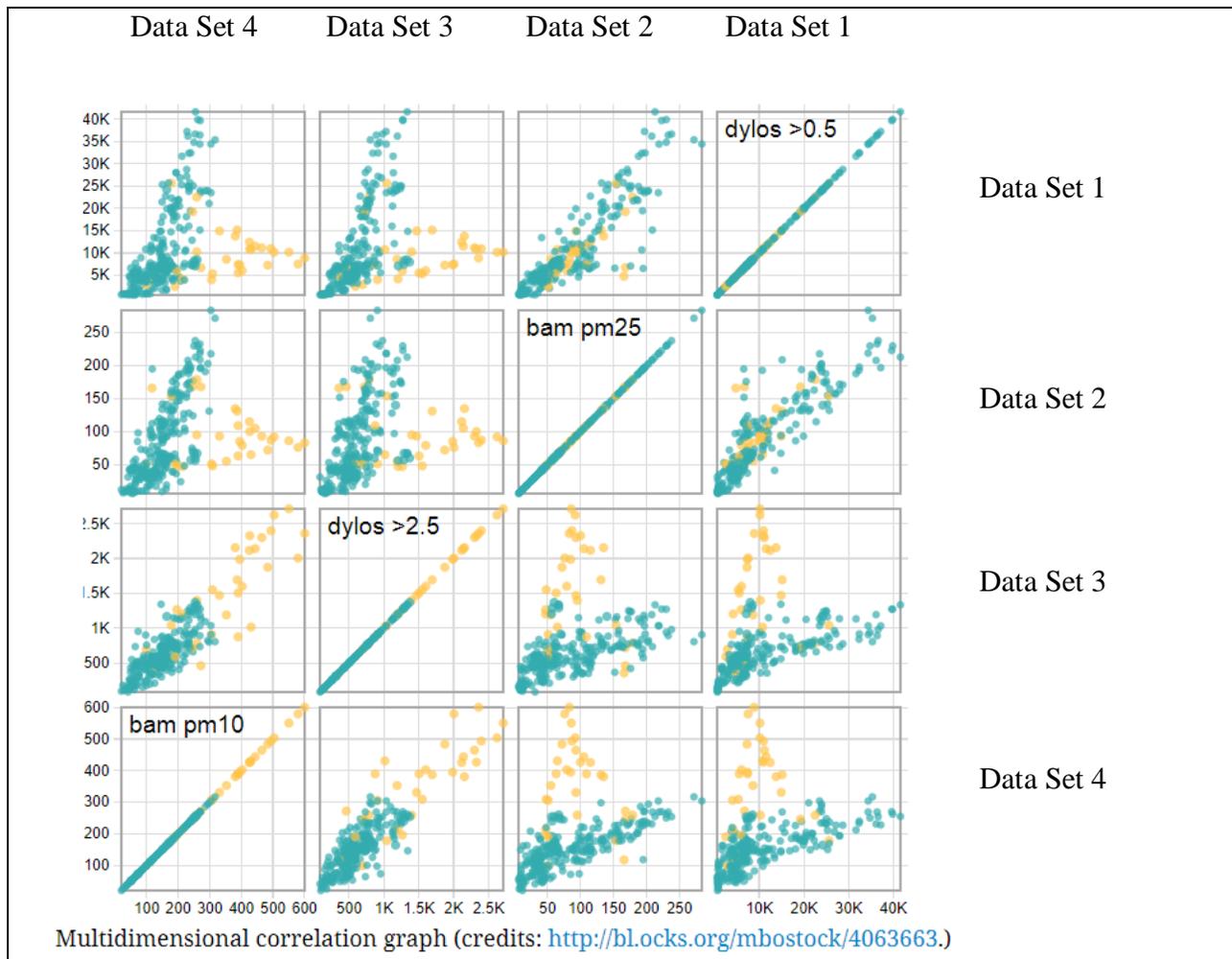
Figure B.16. Multidimensional Correlation Graph from the Air Quality Index Study

(Each Data Set 1, 2, 3 and 4 is labeled in the diagonal of the 4x4 diagram. Along this diagonal, it can be seen that identical datasets are perfectly correlated (AQIT, 2014)).

Because the two data sets chosen by the WAQIT for their comparison were cropped from larger data sets which, in their entirety, are characterized by several spikes and drops in particulate levels, as shown below in yellow (see Figure B.17), the data collected in our indoor study will be similarly cropped as to describe several individual rises or drops in particulate levels. Each data set given in Figure B.18 can be broken into four parts: two rises, and two drops. Between the large and small particle tests, that makes a total of eight separate data sets.
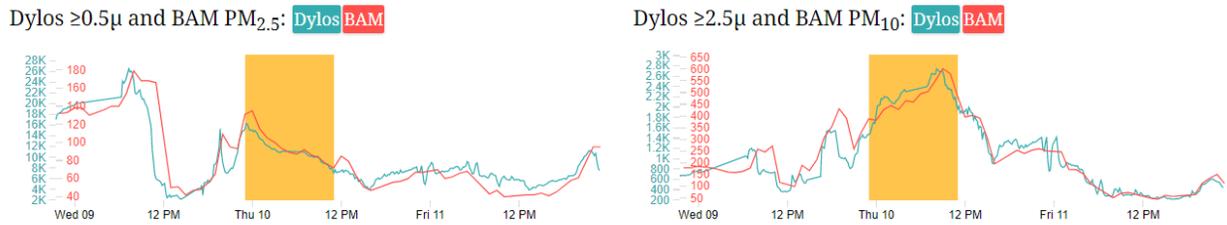
Figure B.17. Data Used in the WAQIT Study to Compare PM$_{2.5}$ Data
Obtained by Dylos DC1700 and BAM (AQIT, 2014)
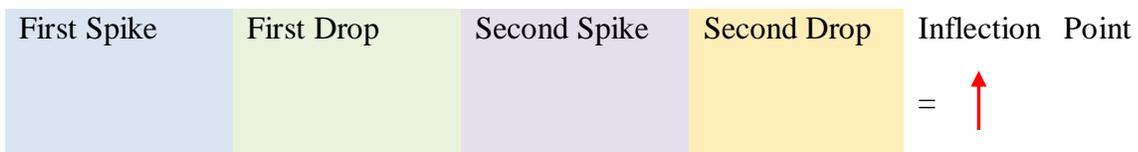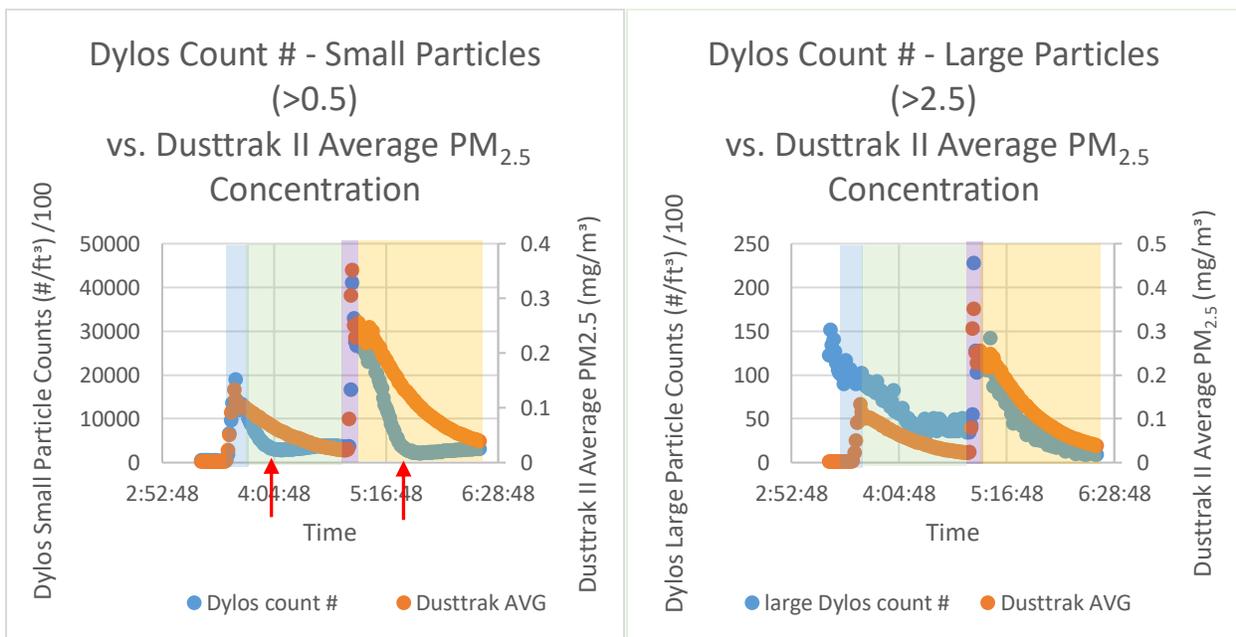
Figure B.18. Data Taken from the First Nations Test Home While the Woodstove was in Use
for Instrument Comparison Purposes in the Current Study

The WAQIT study measured ambient particulate levels and focused on a data collection period during which particulate levels only rose or dropped at a slow and steady pace. In contrast, in our study, which was conducted indoors in a First Nations home with a woodstove in use, particulate levels are characterized by abrupt spikes and subsequent steady drops. These partitions are colour-coded in Figure B.18. The first spike in particulate levels is due to the initial biomass combustion process while the woodstove door is open and the first drop off is after the woodstove door is initially closed and the wood begins to burn sufficiently. The second spike is caused by the release of particulate into the air after opening the woodstove door for refueling. The second drop occurs after the woodstove door is closed for a second time. An inflection point can be seen during both drop-off periods in Figure B.18 where the small particle count measured by the Dylos DC1700 rises slightly after dropping. This corroborates what was observed in the WAQIT study, where small particle and large particle counts measured by the Dylos DC1700 rose and fell countering each other at points during the same period of testing.

In our study, PM_{10} was not monitored by the Dusttrak II instruments, and, therefore, large and small particle counts measured by the Dylos DC1700 were compared only with $PM_{2.5}$ concentrations, which were measured and averaged across several closely positioned Dusttrak II instruments. In the case of the WAQIT study, it was found that the small particles detected by the Dylos instrument were well correlated with $PM_{2.5}$ data detected by the BAM setup. It remains to be seen if $PM_{2.5}$ levels in the test home from our study detected by the Dusttrak II instruments closely correlate with large or small particle counts detected by the Dylos DC1700.

In the WAQIT study, twelve hours of data were specifically selected from a larger data set obtained during a four-day period, and was characterized by only a single rise or a single drop in particulate levels. Although no comparison was made with all the rises and drops included, when such a comparison is made using data collected in our study, no simple linear correlation was found. This can be seen in Figures B.19 and B.20 below where data from the small as well as from large particle counts measured by the Dylos DC1700 are compared to $PM_{2.5}$ concentrations as measured by the Dusttrak IIs.
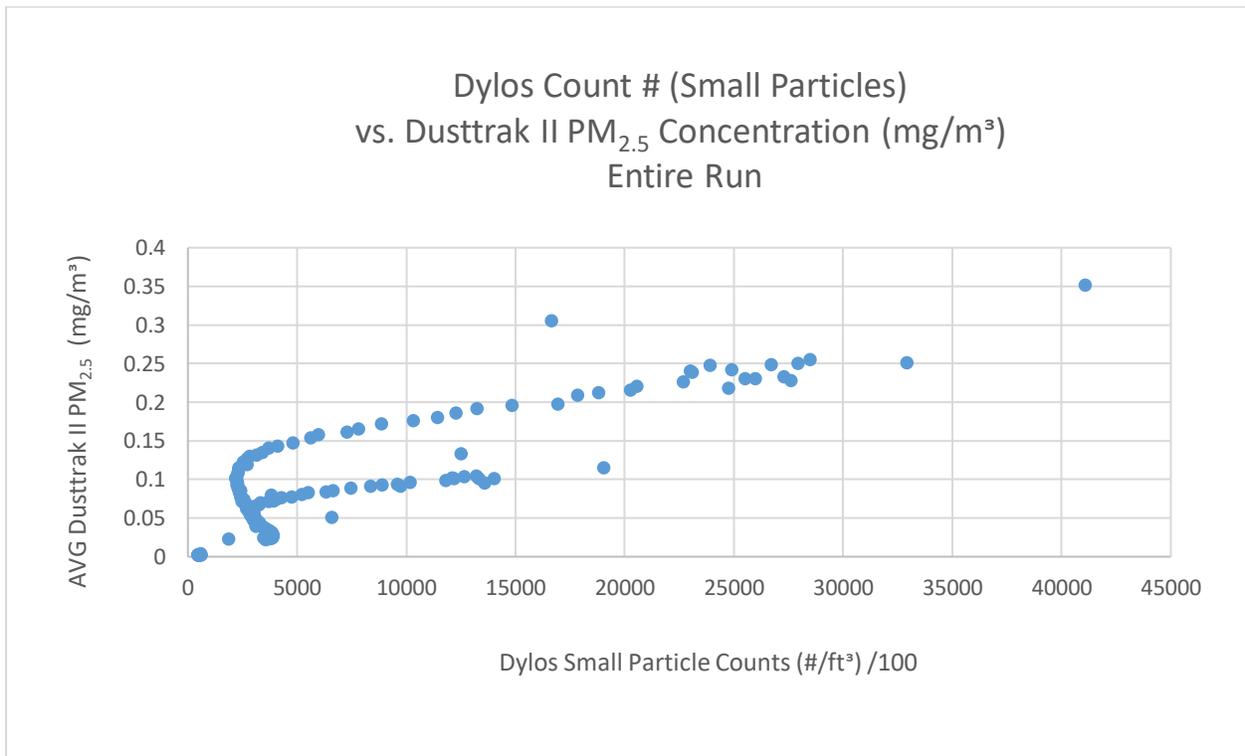

Figure B.19. Correlation Analysis Comparing the
Dylos DC1700 Small Particle Counts and Dusttrak II Concentrations
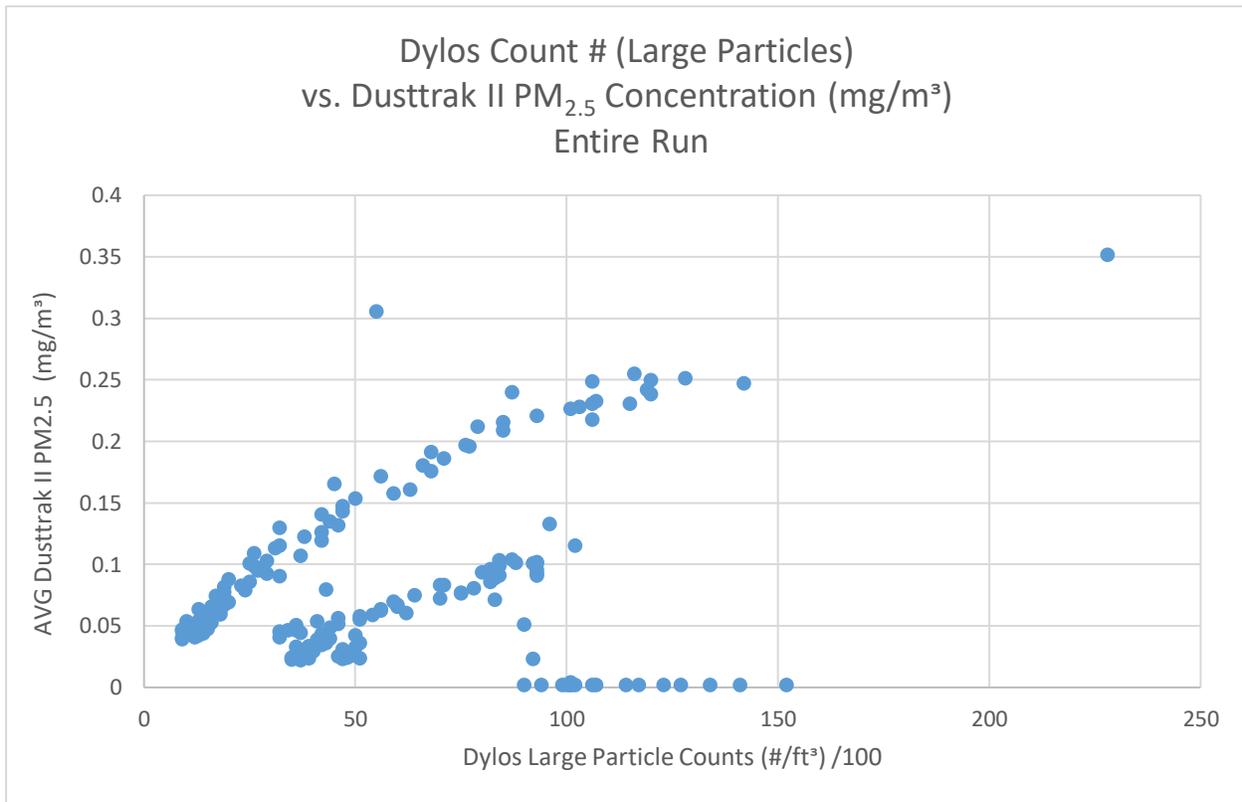
Figure B.20. Correlation Analysis Comparing
the Dylos Large Particle Counts and Dusttrk II Concentrations

During the entire period of data collection, from start to finish, a non-linear power law correlation best describes the relationship between the results from the two types of instruments. There appears to be two well defined curves in each of Figures B19 and B.20 and that matches the number of spikes in PM$_{2.5}$ seen in each plot of Figure B.18. As shown in both figures, the ambient pre-burn concentrations are near zero at the start and are not representative of wood smoke particulate. These can be ignored when making instrument comparisons of wood smoke particulate.

Examining Figures B.19 and B.20, it can be seen that there are many possible predicted $PM_{2.5}$ concentration levels that can be determined from a single Dylos DC1700 count, or vice versa. According to Dacunto et al. (2015), the Dylos DC1700 detects particulate differently than the Dusstrak II and will only be useful in accurately determining $PM_{2.5}$ concentration levels when particulate is freshly introduced to the device or when the particulate concentration is quickly decaying. This can be seen in the WAQIT's own website, where continuous BAM and Dylos $PM_{2.5}$ data are being monitored and uploaded near Beijing Chaoyang Agricultural Exhibition Hall.

Although the Beijing study deals mostly with measuring air pollutants, predominantly generated from factories, the difference between the Dylos DC1700 and a gravimetrically calibrated $PM_{2.5}$ BAM system is obvious.
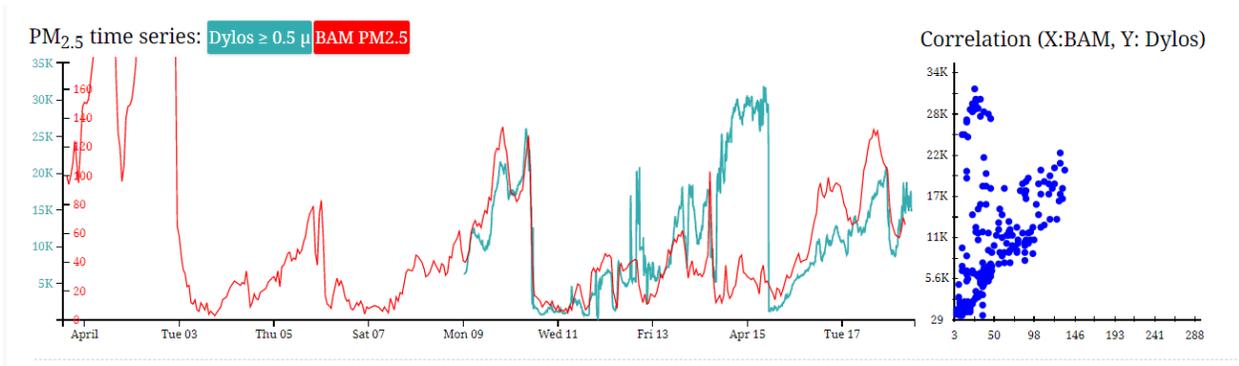

Figure B.21. Data Collected by the WAQIT Over a Span of Three Days Comparing Dylos Small Particle Counts to BAM $PM_{2.5}$ Concentrations

(As can be seen, there is no clear linear correlation between the results during the period of the data collection. Although at first, when $PM_{2.5}$ is freshly introduced to the Dylos DC1700, the data sets seem, visually, well correlated, there is a point in which there is a divergence between the Dylos counts and the BAM $PM_{2.5}$.)

In light of this observation, and the information from the WAQIT, the results in our study were analyzed using segmented data where the rise or drop of particulate levels were observed separately. In all, a total of eight comparisons were made, as shown in Table B.5 and plotted in Figures B.22 (a) to (f).

Table B.5. Time-Segmented Partition Labels for Dylos DC1700 vs. Dusttrak II Data

$PM_{2.5}$ Count per Cubic ft (Dylos DC1700) vs. $PM_{2.5}$ Concentration (Avg. of Four Dusttrak II) (Eight Comparisons using Data from 1-min Logging Interval)			
(a) # Small Particles (>0.5µm) - DC1700		(b) # Large Particles (>2.5 µm) - DC1700	
Case	Position	Case	Position
i	First Spike (Figure B.22 (a))	i	First Spike (Figure B.22 (e))
ii	First Drop (Figure B.22 (b))	ii	First Drop (Figure B.22 (f))
iii	Second Spike (Figure B.22 (c))	iii	Second Spike (Figure B.22 (g))
iv	Second Drop (Figure B.22 (d))	iv	Second Drop (Figure B.22 (h))

Data acquisition began prior to any biomass burning and continued once the woodstove was fired up. In every case, the first spike in wood smoke concentration occurred at the precise point at which levels rose significantly in both instruments being tested. The initial ambient $PM_{2.5}$ values, prior to each first spike, are excluded from the analysis. This is partially due to the fact that indoor $PM_{2.5}$ levels prior to any biomass burning are not accurately representing wood smoke particulate concentrations, and using this ambient data to compare how each instrument measures smoke particulate levels differently will not be useful. As the woodstove is initially

fired up, and once smoke begins to permeate the room, a clear bump in PM$_{2.5}$ concentration will then be detected by each instrument at a specific point in time. This initial bump from ambient values is clear and can solely be attributed to wood smoke particulate. This is where each first spike is assumed to begin. It should be noted that, in the case of the first spike shown in Figure B.22(e), comparing the DC1700 large particle count with that measured by the Dusttrak II, no such clear bump in PM$_{2.5}$ is visible in the DC1700 large particle count. Instead, the starting point for the first spike is defined solely by the Dusttrak II.

(a) Dylos DC1700 and Dusttrak II Comparison Analysis:

In this section, correlations between data collected by the Dylos DC1700 and that collected by the Dusttrak IIs are analyzed for each segment described in Table B.5.

Case i): Results from Dylos DC1700 small particle counts vs. average results from the Dusttrak PM$_{2.5}$ concentrations (mg/m³) during the first spike are shown in Figure B.22 (a). A linear correlation is evident in this Figure with R² = 0.928.
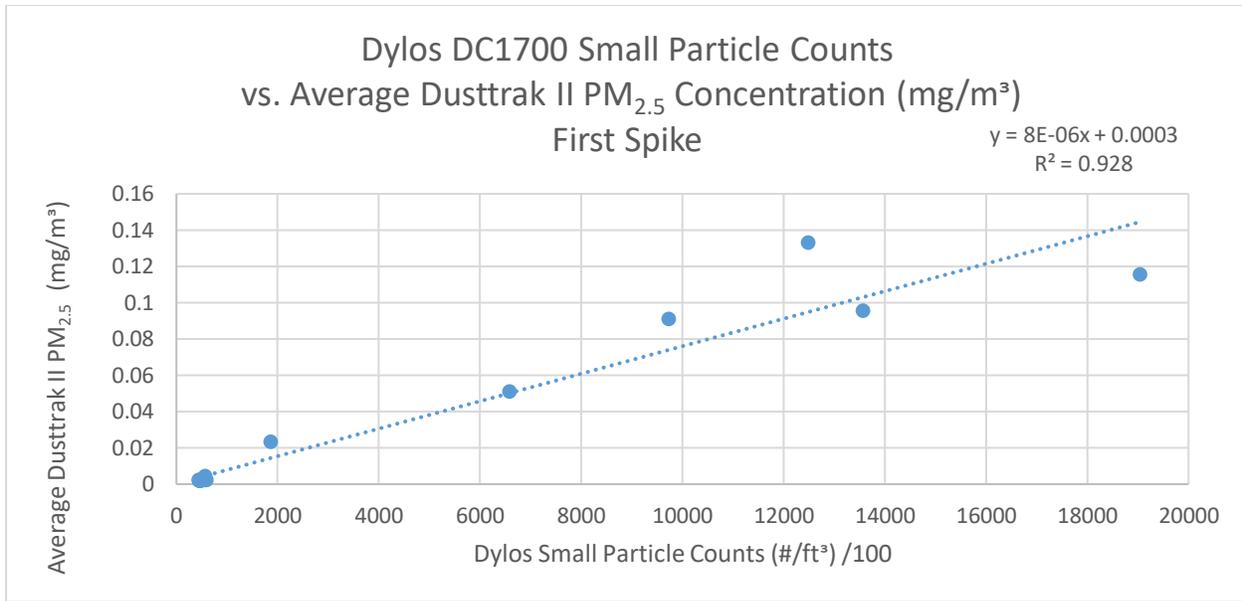
Figure B.22(a) Dylos DC1700 Small Particle Counts
vs. Average Dusttrak PM$_{2.5}$ Concentration (mg/m^3) for the First Spike

Case ii): Results from Dylos DC1700 small particle counts vs. average results from the

Dusttrak PM$_{2.5}$ concentrations (mg/m^3) during the first drop are shown in Figure

B.22(b). Overall, a non-linear correlation can be observed in this Figure. This indicates that the

Dusttrak II equivalent data cannot be synthesized from Dylos DC1700 small particle data once

the rate at which the drop in PM$_{2.5}$ concentration levels off some time after the woodstove door

is closed. During this drop-off stage, little additional particulate is being introduced as the

woodstove door is closed. This behaviour is further demonstrated by the presence of the

inflection points in Figure B.18, shown by the red arrows, after which small particle counts

detected by the DC1700 rise again, despite the fact that PM$_{2.5}$ concentrations, as reported by

the Dusttrak II, follow a constant decay.
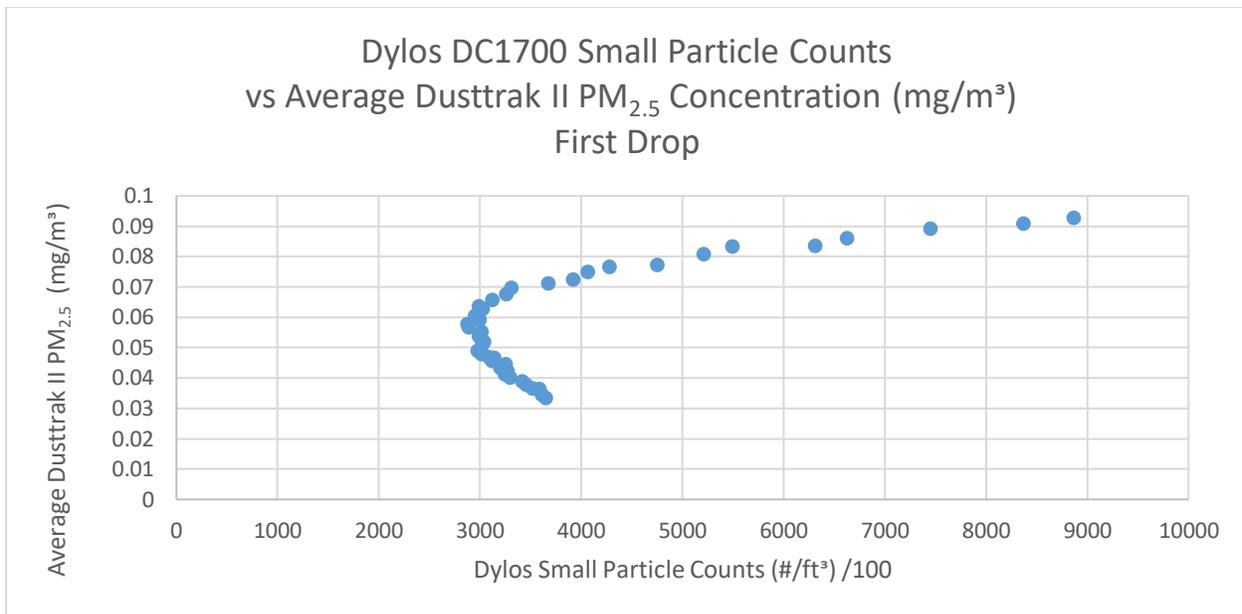
Figure B.22(b) Dylos DC1700 Small Particle Counts vs. Average Dusttrak II PM$_{2.5}$
Concentration (mg/m^3) for the First Drop

Case iii): Results from Dylos DC1700 small particle counts vs. average results from the Dusttrak PM$_{2.5}$ concentrations (mg/m^3) during the second spike are shown in Figure B.22 (c). During the second spike in PM$_{2.5}$ resulting from the woodstove door being reopened for refuelling, a linear correlation is observed when comparing the data from the two instruments. In this case, however, the correlation coefficient drops significantly to R^2 = 0.5877. The outliers in the correlation plot are well within the boundaries of what was defined to be the period of the spike in question.
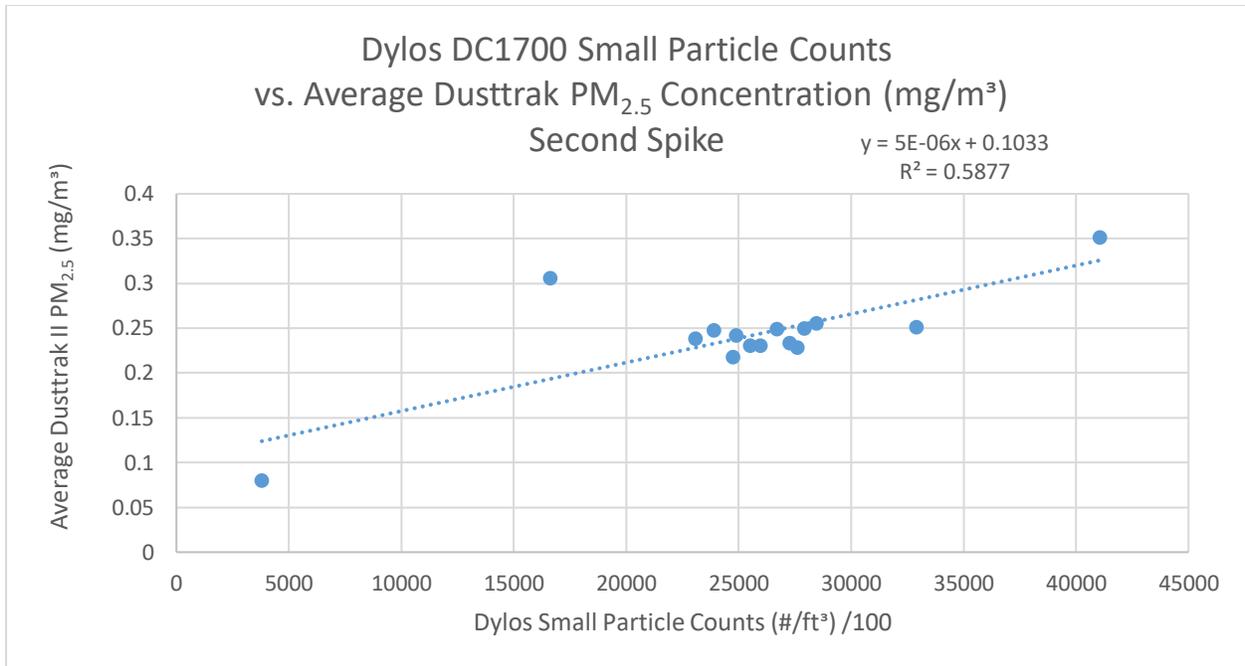
Figure B.22(c) Dylos DC1700 Small Particle Counts vs. Average Dusttrak II PM$_{2.5}$ Concentration (mg/m³) for the Second Spike

Case iv): Results from Dylos DC1700 small particle counts vs. average results from the Dusttrak PM$_{2.5}$ concentrations (mg/m³) during the second drop are shown in

Figure B.22 (d). During the second drop, once again, the linear relationship disappears. It can be seen, however, that as concentration levels decay abruptly from their maximum, a linear correlation is evident.
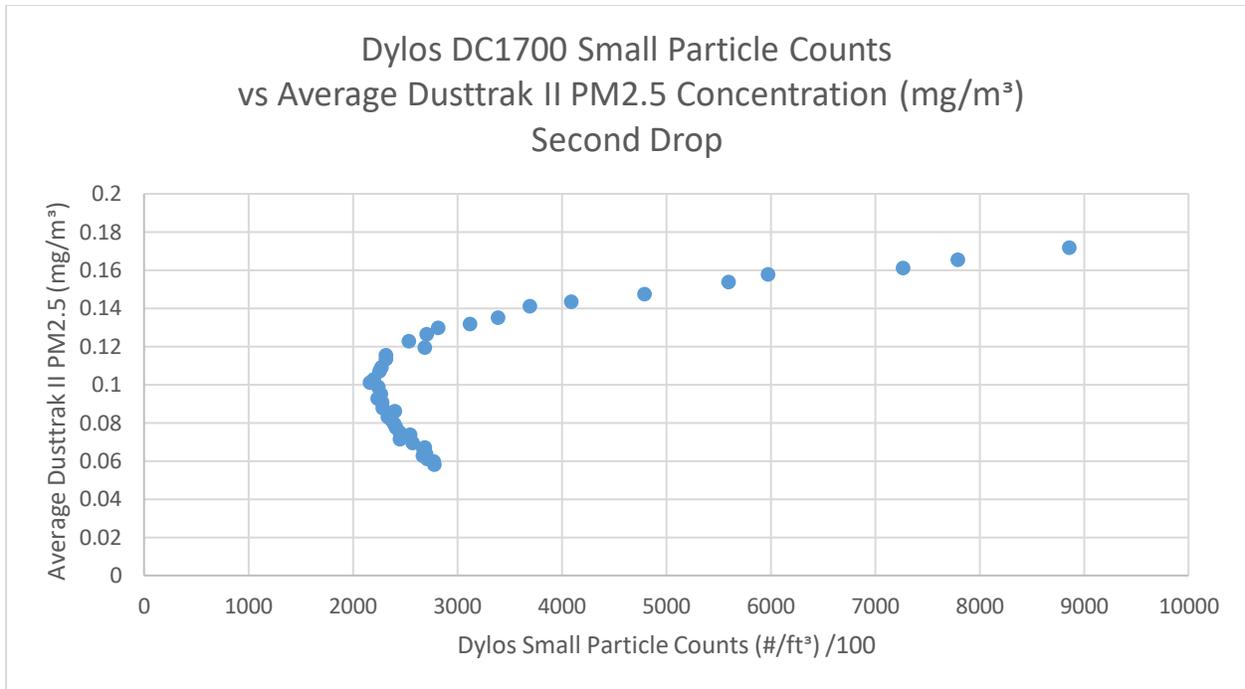
Figure B.22(d) Dylos DC1700 Small Particle Counts vs.
Average Dusttrak II PM$_{2.5}$ Concentration (mg/m^3) for the Second Drop

(b) Dylos DC 1700 Large Particle Count vs. Dusttrak II PM$_{2.5}$ Comparison Analysis:

Case i): Results from Dylos DC1700 large particle counts vs. average results from the Dusttrak PM$_{2.5}$ concentrations (mg/mm the Dusttrak PMcle coun are shown in Figure B.22 (e). There is no clear correlation during this first spike in the data between instruments. This is in contrast to what was observed in the comparison with the small particle counts in

Figure B.22(a).
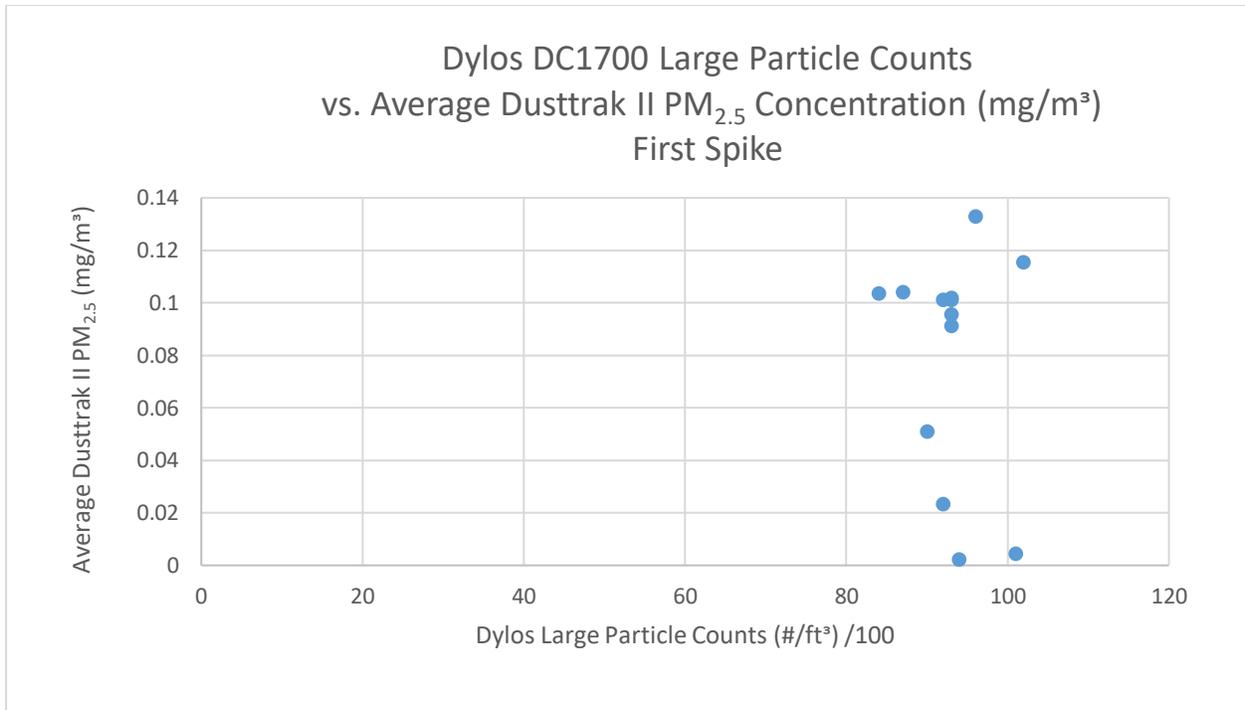
Figure B-22(e). Dylos DC1700 Large Particle Counts
vs. Average Dusttrak II PM$_{2.5}$ Concentration (mg/m^3) for the First Spike

Case ii): Results from Dylos DC1700 large particle counts vss average results from the Dusttrak PM$_{2.5}$ concentrations (mg/m^3) during the first drop are shown in

Figure B.22 (f). During the first drop, the comparison shows a linear correlation with R² = 0.8301. In contrast to the comparison with the small particles and the Dusttrak II data (Figure B.22 (b)), during the drops, the correlation becomes linear across all ranges of large particle counts measured by the Dylos DC1700.
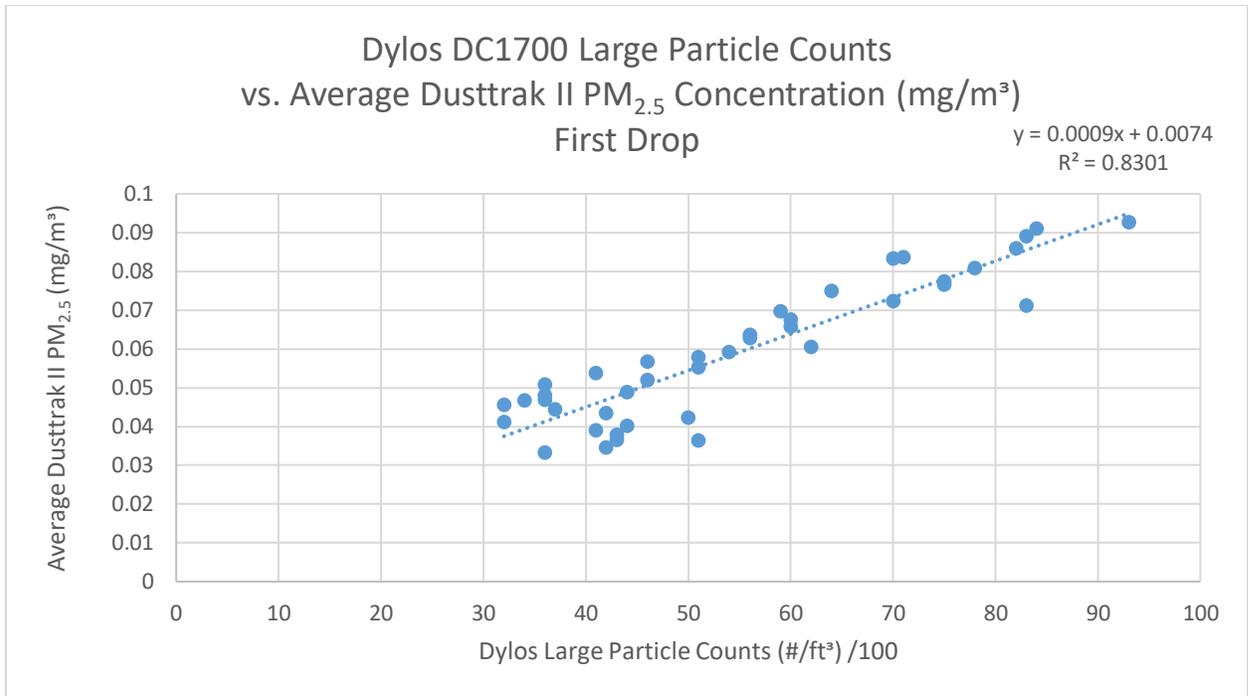
Figure B.22(f) Dylos DC1700 Large Particle Counts vs. Average Dusttrak II PM$_{2.5}$ Concentration (mg/m^3) for the First Drop

Case iii): Results from the Dylos DC1700 large particle counts vs. average results from the Dusttrak PM$_{2.5}$ concentrations (mg/m^3) during the second spike are shown in Figure B.22 (g). Although the correlation now appears linear, the relationship is too weak (R^2 = 0.5261) to draw any meaningful conclusions.
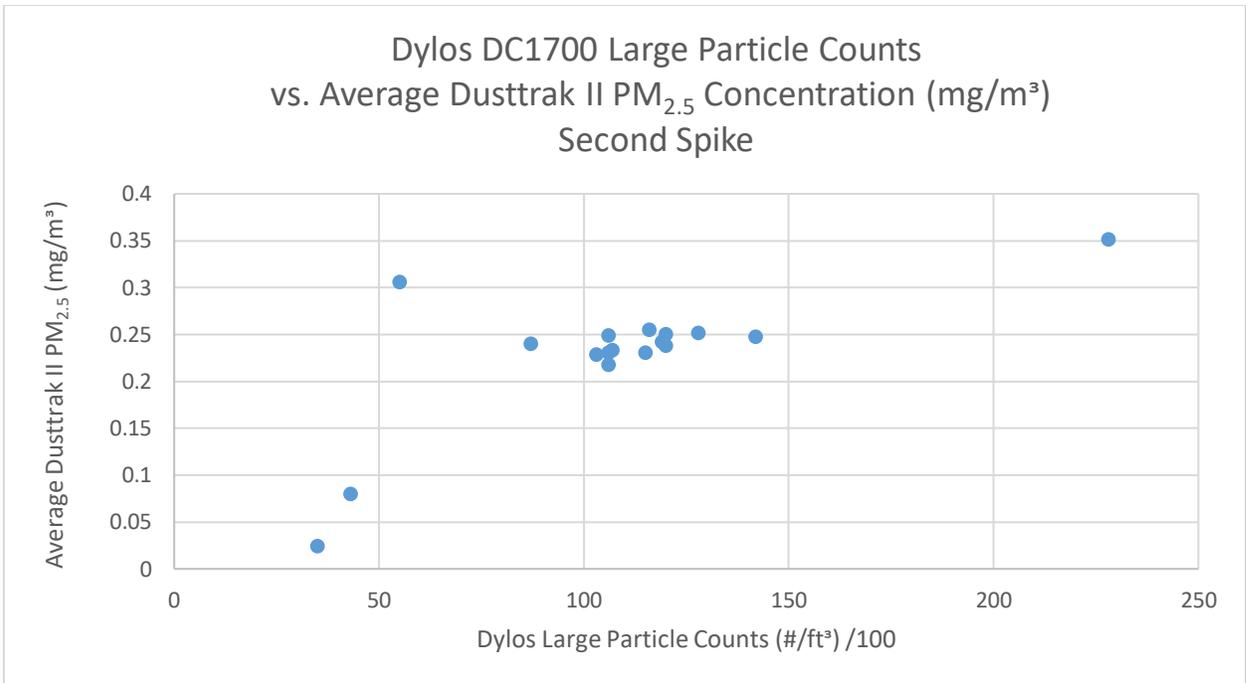
Figure B.22(g) Dylos DC1700 Large Particle Counts
vs. Average Dusttrak II PM$_{2.5}$ Concentration (mg/m^3) for the Second Spike

Case iv): Results from Dylos DC1700 large particle counts vs. average results from the Dusttrak PM$_{2.5}$ concentrations (mg/m^3) during the second drop are shown in Figure B-22(h). Once particulate levels drop for the second time, the correlation is observed to be linear when PM$_{2.5}$ concentrations measured by the Dusttrak II instruments and large particle counts measured by the Dylos DC1700 are compared. The correlation coefficient R^2 = 0.9082, indicates a strong correlation. Unlike the case of the small particle counts, the large particle counts display a linear correlation with the Dusttrak II concentrations across the entire drop-off period and not just shortly after, as levels abruptly decay from their maximum point.
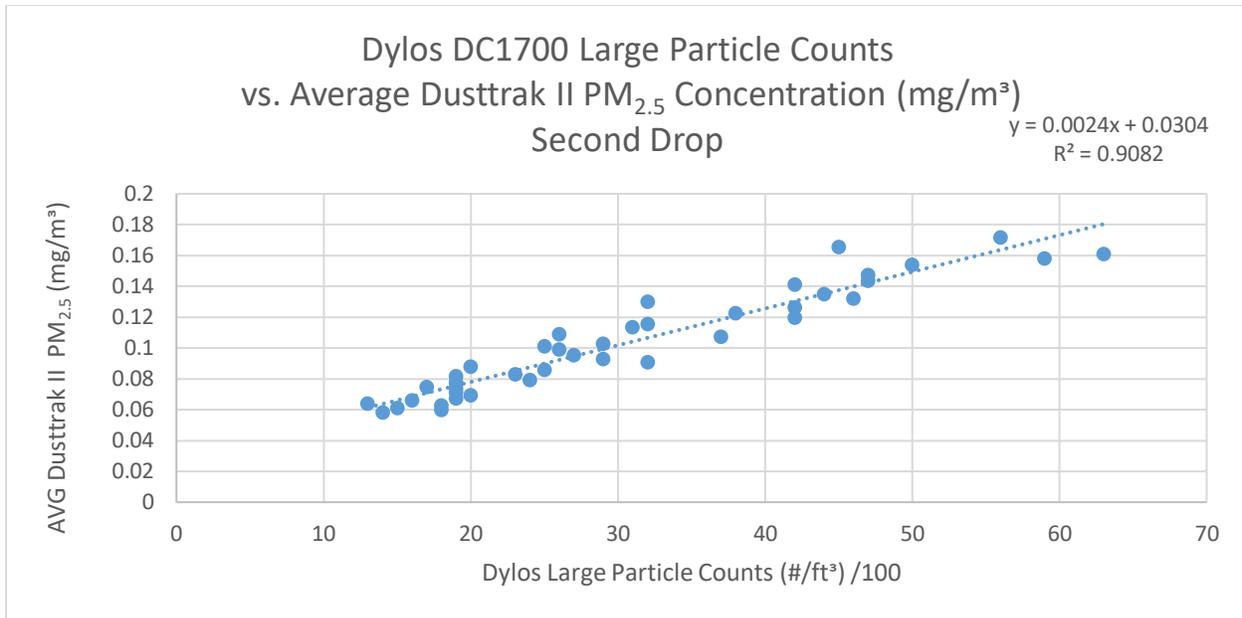
Figure B.22 (h) Dylos DC1700 Large Particle Counts
vs. Average Dusttrak II PM$_{2.5}$ Concentration (mg/m^3) for the Second Drop

Concluding Observations:

It is clear from the above analysis of each separate component period, Cases: a(i) to a(iv) for small particles and Cases b(i) to b(iv) for large particles, the instrument comparison between the Dylos DC1700 and the Dusttrak II cannot yield a clear calibration factor to describe their relationship. Although strong linear relationships exist for Cases a(i) and a(iii) as well as for Cases b(i) and b(iii), comparisons of data in Cases a(ii), a(iv), b(i), and b(iii) show nonlinear or weakly linear correlations. It would have been ideal, as was the case with the UNI-T comparison with Dusttrak II, to have a single calibration factor for the complete period of data collection for either the large particle count (Cases a(i) to a(iv) combined) or the small particle count (Cases b(i) to b(iv) combined). From the data analysis above, however, this was not possible.

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It was observed that when woodstove particulate is freshly introduced into a room during the first spike, there was a clear linear correlation between the small particle counts measured by the DC1700 and the $PM_{2.5}$ concentrations measured by the Dusttrak II. This linearity disappears only after concentration levels are given time to drop off, indicating that the rate at which the small particle count diminishes will affect how well it can describe equivalent $PM_{2.5}$ concentrations as measured by the Dusttrak II. When the levels quickly decay from an initial peak, they approximate a linear relationship. When levels decay slower after some time, this linear relationship disappears.

It is not clear whether the linearity observed in the correlations, as the concentrations begin to drop from their peak values in Cases a(ii) and a(iv), is due to differences in how the Dylos DC1700 observes decaying particulate when compared to the Dusttrak II and not due to a $PM_{2.5}$ threshold. In Case a(ii) the linear correlation roughly spans a $PM_{2.5}$ concentration range of 0.07 to 0.10 mg/m³, while in Case (iv) the linear correlation roughly spans a $PM_{2.5}$ concentration range of 0.13 to 0.18 mg/m³. There is no set range of Dusttrak II $PM_{2.5}$ concentrations in which a linear correlation will exist with the Dylos DC1700 small particle count. The linearity only arises either when a significant amount of particulate is freshly introduced to the device, or as seen in Cases a(ii) and a(iv), when $PM_{2.5}$ levels are quickly decaying.

This finding is corroborated by Dacunto et al. (2015, p.1) who report "The relationship between the exponentially-decaying Dylos particle counts and $PM_{2.5}$ mass concentration can be described by a theoretically-derived power law with source-specific empirical parameters. A linear relationship

(calibration factor) is applicable to fresh or quickly decaying emissions (i.e., before the aerosol has aged and differential decay rates introduce curvature into the relationship)."

In the testing by Dacunto et al. (2015), which measured across a wider variety of particulate types and not just wood smoke, the Dylos data were represented by the difference in the small particle count and large particle count.

In our study, where $PM_{2.5}$ is due solely to wood smoke, this difference is virtually identical to the small particle count number alone. This can be seen when comparing the ranges for the small and large particle count in Figure B.17. In this sense, using solely the small particle counts as opposed to the difference between the small and large counts suffices.

As in the study conducted by Dacunto et al. (2015), analysis of our data shows a power-law relationship which approximates linearity during the first spike (Case a(i)) as well as immediately after start in Case a(ii)), as $PM_{2.5}$ levels quickly decay from their highest point on the right side of the figure. This linear relationship once again disappears when levels drop at a steadier rate, as seen on the left side Figure B.22(b). The existence of the inflection point shown by the red arrow in Figure B-17 shows how the Dylos DC1700 data closely resemble $PM_{2.5}$ data concentrations across the entire test and drop off at a fixed decay rate, as shown by the Dusttrak II data.

Dacunto et al. (2015, p. 1) concluded that "The Dylos Air Quality Monitor is likely most useful for providing instantaneous feedback and context on mass particle levels in home and work situations for field-survey or personal awareness applications." This instrument, however, would not be beneficial in studies that require 24-hr average measurements, as the accuracy of the Dylos relies on certain conditions which can only be met for fractions of the entire 24-hr period. This situation would introduce large errors, and results cannot be compared from home to home in any broader study.

Overall, when comparing the results from Dylos DC1700 to those from Dusttrak II instruments, it can be seen that the Dylos DC1700 cannot be used to develop equivalent $PM_{2.5}$ data as that of the Dusttrak II, as the relation only approaches linearity under limited conditions. In the case of the Dylos DC1700 particle count, this occurs only when $PM_{2.5}$ is freshly introduced or when it is quickly decaying, an observation which is also supported by Dacunto et al. (2015). For the large particle count comparison, there is no consistent relationship between the results from the two types of instruments other than that they are characterized by a fixed decay constant during their drop-off stage (Cases b(i) to b(iv)). In any case, even under ideal conditions for the Dylos DC1700, the correlation coefficient when compared to the Dusttrak II was no more than 0.928, which occurred only when fresh particulate matter was introduced to the DC1700 during the first spike (Case a(i)) in the small particle count. Because their linear correlation is limited to only specific time frames, no single calibration factor can be determined for an entire period of testing during which $PM_{2.5}$ levels may change abruptly and significantly, depending on occupant behaviour. Because no clear calibration factor can be determined across the entirety of the data collection period, no accurate indoor Dusttrak equivalent $PM_{2.5}$ concentration data can be synthesized from either the small or large particle data collected by the Dylos DC1700.

The question, however, remains whether the small or large particles detected by the Dylos DC1700 best describe $PM_{2.5}$. In two papers, one by Dacunto et al. (2015) and one by the World Air Quality Index Team (AQIT, 2014, p. 1), the small particle count, or the difference between the small and large particle size best approximates $PM_{2.5}$. The World Air Quality Index Team explains: "The most common sizes in the Air Quality industry is 2.5 µm (referred as $PM_{2.5}$) and 10 µm (referred as PM_{10}). In this case, does it really make sense for the Dylos to measure particulates as small as 0.5 µm? Shouldn't it instead measure 2.5 µm and 10 µm particulates? Actually, maybe this 0.5 µm claim is more a marketing argument rather than a technology argument, since, from the empirical data, what it measures is closer to 2.5 µm and 10 µm rather than 0.5 µm and 2.5 µm."

Interestingly, Dylos describes the particle sizes through ambiguous descriptors labelled: >0.5µm and >2.5 µm. More often than not, these are interpreted not as $PM_{0.5}$ and $PM_{2.5}$, which describe particles as small as 0.5µm and 2.5 µm, respectively, but instead indicate the number of particles larger than 0.5µm and 2.5 µm being detected. In this sense, their difference will be the most accurate representation of $PM_{2.5}$.

 As mentioned earlier, during the entire period of testing in our study, the number of large particles detected by the Dylos DC1700 are negligible when compared to the small particles detected by the same instrument. This is indicative that their difference is virtually identical to the small particle count alone. Because the large particle counts decay more constantly, when compared to the small particle counts, the particle count approximates a linear relationship to that of

the Dusttrak II instrument, which is also characterized by a constant decay. The existence of the inflection point in the small particle count data only indicates a lack of decay like behavior. This similarity is the only common observation between the large particle count by Dylos DC1700 and Dusttrak II.

In contrast, with the Dylos DC1700, the $PM_{2.5}$ data collected by the UNI-T, surprisingly, proved to be consistently linearly correlated with the Dusttrak II data across the entire period of data collection. This correlation prevailed despite any differences in how environmental effects may affect each instrument's ability to measure wood smoke particulate differently. During the entire duration of the test, except for the initial ambient values, the comparison yields a correlation coefficient $R^2 = 0.9639$. Although the UNI-T does not have data logging capabilities, utilizing the SSOCR method to tabulate its measurements, as discussed earlier in the study, costs very little to conduct simple and effective data analysis using a single calibration factor, obtained through instrument comparison. This factor can be applied to the UNI-T data in order to synthesize Dusttrak II equivalent data with little added error.

$PM_{2.5}$ is often described as harmful to respiratory health and, when it is discussed, a 24-hr average mass concentration is typically used as bench mark. It is these 24-hr average levels that are of most interest when discussing a healthy air index.

A 24-hr average mass concentration was synthesized from UNI-T data (mg/m³) using the UNI-T

conversion calibration curve. This was then compared with Dusttrak II 24-hr average data. The

procedure used during this conversion process is outlined in Table B.6.

Table B.6. Steps in Order to Test Viability of Using the UNI-T, as Opposed to the Dusttrak II, In Order to Measure 24-hr PM$_{2.5}$ Indoor Concentrations in a Home Actively Using a Woodstove

Stages	UNI-T	Dusttrak II 8530
1	UNI-T displays data (no internal data logging) at 1-second internal built interval	Dusttrak II Instruments: #4, #6, #2 1-min interval averaged for single data set to compare instruments with
2	**SSOCR method** Webcam captures image every 5s 5-s equivalent interval snapshot data	
3	Excel converted UNI-T 60-s equivalent interval Data	
4	Comparison between Dusttrak II average Data and UNI-T data yields **calibration curve**	
5	Calibration curve conversion Dusttrak II equivalent Data 60-s interval	
6	Excel converted from a 2.5+ hrs test period to a 24-hr average UNI-T Data (mg/m³)	Excel converted from a 2.5+ hrs test period to a 24-hr average Dusttrak II Data (mg/m³)
7	**Compare**: 24-hr average UNI-T data with 24-hr average Dusttrak II sata	

The comparison made in Stage 7, listed in Table B.6, will yield a rough idea of how much error

can be additionally introduced if a UNI-T Instrument is used instead of an actual Dusttrak II to

measure 24-hr average PM$_{2.5}$ concentrations. The UNI-T is used along with an implementation of

the SSOCR method and then also a calibration curve to synthesize Dusttrak II equivalent 24-hr

average data.

Calibration Curve

The following calibration curve, used to convert UNI-T data into Dusttrak II Data, was obtained from Figure B.15 which shows indoor $PM_{2.5}$ concentrations due to wood smoke:

y = 4.727x - 0.181

Where: y = Synthesized Dusttrak II concentration (mg/m³)

x = original UNI-T $PM_{2.5}$ data (mg/m³)

Using this calibration curve, a new Dusttrak II equivalent data set was created. Because the units match on either side of the calibration curve, 24-hr average data can be stipulated as well.

Upon averaging the data over the entire test period, a $PM_{2.5}$ average concentration is determined over that period. Comparing the average $PM_{2.5}$ concentrations over the entire test period provides an indication of the error introduced by using the less expensive UNI-T over a Dusttrak II..

The % error is calculated as follows:

$$\% \ error = \left| \frac{\#_{experimental} - \#_{theoretical}}{\#_{theoretical}} \right| \cdot 100$$

The results are shown in Table B.7.

Table B.7. Error Introduced from Using 24-hr Average PM$_{2.5}$ Concentrations (mg/m³)
from Synthesized Data Set Using a Calibration Curve

Comparison of Synthetic vs. Actual Dusttrak II Data Sets (Created using a Calibration Curve for UNI-T data)			
Woodstove Use (hours)	Synthesized Average PM$_{2.5}$ (mg/m³)	Dusttrak II Average PM$_{2.5}$ (mg/m³)	Error % introduced
2.5 (Burn Test)	0.102221383	0.096539258	
24 (Estimate)	0.102221383	0.096539258	5.56

As shown in Table B.7, when using the UNI-T to ultimately determine Dusttrak II equivalent 24-hr average data, utilizing the calibration curve, the additional error introduced is estimated to be just 5.56%.

One limitation of this method is that only a 2.5-hr test duration was used to determine the calibration curve. Additionally, the woodstove may not be in constant use over a 24-hr period, as it was in the 2.5-hr burn test. In future work, a 24-hr burn cycle test is recommended with many instances of the woodstove door being open and closed. This would be an ideal test for a more accurate determination of the calibration curve. Additionally, images of the UNI-T display should be captured every second, as opposed to every five seconds, to improve the accuracy of the calibration curve.

In this study, the ability of inexpensive instruments to accurately measure PM$_{2.5}$ was assessed through a comparison of the data captured during an identical burn period by the various instruments. The Dylos DC1700 proved to be inadequate in determining Dusttrak II equivalent

PM$_{2.5}$ concentrations across the entire period of testing. In contrast, the UNI-T data displayed a strong linear correlation with the Dusttrak II data across the entire period of testing. Because of this strong correlation, a calibration curve could be developed in order to describe their relationship. This calibration curve could then be used to synthesize a Dusttrak II equivalent data set from measurements displayed on the UNI-T monitor. When comparing the true Dusttrak II data set to the newly synthesized data set, only a 5.56% additional error was introduced in a 24-hr PM$_{2.5}$ average. The Dylos DC1700, however, can only be used to accurately determine Dusttrak II equivalent PM$_{2.5}$ concentration data when instantaneous PM$_{2.5}$ levels are measured as the particulate is freshly introduced to the device, and also when these PM$_{2.5}$ levels are quickly decaying, shortly after their spikes. At all other points in the testing period, a poor correlation between the actual Dusttrak II data and the Dylos DC1700 PM$_{2.5}$ data (small particle counts) exists. Thus, the Dylos DC1700 cannot be used to measure live PM$_{2.5}$ concentrations accurately across an entire period of testing. Additionally, any comparisons made with a health index standard, which is based on 24-hr average PM$_{2.5}$ concentration data, will be misleading.