Measurement and analysis of toxic volatile organic compounds (VOC) and polycyclic aromatic hydrocarbons (PAH) at Singapore's integrated transport hubs (ITH)

Lee, Jeff Suk Hyun

2021

Lee, J. S. H. (2021). Measurement and analysis of toxic volatile organic compounds (VOC) and polycyclic aromatic hydrocarbons (PAH) at Singapore's integrated transport hubs (ITH). Master's thesis, Nanyang Technological University, Singapore.
<https://hdl.handle.net/10356/153597>

<https://hdl.handle.net/10356/153597>

<https://doi.org/10.32657/10356/153597>

This work is licensed under a Creative Commons Attribution-NonCommercial 4.0 International License (CC BY-NC 4.0).

Downloaded on 26 Apr 2023 18:12:44 SGT

**NANYANG
TECHNOLOGICAL
UNIVERSITY**

SINGAPORE

**MEASUREMENT AND ANALYSIS OF TOXIC VOLATILE
ORGANIC COMPOUNDS (VOC) AND POLYCYCLIC
AROMATIC HYDROCARBONS (PAH) AT SINGAPORE'S
INTEGRATED TRANSPORT HUBS (ITH)**

LEE SUK HYUN, JEFF

SCHOOL OF CIVIL AND ENVIRONMENTAL ENGINEERING

2021

**MEASUREMENT AND ANALYSIS OF TOXIC VOLATILE
ORGANIC COMPOUNDS (VOC) AND POLYCYCLIC
AROMATIC HYDROCARBONS (PAH) AT SINGAPORE'S
INTEGRATED TRANSPORT HUBS (ITH)**

LEE SUK HYUN, JEFF

School of Civil and Environmental Engineering

A thesis submitted to the Nanyang Technological University
in partial fulfilment of the requirement for the degree of
Master of Engineering

Statement of Originality

I hereby certify that the work embodied in this thesis is the result of original research, is free of plagiarised materials, and has not been submitted for a higher degree to any other University or Institution.

05/07/2021

Date

0/13/21

LEE SUK HYUN

Supervisor Declaration Statement

I have reviewed the content and presentation style of this thesis and declare it is free of plagiarism and of sufficient grammatical clarity to be examined. To the best of my knowledge, the research and writing are those of the candidate except as acknowledged in the Author Attribution Statement. I confirm that the investigations were conducted in accord with the ethics policies and integrity standards of Nanyang Technological University and that the research data are presented honestly and without prejudice.

03/07/2021

Date

FANG MINGLIANG

Authorship Attribution Statement

This thesis contains material from a paper submitted in the following peer-reviewed journal in which I am listed as an author.

Most of the contents will be published as Lee, S. H., Shen, J., Tan, S. T., Ng, L. C., Fang, M., & Jia, S. Effects of Architecture Structure on Volatile Organic Compound and Polycyclic Aromatic Hydrocarbon Diffusion in Singapore's Integrated Transport Hubs. *Chemosphere*.

The contributions of the co-authors are as follows:

A/Professor Fang Mingliang provided the initial project direction and edited the manuscript drafts.

Dr. Jia Shenglan supervised this research from beginning to the end. She designed laboratory work, collected air samples at bus interchanges, conducted instrumental analyses and helped the author the data analyses.

Ms. Shen Joanna assisted in the air sampling and provided guidance in the interpretation of the air pollutant data.

Mr. Tan Sze Tat and Ms. Ng Lee Ching reviewed the manuscript and give professional advice to author.

05/07/2021

Date

0/3/21

LEE SUK HYUN

ABSTRACT

Millions of commuters should spend time waiting for their buses at bus interchanges every day in Singapore. Environmental exposure to vehicular emissions at transport hub and associated risks are of public concern. To date, limited studies have focused on the indoor air quality at Integrated Transport Hubs (ITHs), which are fully air-conditioned bus interchanges adjoined with subway stations and commercial facilities.

In this study, 12 representative Volatile Organic Compounds (VOCs) and 16 EPA priority pollutant Polycyclic Aromatic Hydrocarbons (PAHs) were selected, collected from 9 bus interchanges at both indoor waiting area and bus park sampling points, and then analyzed. The daily VOCs and PAHs exposure for commuters were estimated by considering inhalation and dermal doses using Monte Carlo simulation ($n=100,000$), and human health risks for the exposure to selective pollutants were further evaluated.

The median concentration of total VOCs (TVOCs), total gaseous PAHs (TgPAHs) and total particulate-bound PAHs (TpPAHs) in indoor areas were $30.42 \mu\text{g}/\text{m}^3$, $18.99 \text{ ng}/\text{m}^3$ and $1.38 \text{ ng}/\text{m}^3$, respectively. Besides, the median concentrations for bus parks were $20.24 \mu\text{g}/\text{m}^3$, $24.26 \text{ ng}/\text{m}^3$ and $1.94 \text{ ng}/\text{m}^3$ in sequence. In general, the pollutant concentration levels were not significantly different between waiting areas and bus parks, but showed positive correlations (TVOCs: $r = 0.90$, $p < 0.001$; TgPAHs: $r = 0.75$, $p < 0.001$; TpPAHs: $r = 0.39$, $p < 0.05$), suggesting some air exchange dependency between the spaces. The high correlations between VOCs and gPAHs components were found, which implied the bus exhaust might be the dominant source of the pollutants. The indoor waiting area to bus park pollutant concentration ratio (I/B ratio), a new indicator representing the concentration relationship between indoor and bus park samples, was calculated for each pollutant group, and lower I/B ratios of pollutants were found (TVOCs: 0.98; TgPAHs: 0.76; TpPAHs: 0.71) in the bus interchanges with fully sheltered bus parks than those with open/partially enclosed bus parks (TVOCs: 1.28; TgPAHs: 1.31; TpPAHs: 0.90). The findings in this study imply that the fully sheltered bus park may affect the dilution of air pollutants resulting in the accumulation in adjacent indoors.

From the result of the dose estimation, it was found that inhalation was the dominant exposure route of VOCs accounted for 99%, while dermal absorption was the major route of PAHs accounted for 82%. Most concentration levels of VOCs at Singapore's ITHs were found to be below EPA's reference concentrations (RfC) except for naphthalene, but the maximum concentration was less than the twice of its safety benchmark. However,

benzene, ethylbenzene, styrene and naphthalene exceed the stringent OEHHA benchmarks, as similar to other countries' reports. The concentration levels of PAHs at ITHs were found to be within their safety benchmarks through the estimation of Incremental Lifetime Cancer Risk (ILCR) for carcinogenic risk and the comparison with EPA's RfC for noncarcinogenic risk.

ACKNOWLEDGEMENT

This air pollution research was funded and assisted by the Singapore National Environment Agency (NEA) and the author is grateful for their unstinting support. Moreover, the author would like to express his enormous gratitude to all people who had guided, taught and supported him throughout the project. This valuable opportunity helped him to broaden his perspective of environmental problems and motivated him to keep learning from various research subjects.

First of all, the author sincerely appreciated that Professor Fang Mingliang let him involve in this project. While conducting this project, Professor Fang led our team with his profound insight and broad knowledge of various chemicals. His wide experience was absolutely supportive to initiate this project and to make steady progress.

Secondly, the author would like to express his deep gratitude to his mentor, Dr Jia Shenglan, for her exceptional concern and caring about this project. Dr Jia made substantial efforts on this project from beginning to end, so the author could be able to deal with air sampling and analysis instrument, and also to analyze data of air pollutants.

Lastly, the author feels many thanks to research officers of NEA, Joanna Shen and Nazeem M Kamal, for supporting air sampling at ITHs and their helpful advice. In addition to this, a Final Year Project Student, Jeremy Tan, contributed to this project and the author feels so grateful for his enthusiastic participation and constructive feedback.

TABLE OF CONTENTS

Abstract	v
Acknowledgement	vii
1. List of Tables	x
2. List of Figures	xi
3. Chapter 1 – Introduction	1
3.1 Background	1
3.2 Objectives & Scope	3
4. Chapter 2 – Literature Review	4
4.1 Volatile Organic Compounds	4
4.1.1 Definition	4
4.1.2 Properties of VOCs	4
4.1.3 Sources of VOCs	6
4.1.4 The fate of VOCs	7
4.1.5 The Exposure Routes of VOCs	10
4.1.6 Human Health Risks of VOCs	10
4.2 Polycyclic Aromatic Hydrocarbons	16
4.2.1 Definition	16
4.2.2 Properties of PAHs	16
4.2.3 Sources of PAHs	18
4.2.4 The Fate of PAHs	19
4.2.5 The Exposure Routes of PAHs	23
4.2.6 Human Health Risks of PAHs	23
5. Chapter 3 – Experimental Design	25
5.1 Sampling Sites	25
5.2 Sampling Procedures	27
5.3 Analytical Procedures	28
5.4 Chemical and Reagents	29
5.5 Quality Control	29
5.6 Statistical Analysis	30

5.7 Health Risk Assessment.....	30
5.7.1 Dose Calculation	32
5.7.2 Risk Characterization	36
6. Chapter 4 – Results and Discussion.....	40
6.1 Volatile Organic Compounds	40
6.1.1 VOC concentrations at Indoor and Bus Park	40
6.1.2 Effect of Bus Interchange Structure on VOCs migration.....	41
6.2 Gaseous Polycyclic Aromatic Hydrocarbons	43
6.2.1 gPAHs concentrations at Indoor and Bus park	43
6.2.2 Effect of Bus Interchange Structure on gPAHs migration	45
6.3 Particulate Polycyclic Aromatic Hydrocarbons.....	47
6.3.1 Indoor and Bus park pPAHs concentrations	47
6.3.2 Effect of Bus Interchange Structure on pPAHs migration	48
6.4 Effect of Bus Park Structure on I/B Air Pollutants Ratios	51
6.5 Risk Assessment for VOCs and PAHs exposure	52
6.5.1 Dose Calculation	52
6.5.2 Risk Characterization	54
7. Chapter 5 – Conclusions and Recommendations	59
7.1 Summary of Findings and Interpretation	59
7.2 Limitations and Recommendations.....	61
8. Chapter 6 – References.....	62
9. Chapter 7 – Appendix.....	70
A. Instrument Settings	70
a. TD Settings for VOCs and PAHs	70
b. GC-MS Settings for VOCs and PAHs.....	71
B. Method Detection Limits (MDLs)	72
C. The total daily doses (D_{total}) of VOCs and PAHs	72
D. Concentrations of VOCs and PAHs in Various Microenvironments.....	77

1. LIST OF TABLES

Table 1. Properties of selective 12 VOCs	5
Table 2. Partitioning Coefficient of selective 12 VOCs at 25°C	8
Table 3. Noncarcinogenic and Carcinogenic Health Effect of VOCs.....	13
Table 4. Properties of selective 15 PAHs.....	17
Table 5. Partitioning Coefficient of selective 15 PAHs at 25°C.....	22
Table 6. Summary of human health risks from exposure to PAHs.....	24
Table 7. The backgrounds of Singapore's 9 bus interchanges.....	25
Table 8. Equations for the calculations of inhalation and dermal absorption dose	32
Table 9. The mean values and standard deviations of inhalation rate (IR), body surface area (SA) and body weight (BW) used for dose calculations	33
Table 10. Equations for the calculation of pollutant concentration in air	33
Table 11. Equations applied to estimate indoor air transdermal coefficients	34
Table 12. Partition coefficients of selective VOCs and PAHs.....	35
Table 13. A summary of adjusted NSRL and MADL of analytes	37
Table 14. EPA's Reference Concentration (RfC) for selective VOCs.....	37
Table 15. Toxic Equivalent Factors (TEFs) of selective PAHs	38
Table 16. An equation of ILCR for PAH risk assessment	39
Table 17. Summary of VOCs concentrations in Singapore's 9 bus interchanges.....	40
Table 18. Pearson Correlation Coefficients between VOCs components (n=75).....	43
Table 19. Summary of gPAHs concentrations in Singapore's 9 bus interchanges	44
Table 20. Pearson Correlation Coefficients between gPAHs components (n=49)	46
Table 21. Summary of pPAHs concentrations in Singapore's 9 bus interchanges	47
Table 22. Pearson Correlation Coefficients between pPAHs components (n=50)	50
Table 23. The mean (SD) doses of VOCs ($\mu\text{g/kg/day}$) and PAHs (ng/kg/day) by age groups.....	52
Table 24. Summary of Hazard Quotients (HQ) for VOCs based on OEHHA's guidelines	54

Table 25. Summary of Hazard Quotients (HQ) for VOCs based on EPA's guidelines..	56
Table 26. Incremental lifetime cancer risk (ILCR) of PAHs at bus interchanges.....	57
Table 27. Summary of Hazard Quotients (HQ) for PAHs based on EPA's guidelines ..	58

2. LIST OF FIGURES

Figure 1. The photolysis pathway of gaseous VOCs in troposphere	9
Figure 2. Layouts of Singapore's 9 bus interchanges	26
Figure 3. Fitting of sorbent tubes for pPAHs analysis	28
Figure 4. Box plot of indoor and bus park VOC concentrations at bus interchanges categorized as fully sheltered (a) and open/partially enclosed (b)	41
Figure 5. Plot of TVOCs concentration levels from indoor and bus park air samples (n=26)	42
Figure 6. Box plot of indoor and bus park gPAHs concentrations at bus interchanges categorized as fully sheltered (a) and open/partially enclosed (b)	45
Figure 7. Plot of TgPAHs concentration levels from indoor and bus park air samples (n=17)	46
Figure 8. Box plot of indoor and bus park pPAHs concentrations at bus interchanges categorized as fully sheltered (a) and open/partially enclosed (b)	49
Figure 9. Plot of TpPAHs levels from indoor and bus park air samples (n=25).....	49
Figure 10. Box plot of I/B ratios of TVOCs, TgPAHs and TpPAHs by bus interchanges classified by two categories including fully sheltered and open/partially enclosed.....	51
Figure 11. Relative contribution of VOCs and PAHs dose from inhalation and dermal absorption.....	53

3. CHAPTER 1 – INTRODUCTION

3.1 Background

Singapore, one of the most densely populated cities in the world, has controlled high traffic volume and associated air pollution by relatively rigid traffic policies (UN, 2019). In 1990, the Singapore government introduced an acquisition tax policy called Certificate of Entitlement for vehicles to regulate the total number of them (Omar, M., 2006). Due to the expensive price of the certification, Singapore citizens prefer to use public transportation and, as a consequence, its transit network has become developed and complexed. Land Transport Authority (LTA, 2019) under the Singapore government recently compared the number of cars per 100 persons of five big cities in the world, Singapore, London, New York, Tokyo and Hong Kong. Among the five cities, Singapore ranked at fourth and its value was only a third of London's, even though Singapore's GDP was the highest. Furthermore, the yearly increased number of public transport ridership would explain how many Singaporeans have depended on public transports (LTA, 2015). As the traffic network has been intricated, the Singapore government has redeveloped old bus interchanges and combined them with subway stations and commercial buildings, such as shopping malls. That structure is called Integrated Transport Hub (ITH). At the multipurpose facility, people can take or transfer transportation and also are able to shop and dine with convenience, because all of buildings are internally connected.

However, air pollution issues are also raised at ITH. Before the renovation, all bus interchanges were located outdoor, so air pollutants from the bus exhaust gas were naturally diluted. Whereas, after the renewal, all waiting areas were moved to indoor and most of the boarding areas and bus parks were sheltered. Consequently, people would be readily exposed to atmosphere pollution matters generated from a lot of buses. In Singapore, 99.5% of buses in operation are using diesel fuel and their exhaust gas contains hazardous chemical compounds (LTA, 2020). The major components of diesel exhaust gas, nitrogen, carbon dioxide, water vapor and oxygen, do not matter, but other components from incomplete combustion of diesel fuel, such as carbon monoxide(CO), hydrocarbons (HC), nitrogen oxides(NOx) and sulfur dioxide(SO₂) could be a problem (Reşitoğlu, I. A. et al., 2014; Schauer, J.J. et al., 1999). The pollutants can cause short-term symptoms, such as coughing, headache and nausea, and people could be taken with more severe illnesses like lung cancer, if they are exposed to the pollutant over the long term (Sydbom, A. et al., 2001). Although the air quality of bus park area has been properly managed by natural ventilation (BCA, 2010), the bus exhaust could be continuously

transported to indoor and exacerbate the air quality of bus interchanges and connected facilities. Moreover, the increasing number of users might aggravate the indoor air quality at ITH. Anthropogenic activities, such as smoking, using air fresheners, applying paints and cooking activities would generate Volatile Organic Compounds (VOCs) and Polycyclic Aromatic Hydrocarbons (PAHs), and might deteriorate indoor air quality (Steinemann, A., 2017; Purcaro, G. et al., 2006; Kim, Y.M. et al., 2001; Maroni M. et al., 1995).

As a case study, ITH air pollution research was aimed for two air pollutant groups, VOCs and PAHs. In particular, benzene and toluene, known as VOC constituents in the diesel exhaust, have been identified as carcinogen and reproductive toxicants (OEHHA, 2019; Kim, Y. M. et al., 2001). Benzo(a)pyrene, benz(a)anthracene, benzo(b)fluoranthene and chrysene, derived from diesel-fueled vehicles (Shen, H. et al., 2013; WHO, 2010), are well-known carcinogens, mutagens, and teratogens (Kim, K. et al., 2013; Boström, C. et al., 2002). Although the extent of health effect will depend on the toxicities and the levels of the compounds, the exposure to bus exhaust in everyday life may cause harmful influence on the commuters' health.

So far, several air pollution studies at bus interchanges have been conducted for certain pollutants, such as particulate matters (Cheng, Y. H. et al., 2012; Yang, F. et al., 2012), common VOCs (Golchorshidi, F. et al., 2019; Kim, Y. M. et al., 2001) and EPA's 16 PAHs (Kishida, M. et al., 2008; See, S. W. et al., 2006), and most of the studies collected air samples only at waiting area to simulate human exposure to the pollutants. However, there is still a lack of studies regarding the effects of the building architectural design on the bus exhaust transport, although the source is getting closer to indoor space for the purpose of convenience.

Therefore, this study aimed mainly to identify whether the building envelope of bus interchanges could affect the migration of air pollutants. The concentration levels of each air pollutant (VOCs and PAHs) in indoor and bus park samples were compared and the further converted to the indoor concentration to bus park ratio (I/B ratio) for the comparison of architectural design of bus interchanges (fully sheltered or open/partially enclosed). Additionally, the daily exposure to VOCs and PAHs were estimated for commuters due to the significance of dermal exposure to lipophilic air pollutants (Jia, S. et al., 2019; Gaspar, F. W. et al., 2014).

3.2 Objectives & Scope

This project aimed to characterize the air quality of ITHs in Singapore. From January to April 2019, 9 of ITHs (100%) were investigated and their characteristics of VOCs and PAHs were summarized. There were two major reasons why this air pollution research was conducted.

Firstly, this research was to measure the air quality of ITH for both VOCs and PAHs. Since the ITH was comprised of subway station, bus interchange, commercial buildings and housing facilities, if the air pollutants generated from the bus parks infiltrate to adjacent indoor waiting areas, the pollutants could be dispersed and deteriorate indoor air quality. Therefore, the air sample collection was proceeded for both bus park areas and indoor waiting areas to check whether the air pollutants from bus parks would migrate to the indoor spaces. Furthermore, it was expected that the architectural structure of bus parks could affect the migration of air pollutants depending on how much the area exposed to outdoor environment. Then, all bus interchanges were categorized as fully sheltered or open/partially enclosed and further analyzed with air samples by comparing each other.

Secondly, this research was to estimate how badly the air pollutants could affect human health by proceeding risk assessment. The risk assessment was based on the U.S. EPA's guidance document to estimate the carcinogenicity and reproductive toxicity risk of VOCs and PAHs. The daily dose for each VOC was calculated individually and compared to the OEHHHA's safe harbor levels, No Significant Risk Level (NSRL) for carcinogenic risk and Maximum Allowable Dose Level (MADL) for non-carcinogenic risk. The PAHs risk assessment based on the safe harbor levels was not conducted, since the OEHHHA only provided the level of benzo[a]pyrene. As an alternative, the Incremental Lifetime Cancer Risk (ILCR) for the exposure to PAHs was estimated to assess carcinogenic risk with Toxic Equivalent Factor, which takes the toxicities of the other PAHs into account through the conversion to benzo[a]pyrene. Additionally, the reference concentration, a maximum acceptable concentration of an air pollutant from EPA's Integrated Risk Information System, and an acceptable limit of TVOCs from indoor air quality guideline of Singapore were collected and compared to the concentrations of real samples from bus interchanges.

4. CHAPTER 2 – LITERATURE REVIEW

4.1 Volatile Organic Compounds

4.1.1 Definition

Volatile Organic Compounds (VOCs) are generally defined as carbon containing compounds which can evaporate at room temperature. Some definitions contain chemical characteristics, temperature and pressure, because VOC is classified based on its volatility. For example, the definition of VOC from EU Directive 2004/42/CE is any organic compound having an initial boiling point less than or equal to 250°C measured at a standard atmospheric pressure of 101.3 kPa (EU, 2004). Environmental Protection Agency (EPA) defines the VOCs as carbon containing compounds which are able to cause atmospheric photochemical reactions excepting few chemical groups such as metallic carbides and ammonium carbonate (EPA, 2017). Moreover, World Health Organization (WHO) subdivides indoor VOCs into three groups based on the boiling points of compounds, Very volatile organic compounds (< 0°C to 50-100°C), Volatile organic compounds (50-100°C to 240-260°C) and Semi-volatile organic compounds (240-260°C to 380-400°C) (WHO, 1989).

4.1.2 Properties of VOCs

As the definition of VOC is somewhat extensive, VOCs encompass diverse classes of organic compounds such as alkanes, aromatics, aldehydes, ketones, alcohols and ethers, and the physicochemical properties of these compounds are all different (Fang, P. et al., 2016; Hodgson, M. et al., 1994). However, a common characteristic of the VOCs is that the chemical group could be readily evaporated at room temperature. Literally, VOCs are organic compounds, so their structures are mostly based on hydrocarbons comprised of only hydrogen and carbon atoms, such as benzene, toluene and hexane. As the hydrocarbons are electrically neutral, most VOCs are neutral and non-polar, and these chemical properties make them volatile. However, some VOCs containing polar functional groups, such as alcohols and ketones, are not evaporated well like hydrocarbons. Then, only few compounds of them are considered as VOCs, which molecular weights are low. Like as WHO divides VOCs into three groups based on boiling point, intermolecular force is sometimes used to define and characterize VOCs (WHO, 1989).

However, the definition could not completely represent all VOCs in environment because the concept of boiling point is only applicable to the phases between liquid and gas. The

VOCs could exist as airborne particles, liquid and sometimes solid. For example, the VOCs can be emitted from smoke, liquid adhesive and solid repellent (Wallace L.A., 1993). Then, the concept of vapor pressure, defined as pressure of a substance in equilibrium with its pure condensed phase (liquid and solid), is commonly used to characterize VOCs.

Table 1 summarizes the properties of selected VOCs for this research study. The table contains molecular weight, structure, boiling point and vapor pressure of each compound.

Table 1. Properties of selective 12 VOCs

Compounds	M.W.(g/mol)	Structure	Boiling Point(°C)	Vapor Pressure (mmHg at 25°C)
Benzene	78.11		80.1	94.80
Toluene	92.14		110.6	28.40
Tetrachloroethylene	165.83		121.1	18.50
Ethylbenzene	106.17		136.0	9.60
m-Xylene	106.16		139.0	8.29
p-Xylene	106.16		138.4	8.84
o-Xylene	106.16		144.0	6.65
Styrene	104.15		145.0	6.40
1,4-dichlorobenzene	147.01		174.0	1.74

1,2-dichlorobenzene	147.01		180.0	1.36
1,3-dichlorobenzene	147.01		173.0	2.15
Naphthalene	128.17		218.0	0.08

4.1.3 Sources of VOCs

Volatile organic compounds are present everywhere because the sources of VOCs are diverse and widespread. Many researchers classify the sources of VOCs into two groups, natural or anthropogenic sources. As for the natural sources, microorganisms living in the soil could emit various VOCs by decomposing soil organic matters. The VOCs also could be generated by evaporation from plants and their debris (Peñuelas, J. et al., 2014). Furthermore, wildfire could be a source of VOCs, if organic matters in nature are not completely combusted (Dreessen, J. et al., 2016).

Anthropogenic VOCs sources are more diverse and ubiquitous, since they have been synthesized and used for commercial purpose. The major sources are obviously from the evaporation of petroleum such as oil drilling, refining, transferring and storage (Piccot, S.D. et al., 1992). Moreover, VOCs are commonly generated from the combustion of fossil fuel, as reported in many studies at traffic related sites, such as bus stations (Lin, C. et al., 2020; Kim, Y.M. et al., 2001), in buses (Parra, M. et al., 2008; Lau, W.-L., & Chan, L.-Y., 2003), on roads with high traffic (Yu, C. H. et al., 2014; Ho, K., & Lee, S., 2002), and indoor car parks (Castro, B. P. et al., 2015). However, VOCs could be readily found in people's daily life as inconspicuous form. Many consumer products, paints, adhesives, building materials, furnishings and cleaning products contain diverse types of VOCs, and human activities, such as smoking, heating and cooking activities could emit different types of VOCs (Gammage, R.B. et al., 1988; Clobes, A.L. et al., 1992; Jones, A.P., 1999; WHO, 2010; Kim, Y.M. et al., 2001; Gennaro, G.D. et al., 2016). Therefore, many research papers reported VOCs were detected in diverse indoor microenvironments, such as offices (Zuraimi, M. et al., 2006; Daisey, J. et al., 1994), residents (Lee, J. Y. et al., 2018; Yamaguchi, T. et al., 2006) and shopping malls (Tang, J. et al., 2005).

4.1.4 The fate of VOCs

In reality, VOCs could exist in various forms depending on their surroundings. Once the VOCs are evaporated, they can be physically removed from the atmosphere via dry and wet deposition, and a number of photochemical reaction transform the compounds into mostly polar and water-soluble compounds of low volatility.

First of all, gaseous VOCs could be adsorbed on solid surfaces. Basically, the attraction force between adsorbent and adsorbate is a key factor to determine adsorption capacity. Polar VOCs tend to adhere on the adsorbent which has electron-rich surface, such as charcoal. In contrast, non-polar VOCs like benzene stick to non-polar adsorbent because π - π interaction dominates the adsorption capacity (Yang, C. et al., 2019; Franz, M. et al., 2000). This attribute makes VOCs be attached to particular solids in accordance with their physicochemical characteristics. In indoor environment, VOCs could be on the ceiling and building materials (Huang, H. et al., 2006; Liu, X. et al., 2011). Besides, the outdoor environment also has natural solid adsorbents for VOCs, such as soil. A mineral particle, in itself, does not have good sorption capacity, but soil mixture containing water and soil organic matter is capable of attracting VOCs in the soil environment (Goss, K.-U. et al., 1996; Shih, Y.-H. et al., 2005; Petersen, L. W. et al., 1995). Furthermore, many adsorption research projects have been conducted and still studied to know the kinetic mechanism of adsorption and, as a result, various types of adsorbents have invented such as activated carbons, zeolites, resins and metal-organic frameworks (Yang, C. et al., 2019).

Secondly, gaseous VOCs could be dissolved in liquid. This process should be differentiated from condensation. Condensation is a process that vapor molecules are precipitated, so the liquid product would be pure chemical. However, the dissolution would be a different process that vapor molecules are dissolved into a liquid solvent. Consequently, the solvent contains the molecules as a mixture. For this reason, the former process applies the concept of vapor pressure and the latter uses equilibrium partitioning. In brief, partitioning is a phenomenon that specific molecules make a choice into which they are blended among different types of fluid at dynamic equilibrium. The fluid could be air, polar solvent and apolar solvent. As for gaseous VOCs, the concepts of air-polar solvent partitioning and air-apolar solvent partitioning are applicable. For example, if gaseous VOCs exist nearby the aquatic environment, air-water partitioning coefficient (K_{aw}) could be measured by the VOCs concentrations in both air (C_a) and water (C_w). The following equation (4-1) represents how to get K_{aw} from two VOCs concentrations.

$$K_{aw} = \frac{C_a}{C_w} \quad (4-1)$$

Another instance is in which gaseous VOCs encounters organic solvents. The representative of organic solvent is n-octanol ($C_8H_{18}O$), because its chemical structure is so similar to phospholipid in human body that it can be utilized as an indicator of human health risk especially in dermal absorption. Thus, air-octanol partitioning coefficient (K_{oa}) is widely used and it could be calculated by the VOCs concentrations from both air (C_a) and octanol (C_o).

$$K_{oa} = \frac{C_o}{C_a} \quad (4-2)$$

In addition to this, K_{oa} can be derived from octanol-water partitioning coefficient (K_{ow}) and K_{aw} as equation (4-3).

$$K_{oa} = \frac{K_{ow}}{K_{aw}} = \frac{C_o/C_w}{C_a/C_w} = \frac{C_o}{C_a} \quad (4-3)$$

Partitioning coefficients for selective VOCs are extracted from a textbook, environmental organic chemistry, and arranged in Table 2 (Schwarzenbach René P. et al., 2002).

Table 2. Partitioning Coefficient of selective 12 VOCs at 25°C

Compounds	M.W.(g/mol)	-log K_{aw}	log K_{ow}
		Calculated (experimental)	
Benzene	78.11	0.65(0.65 exp.)	2.17
Toluene	92.14	0.60(0.62 exp.)	2.69
Tetrachloroethylene	165.83	-0.08(0.12 exp.)	2.88
Ethylbenzene	106.17	0.50(0.47 exp.)	3.20
m-Xylene	106.16	0.53(0.52 exp.)	3.30
p-Xylene	106.16	0.55(0.54 exp.)	3.27
o-Xylene	106.16	0.69(0.68 exp.)	3.16
Styrene	104.15	0.93(0.92 exp.)	2.95
1,4-dichlorobenzene	147.01	1.04(1.10 exp.)	3.45
1,2-diclorobenzene	147.01	1.04(1.00 exp.)	3.40
1,3-dichlorobenzene	147.01	0.86(0.82 exp.)	3.47
Naphthalene	128.17	1.74(1.72.exp.)	3.33

A review of the characteristics of VOCs in the aquatic ecosystem would be helpful in understanding those concepts mentioned above. It concludes that volatilization based on Henry's Law is obviously a dominant process in the streams of VOCs, but other physical processes are also able to contribute to the fate of VOCs (Rathbun, R.E., 2000).

Furthermore, the concept of aerosol, defined as a suspension of fine solid particles or liquid droplets in the atmosphere, is sometimes applied to explain the fate of VOCs. Since gaseous VOCs could exist not only as molecules in air, but also as solutes or adsorbates in aerosol (e.g. VOCs in fogs or fine dust), more detailed studies on partitioning mechanism have been conducted to suggest more accurate human health risk assessment. However, the aerosol partitioning studies are still limited, and the present predictive models of the partitioning coefficient are mostly developed for nonpolar VOCs. Since VOCs have their own physicochemical attributes and they could be involved in varied types of aerosols, this diversity makes the research more complicated and challenging. The uncertainty and complexity have discouraged researchers to develop normalized formulas based on the characteristics of VOCs. However, a better understanding of the VOCs partition mechanism and a technological improvement in motivating real condition would make it possible to approximately estimate the fate and transport of VOCs (Rao, G. et al., 2018).

Lastly, gaseous VOCs could be transformed by chemical processes of photolysis. Their transformation processes can be provoked by radicals in troposphere, such as hydroxyl radicals and nitrate radicals, and also initiated by ozone, halogen atoms or photons (Atkinson, R. et al., 2003; Atkinson, R., 2000; Hester, R. E. et al., 1995). After the initial reaction of VOCs, subsequent reactions generally form alkyl or substituted alkyl radicals. Figure 1 shows the generalized decomposition processes of VOCs in troposphere.

Figure 1. The photolysis pathway of gaseous VOCs in troposphere (Atkinson, R. et al., 2003)

The products and the intermediates such as organic peroxy ($\text{RO}_2^\bullet$) and alkoxy ($\text{RO}^\bullet$) radicals are varied depending on the chemical attribute of VOCs, and the physicochemical characteristics of VOCs are mostly changed to polar and water-soluble. Since this study is not about photolysis, the more detailed processes are not dealt with here.

4.1.5 The Exposure Routes of VOCs

Basically, many environmental agencies including EPA classify the exposure routes into three, inhalation, ingestion and dermal absorption. Regarding air pollutants, the major pathway for human exposure is inhalation (Vallero, D., 2014).

By breathing contaminated air, the pollutants could spread through the respiratory system, and much smaller pollutants could be transferred to the blood circulation system across alveolus (Damato, G. et al., 2002). As for ingestion, air pollutants might be difficult to directly get into the gastrointestinal tract, because the esophagus, a linkage between mouth and stomach, is only opened, while a person intakes some foods. Therefore, they could be ingested, only if people consume foods or drinks that contain air pollutants. However, those routes are regarded as negligible or labeled as other categories such as food ingestion (ATSDR, 2005). This is because it takes considerable time for the air pollutants to change media from air to foods or drinks. Then, it might not be plausible to include the ingestion as a human exposure pathway of air pollution. Furthermore, the pathways are not direct ways to humans, so they are commonly excluded. Lastly, some types of organic air pollutants could permeate human skin through the sorption mechanism. For example, the phthalates exposure through skin absorption is comparable to via inhalation due to their lipophilicity (Gong, M. et al., 2013). Although the dermal absorption of gaseous VOCs is not that strong like phthalates, it should be regarded as an exposure pathway considering potential human health risk. Once people are exposed to VOCs for a long period, the pollutants would accumulate in skin-surface lipids, and it might cause allergy symptoms (Little, J. C. et al., 2012). For these reasons, two types of pathways, inhalation and dermal contact, are selected for risk assessment and presented in the latter part.

4.1.6 Human Health Risks of VOCs

Historically, VOCs has been perceived as a precursor of photochemical smog with NO_x and O_3 . Since early 1900's, Los Angeles had been through occasional photochemical smog until 1970's at which the Clean Air Act enforced automobile manufacturers to reduce emissions. Due to the smog problem, 27 percent of local people had to live with severely impaired lungs and almost 10,000 people died every year (Dundon, R., 2018).

Besides, VOCs still cause some environmental issues nowadays, even though the smog problem has managed well through strict air pollution regulations. Excluding accidental chemical spill, Sick Building Syndrome (SBS) could be a major topic of VOC exposure in indoor atmosphere. Literally, SBS is a syndrome that people feel unpleasant and sick while they stay indoor environments such as building and home. The cause of SBS was identified as the emission of VOCs used in building materials, furniture and other indoor sources. Some of the VOCs, such as formaldehyde and benzene, cause severe health problems in humans. Formaldehyde, extensively used as adhesives for wood and wood composites, would irritate eyes and upper respiratory system due to its high solubility in water (Salthammer, T. et al., 2010). Furthermore, if people are exposed to formaldehyde for the long-term, it might cause cancer (Kerns, W. D. et al., 1983; Swenberg, J. A. et al., 1980). Benzene, which is also known as a carcinogen, is mostly generated from the exhaust gas and cigarette smoke. It could reduce the number of blood corpuscle in the human body, and then the risk of anemia, leukopenia and thrombocytopenia would be raised consequently. Moreover, benzene has neurotoxicity, so it may cause headache, dizziness, drowsiness and even loss of consciousness (EPA, 2002). In general, VOCs could bring typical symptoms to people, such as irritation of eyes, nose and throat, headaches and dizziness in indoor environment. However, when people are exposed to VOCs in the long term, the severities of health effects would be diversely intensified relying on the toxicity of individual VOCs.

EPA provides the toxicological information about chemical compounds including VOCs in detail via Integrated Risk Information System (IRIS), and Agency for Toxic Substances and Disease Registry (ATSDR), a federal public health agency affiliated to United States Department of Health and Human Services (HSS), gives toxicological profiles of hazardous substances. Referring to the two kinds of databases, relevant VOCs in this research and their known symptoms are listed and summarized in Table 3.

The noncarcinogenic health effect part was organized based on ATSDR, and the carcinogenicity part was referred to IRIS. As for noncarcinogenic health effects, it was basically classified as two, acute effects and chronic effects. Both categories were established based on the real case of chemical exposure and animal experiments. Moreover, when a compound caused especial symptoms (e.g. carcinogen), additional subdivisions were applied. For the carcinogenicity part, the criteria for the classification of carcinogens were inconsistent because some chemicals were reviewed in former days and the others were recently examined. Since the chemical data of carcinogenicity highly depends on previous incidents of chemical exposure to humans, it would not be frequently

updated. Furthermore, the toxicity of chemical could be changed when more reliable data come out from other experiments. Therefore, it might not be possible to sort and assess all chemical compounds with a guideline. For this reason, the carcinogenicity part was extracted from several EPA guidelines and presented in the latest guideline to avoid confusion.

Table 3. Noncarcinogenic and Carcinogenic Health Effect of VOCs

Compounds	CASRN	Noncarcinogenic Health Effect	Carcinogenicity
Benzene	71-43-2	<p>Acute Effects: Drowsiness, dizziness, headaches, unconsciousness, vomiting, convulsion, irritation of the upper respiratory tract, eyes and skin.</p> <p>Chronic Effects: Benzene affects bone marrow, so the number of blood cell could be diminished (oligocythemia, leukopenia and thrombocytopenia). Structural and numerical chromosomal aberrations.</p> <p>Cancer Risk: Leukemia.</p>	Known/likely human carcinogen (EPA, 1996)
Toluene	108-88-3	<p>Acute Effects: Central nervous system dysfunction, necrosis, fatigue, sleepiness, headaches, nausea, cardiac arrhythmia, constriction, necrosis of myocardial fibers, swollen liver, congestion, tubular kidney necrosis.</p> <p>Chronic Effects: Central nervous system dysfunction, drowsiness, ataxia, tremors, cerebral atrophy, nystagmus, impaired speech, hearing and vision, irritation of the upper respiratory tract and eyes, sore throat, dizziness, headache, insomnia.</p> <p>Developmental Effects: Temporary renal tubular acidosis.</p>	Inadequate information to assess the carcinogenic potential (EPA, 2005)

Tetrachloroethylene	127-18-4	<p>Acute Effects: Irritation of the upper respiratory tract and eyes, kidney dysfunction, reversible mood, behavioural changes, impairment of coordination, dizziness, headache, sleepiness, unconsciousness, liver damage.</p> <p>Chronic Effects: Headaches, impairments in cognitive and motor neurobehavioral functioning, color vision decrements, liver damage, kidney effects, immune and hematologic effects.</p>	Likely to be carcinogenic in humans (EPA, 2005)
Ethylbenzene	100-41-4	Acute Effects: Irritation of the upper respiratory tract and eyes, dizziness, chest constriction, liver damage and kidney damage.	Group D - Not classifiable as to human carcinogenicity (EPA, 1986)
Xylenes	1330-20-7	<p>Acute Effects: Dyspnea and irritation of the nose and throat, nausea, vomiting, gastric discomfort, short-term memory loss, alterations in equilibrium and body balance, skin irritation.</p> <p>Chronic Effects: Headache, dizziness, fatigue, tremors, incoordination, anxiety, short-term memory loss, heart palpitation, severe chest pain, kidney damage.</p>	Data are inadequate for an assessment of human carcinogenic potential (EPA, 1999)
Styrene	100-42-5	Acute Effects: Irritation to the eyes and respiratory tract, nausea, decreased digestive function	Not assessed under the IRIS.

		Chronic Effects: Headache, fatigue, depression, central nervous system dysfunction, hearing loss, peripheral neuropathy, minor effects on kidney and blood.	
1,4-dichlorobenzene	106-46-7	Acute Effects: Irritation to the eyes and respiratory tract, coughing, difficulty breathing, headache, dizziness, upset stomach liver damage, burning sensation on skin. Chronic Effects: Skin blotches, anemia, liver tumor.	Not assessed under the IRIS.
1,2-dichlorobenzene	95-50-1	Acute Effects: Irritation of the upper respiratory tract and eyes, liver damage, headaches and dizziness.	Group D - Not classifiable as to human carcinogenicity (EPA, 1986)
1,3-dichlorobenzene	541-73-1	Acute Effects: Irritation of the upper respiratory tract and eyes, liver damage, endocrine system damage, headaches and dizziness.	Group D - Not classifiable as to human carcinogenicity (EPA, 1986)
Naphthalene	91-20-3	Acute Effects: Hemolytic anemia, liver damage, neurological damage, headache, nausea, vomiting, diarrhea, malaise, confusion, anemia, jaundice, convulsions, coma and cataracts. Chronic Effects: Cataracts, retinal haemorrhage, lung inflammation and nasal inflammation. Developmental Effects: Maternal hemolytic anemia.	Carcinogenic potential cannot be determined (EPA, 1996)

4.2 Polycyclic Aromatic Hydrocarbons

4.2.1 Definition

Polycyclic Aromatic Hydrocarbons (PAHs) are organic compounds containing multiple aromatic rings (ATSDR,1995). PAHs are non-polar and hydrophobic compounds, since they are only composed of carbons and hydrogens. PAHs are mostly generated during incomplete burning of fossil fuel, wood, wastes and other organic compounds, such as tobacco (Shen, H. et al., 2013). The number of PAHs is over a hundred, and they are usually generated as complex mixtures from various sources.

Among the numerous PAHs, U.S. EPA highlighted 16 PAHs and listed them in Priority Pollutants by following EPA Consent Decree in 1976. Acenaphthene, acenaphthylene, anthracene, benz[a]anthracene, benzo[a]pyrene, benzo[b]fluoranthene, benzo[g,h,i]perylene, benzo[k]fluoranthene, chrysene, dibenzo[a,h]anthracene, fluoranthene, fluorene, Indeno[1,2,3-cd]pyrene, naphthalene, phenanthrene and pyrene are the PAHs in the list. The reasons why those chemical compounds were selected were because (1) they exhibit more harmful effects than others; (2) more information was available; (3) they could be exposed to people more frequently; and (4) the analytic instrument was limited at that time (ATSDR,1995; Lawrence H. Keith, 2015). As the detrimental compounds were listed as 16 PAHs, the particular PAHs have been concerned and studied well to minimize human health risks.

However, many researchers have argued that other organic compounds, not within the list but in the group of Polycyclic Aromatic Compounds (PACs), should be paid more attention and appropriately managed due to their toxicities. They stressed that larger and highly relevant PAHs, alkylated PACs, and heterocyclic compounds such as nitrogen and oxygen-containing rings should also be included in the list (Andersson, J. T., & Achten, C., 2015).

4.2.2 Properties of PAHs

As benzene is known for a chemically stable compound, PAHs, comprised of multiple aromatic rings, are also durable, because the carbons of PAHs are saturated with hydrogen atoms. Since all carbons of PAHs are connected to hydrogen atoms and other carbon atoms, PAHs show electrical neutrality. Due to the stable characteristic, it requires additional force to transform PAHs into other compounds by addition or substitution (McMurry, J., 2012).

Moreover, the chemical properties of PAHs could be diverse relying on their arrangement of benzene rings. The arrangement is categorized as three, linear, angular and cluster. For example, among 4-rings PAHs, tetracene has a linear structure, benz[a]anthracene is an angular and pyrene is a cluster form. The linear arrangement is the weakest type of them, as if linear alkane is more fragile than branched alkane. Clar's aromatic π -sextet rule might give a plausible explanation to which structure would bring chemical stability. Briefly, the Clar's rule was originated to anticipate the reactivity of PAHs by looking at their chemical structure. Among the structural isomers of PAHs, if a structure could form more benzene rings through localization of π -electron pairs, its reactivity is comparatively lower, because the fully localized π -electron pairs make the PAH structure more stable. Since a linear structure of PAHs, such as tetracene, could only have a single localized benzene ring on the edge of the structure, the reactivity is always higher than other nonlinear isomers (Solà, M., 2013; Keyte, I. J. et al., 2013).

The physical properties of PAHs are similar to common characteristics of VOCs. Although the characteristics of PAHs depend on their molecular weight and structure, PAHs have typical attributes, such as lipophilicity. However, PAHs have relatively lower vapor pressure and higher melting and boiling point than VOCs' owing to their larger molecules (Abdel-Shafy, H. I., & Mansour, M. S., 2016).

Table 4 summarizes the properties of selected PAHs for this research study. Naphthalene is excluded from the 16 PAHs because it was presorted as VOCs in this research. The table contains molecular formula, weight, structure, the number of benzene rings in the molecular, boiling point and vapor pressure.

Table 4. Properties of selective 15 PAHs

Compounds	Formula	M.W. (g/mol)	Benz ene Rings	Structure	Boiling Point (°C)	Vapor Pressure (mmHg at 25°C)
Acenaphthylene	C ₁₂ H ₈	152.19	2		280.0	4.80×10 ⁻³
Acenaphthene	C ₁₂ H ₁₀	154.21	2		279.0	2.20×10 ⁻³
Fluorene	C ₁₃ H ₁₀	166.22	2		295.0	6.00×10 ⁻⁴

Phenanthrene	C ₁₄ H ₁₀	178.23	3		340.0	1.21×10 ⁻⁴
Anthracene	C ₁₄ H ₁₀	178.23	3		339.9	6.56×10 ⁻⁶
Fluoranthene	C ₁₆ H ₁₀	202.25	3		384.0	9.22×10 ⁻⁶
Pyrene	C ₁₆ H ₁₀	202.25	4		404.0	4.50×10 ⁻⁶
Benz[a]anthracene	C ₁₈ H ₁₂	228.29	4		437.6	2.10×10 ⁻⁷
Chrysene	C ₁₈ H ₁₂	228.29	4		448.0	6.23×10 ⁻⁹
Benzo[b]fluoranthene	C ₂₀ H ₁₂	252.32	4		481.0	5.00×10 ^{-7*} (at 20°C)
Benzo[k]fluoranthene	C ₂₀ H ₁₂	252.32	4		480.0	9.65×10 ⁻¹⁰
Benzo[a]pyrene	C ₂₀ H ₁₂	252.32	5		495.0	5.49×10 ⁻⁹
Dibenz[a,h]anthracene	C ₂₂ H ₁₄	278.35	5		524.0	9.55×10 ⁻¹⁰
Indeno[1,2,3-c,d]pyrene	C ₂₂ H ₁₂	276.34	5		536.0	1.25×10 ⁻¹⁰
Benzo[g,h,i]perylene	C ₂₂ H ₁₂	276.34	5		550.0	1.00×10 ⁻¹⁰

4.2.3 Sources of PAHs

PAHs are ubiquitous in the environment and can be generated from both natural and anthropogenic sources. According to a global estimation of PAH deposition into the atmospheric environment, 520 Giga grams of PAHs are discharged per year. Among the various sources of PAHs, the usage of biofuel takes the largest proportion, around 56.7%. Wildfire (17.0%), consumer product use (6.9%), traffic oil combustion (4.8%), domestic coal combustion (3.7%), coke production (3.6%), petroleum refining (2.4%), waste incineration (1.9%), aluminum electrolysis (1.4%), and others (1.5%) follow in

descending order of magnitude. The estimation also illustrates that the sources of PAHs are utterly different depending on countries. Since industrialized countries generally use fossil fuel for energy, the dependency of biomass combustion is significantly lower than the dependency of developing countries (Zhang, Y., & Tao, S., 2009).

As for the indoor atmospheric environment, the sources of PAHs are diverse in different conditions. The pollutants can be flown into indoor environments from outdoor sources and also be raised from indoor sources. Firstly, vehicle exhaust, known for the major outdoor sources of PAHs, could pollute indoor environments through airflow. Many of PAHs measurement studies in diverse indoor microenvironments, such as school (Zhu, J. et al., 2019) and childcare center (Oliveira, M. et al., 2014), reported the PAHs originated from vehicle exhaust could deteriorate indoor air quality if the places were surrounded by busy roads. Secondly, tobacco smoke could be a dominant source of PAHs in the indoor environment when periodical smoking activities occur (Chuang, J. C. et al., 1991). A research study measured the concentration of 18 PAHs (16 PAHs from EPA list, dibenzo[a,l]pyrene and benzo[j]fluoranthene) at residential spaces which smokers or non-smokers were living in. In the residences for smokers, higher concentrations of each PAH were detected and also the total concentration of PAHs was 95% higher than for non-smokers (Castro, D. et al., 2011). Thirdly, cooking and heating activities could be major sources of PAHs in the indoor atmosphere. In the developing countries, the use of biofuels, such as woods and agriculture residues, is prevailing and large amounts of PAHs are generated due to the incomplete combustion of those biofuels. Burning biofuels in indoor spaces would raise PAHs concentration levels and it may cause human health issues, when the ventilation system is not functioning well (WHO, 2010). Furthermore, cooking, in itself, would generate PAHs, when foods are high in fat or a high quantity of cooking oil is used for dishes (Purcaro, G. et al., 2006). Lastly, the usage of air freshener and mothballs could also be contributors to raise PAHs concentration levels in the indoor atmosphere (Winkle, M. R. V., & Scheff, P. A., 2001).

4.2.4 The Fate of PAHs

As PAHs represent Semi-Volatile Organic Compounds (SVOCs), the physical properties of PAHs are closely akin to common VOCs'. Basically, PAHs are apolar compounds, so they are not readily soluble in water, but miscible with common organic solvents. However, PAHs are low in volatility, so they exist as two forms in the atmosphere environment, vapor phase and adsorbed phase in airborne particulate matter (Pratt, G. et al., 2018). Low-weight PAHs, which have two or three benzene rings, tend to be remained

as gas due to their relatively high vapor pressure. Naphthalene, acenaphthylene and phenanthrene are the examples within EPA's 16 PAHs. In contrast, high-weight PAHs, which have five or more benzene rings, incline to exist as particle-bound owing to their low volatilities and their good adsorption capacities. PAHs with four rings are capricious depending on their ambient environment. Although the attributes come from chemical structures of PAHs, their forms in the atmosphere are not consistent, since the vapor pressures of PAHs could be changed depending on ambient temperature. High ambient temperature enables them to be vaporized, resulting in an increase in the gaseous phase. For this characteristic, the atmospheric PAHs exhibit seasonal concentration differences between summer and winter (Subramanyam, V. et al., 1994; Lai, I-C. et al., 2011).

In the atmosphere, more than 95% of PAHs exist as vapor phase. This is because the light PAHs compounds, such as naphthalene and phenanthrene, are dominant and they tend to be vaporized in the ambient temperature (Park, J-S. et al., 2001). However, particle-bound PAHs could largely contribute to total PAHs in the atmosphere, if the sampling area is close to heavy PAHs sources, such as densely populated areas. For example, numerous studies have been conducted for PAHs sampling in both rural areas and urban areas in order to characterize how much exhaust gas would aggravate air quality regarding PAHs. Generally, the concentrations of particulate bound PAHs in urban areas were higher than rural areas and the dominant compounds were 4-rings and 5-rings PAHs (Verma, P. K. et al., 2017; Slezakova, K. et al., 2011). In short, the concentration and composition of atmospheric PAHs could be varied relying on their circumstances. After the PAHs are generated and blended into the atmosphere environment, they would be decomposed and transferred to other media.

As for the decomposition of PAHs, atmospheric PAHs could be transformed into other compounds caused by photochemical reactions. High solar energy could make the stable compounds activated, and those excited compounds cause consecutive chemical reactions. In the atmosphere, hydroxyl radicals, ozone and nitrate radicals are formed through photolysis, and they subsequently react with PAHs, both gas form and particulate bound form. As a result, those PAHs are deformed, and some of them become more toxic such as nitro-PAHs, which could be potent mutagen and carcinogen (Arey, J., & Atkinson, R., 2003).

Regarding the transportation of PAHs, atmospheric PAHs are also consistently transferred to soil and water environment through deposition. Once they are captured by rain or snow, they are subsequently blended into other media. This process is called wet deposition. The

other case is dry deposition, a transport triggered by gravitational or wind force. Both depositions are the ways that air pollutants could sprawl and spread, so these pathways have been studied and monitored well to control hazardous air pollutants (Canter, L. W. et al., 2000).

When PAHs are introduced to the soil environment, most of them become immobile due to their non-polar attribute. They are attached to the solid particles especially to SOM rather than remained as a dissolved form in pore water. Relatively high octanol-water partitioning coefficients of PAHs would imply how those compounds would act in the soil environment. The coefficients would be introduced later in this chapter. Furthermore, the PAHs remained in the water can descend through pore spaces in soil and reach the area where microorganisms are plentiful. The pollutants could be removed by the microorganisms through biotransformation or biodegradation. For example, arbuscular mycorrhizal fungi, which inhabit the plant's roots and have a symbiotic relationship with plants, could get rid of PAHs from soil (Binet, P. et al., 2000).

As for the water environment, atmospheric PAHs can be introduced to the environment along the water cycle. Once the PAHs are captured by rain or snow, their fate would be determined by each stage of water circulation. When the rainwater washes off the contaminated areas with PAHs, such as roads with high traffic, the concentration levels of PAHs could be increased due to the influx PAHs from the sites (Motelay-Massei, A. et al., 2006). Specifically, the land usage for industrial purposes has a great effect on the increment of PAH concentration in surrounding environments, especially 4-rings and 5-rings PAHs, whereas rural areas or residential areas have a relatively low impact on the PAHs increment and their major compounds are 2-rings and 3-rings PAHs (Menzie, C. A. et al., 2002). Otherwise, when the PAHs are introduced to the uncontaminated water environment, which is more common, the PAHs would be diluted and decomposed. Once the PAHs precipitate, the compounds incline to attach on organic compounds due to the hydrophobic nature. For example, in the aquatic environment, organic compounds termed as Total Organic Compounds (TOCs) enable the PAHs to move readily in water, and some of PAHs could reach river-bed or sea-bed (Suess, M. J., 1976). High correlations between the three components of water ecosystem, PAHs concentration in water, TOC, and the concentration in soil, were found and it would explain that PAHs are able to wander in the water system, although they are not miscible with water (Castro-Jiménez, J., 2012). Moreover, aqueous PAHs could be decomposed in which sunlight shine down, as atmospheric PAHs could be degraded by photochemical reactions. Furthermore,

bioaccumulation and biodegradation are also possible pathways of the removal of aqueous PAHs (Suess, M. J., 1976; Zeng, Q. et al., 2018).

Partitioning coefficients would be applicable to anticipate the fate of PAHs in a complex environment. Partitioning coefficients for selective PAHs are extracted from a textbook, environmental organic chemistry, and arranged in Table 5 (Schwarzenbach René P. et al., 2002). Due to the lack of information for some compounds, U.S. EPA's EPI Suite program was also applied to estimate the partitioning coefficients and marked with an asterisk. The information would be applicable to understanding the physical attribute of PAHs.

Table 5. Partitioning Coefficient of selective 15 PAHs at 25°C

Compounds	M.W.(g/mol)	-logK_{aw} Calculated (experimental)	logK_{ow}
Acenaphthylene	152.19	1.85	4.00
Acenaphthene	154.21	2.29(2.28 exp.)	4.20
Fluorene	166.22	2.47(2.39 exp.)	4.32
Phenanthrene	178.23	2.85(2.93 exp.)	4.57
Anthracene	178.23	2.80(2.77 exp.)	4.68
Fluoranthene	202.25	3.34(3.35 exp.)	5.23
Pyrene	202.25	3.32(3.36 exp.)	5.13
Benz[a]anthracene	228.29	3.68	5.91
Chrysene	228.29	4.56	5.81
Benzo[b]fluoranthene*	252.32	4.57	5.78
Benzo[k]fluoranthene*	252.32	4.62	6.11
Benzo[a]pyrene	252.32	4.79(4.86 exp.)	6.13
Dibenz[a,h]anthracene*	278.35	5.24	6.54
Indeno[1,2,3-c,d]pyrene*	276.34	4.85	6.70
Benzo[g,h,i]perylene*	276.34	4.87	6.63

* -log K_{aw} and log K_{ow} are estimated from U.S. EPA's EPI Suite program.

4.2.5 The Exposure Routes of PAHs

As mentioned in the fate of PAHs, PAHs are prevalent in diverse media, air, water, soil and others. Through the media, PAHs get into the human body and they could cause human health problems, such as cancer. Environment agencies including EPA classify the exposure routes of PAHs into three, inhalation, ingestion and dermal absorption. Since the sources of PAHs are prevalent in the ordinary life of people, major sources of PAHs and their exposure routes utterly depend on people's patterns of life (WHO, 2010). For example, cigarette smoke could be a major source of exposure for heavy smokers, and then the major exposure route is inhalation (Castro, D. et al., 2011). In addition, those people who enjoy fried foods and charred meats would intake PAHs, and their major exposure route would be ingestion (David H. Phillips, 1999). Furthermore, dermal exposure, which is the least common pathway, is also considered important for workers in particular industries, such as asphalt paving (Väänänen V. et al., 2005).

4.2.6 Human Health Risks of PAHs

PAHs are very detrimental to human health in that some of them could cause cancer at worst. When people are exposed to the chemical compounds, they might suffer from minor symptoms to dysfunction of organs. The severity depends on the length of exposure, exposure route, the toxicity of PAHs and the physical condition of casualties. However, it is not possible to know which PAH would bring human health effects exactly, because PAHs are generated as a mixture from incomplete combustion (Abdel-Shafy, H. I., & Mansour, M. S., 2016).

Historically, before PAHs were identified as carcinogens, there were some records that incomplete combustion materials such as soot could cause cancer and the materials were from specific workplaces. In 1775, a diagnosis record from St. Bartholomew's Hospital in London indicated that chimney sweeper had scrotal cancer from chronic exposure to soot. Moreover, the 1875 report written by a German surgeon, Richard von Volkmann, also notified that soot, coal tar and pitch from the coal industry could cause skin cancer. However, in the middle of the 20th century, Cook, Hewett and Hieger came to the conclusion that particular hydrocarbons could cause cancer. They could separate hydrocarbons from incomplete combustion materials, and they specified that perylene, benzo[a]pyrene and benzo[e]pyrene were carcinogens (Anthony, D., 1985).

Although numerous studies on PAHs have investigated and identified, estimating each PAH toxicity for humans is still challenging and might not be possible. This is because PAHs are generated as a mixture from sources, and there are no sufficient records of

chemical accidents about individual PAH. Due to the difficulties in characterizing human health effects for each PAH, many health-related organizations only provide limited information about PAHs. Human health risk information about PAHs is collected from EPA's fact sheet for hazardous air pollutants (EPA, 2016) and WHO guidelines for indoor air quality (WHO, 2010) and summarized in Table 6.

In short, as for the acute toxicity of PAHs, only animal studies for rats had exhibited high toxicity of PAH, but the acute toxicity for humans is still not clear or it cause a low impact on human health. Chronic exposure to PAHs may cause adverse impacts on the respiratory system, liver, blood and skin. Furthermore, PAHs could hamper the growth and the development of fetus. The exposure to PAHs may lead to reduce infant size, to impede motor and cognitive development, and to diminish fertility rate. Regarding the carcinogenic risks from exposure to PAHs, it may cause cancer in lung, breast, bladder, forestomach and blood. To explain, once PAHs infiltrate the human body, some PAHs become available to bind DNA and they hamper DNA replication. As a result, DNA mutations occur during the replication, and the modified genetic information might cause cancers.

Table 6. Summary of human health risks from exposure to PAHs

Non-carcinogenic Effects
Acute Effects
- No reports of effects on human
- High toxic effects from oral ingestion of benzo[a]pyrene in mice experiments
Chronic Effects
- Respiratory diseases(asthma, bronchitis) from inhalation of particle-bound PAHs
- Skin disorder from dermal exposure to PAHs
- Toxic effects on blood and liver from oral ingestion of Benzo[a]pyrene in animal tests
Reproductive Effects
- Intrauterine growth restriction: lower birth weight, preterm birth and lower fertility rate
Developmental Effects
- Motor developmental delay
- Cognitive developmental delay
Carcinogenic Effects
- Lung cancer risks from exposure to coke oven emission, roofing tar and cigarette smoke
- Breast and bladder cancer risks from PAH-DNA adducts in adults
- Respiratory tract tumor, forestomach tumor, leukemia and lung tumor from exposure to benzo[a]pyrene in animal oral exposure tests

* The health risk information is collected from EPA's fact sheet for hazardous air pollutants(EPA, 2016) and WHO guidelines for indoor air quality(WHO, 2010)

5. CHAPTER 3 – EXPERIMENTAL DESIGN

5.1 Sampling Sites

This study investigated 9 air-conditioned bus interchanges in Singapore to profile IAQ levels in the sites for 3 months from January 2019 to March 2019. Owing to the variances of both VOCs and PAHs sources from different areas, sampling points were cautiously selected for each bus interchange both indoor and bus park with consideration of the flow of human traffic. Samplings were carried out from 6 AM to 10 PM and all instruments were placed at least 1.5 m above the ground to stimulate the breathing zone.

Table 7. The backgrounds of Singapore's 9 bus interchanges

Bus interchanges	Bus park structure	Surroundings	Bus services
#1	Fully Sheltered	Subway stations, shopping mall, residences.	Alighting berth: 4 Boarding berth: 3 Bus routes: 13
#2	Open/Partially Enclosed	Subway station, shopping mall, residences, office buildings.	Alighting berth: 6 Boarding berth: 18 Bus routes: 21
#3	Two separated bus parks: - Fully Sheltered (sampling point) - Open/Partially Enclosed	Subway station, shopping mall, residences.	Alighting berth: 4 Boarding berth: 8 Bus routes: 28
#4	Open/Partially Enclosed	Subway station, shopping mall, manufacturing complex.	Alighting berth: 2 Boarding berth: 4 Bus routes: 9
#5	Open/Partially Enclosed	Subway station, shopping mall, residences, marketplace.	Alighting berth: 3 Boarding berth: 6 Bus routes: 21
#6	Fully Sheltered	Subway station, shopping mall, residences.	Alighting berth: 3 Boarding berth: 10 Bus routes: 38
#7	Open/Partially Enclosed	Subway station, shopping mall, residences, marketplaces.	Alighting berth: 3 Boarding berth: 5 Bus routes: 18
#8	Fully Sheltered	Subway stations, shopping mall, residences.	Alighting berth: 2 Boarding berth: 4 Bus routes: 13
#9	Fully Sheltered	Subway stations, shopping mall, residences.	Alighting berth: 2 Boarding berth: 3 Bus routes: 10

Figure 2. Layouts of Singapore's 9 bus interchanges

Table 7 includes the information of bus interchanges and Figure 2 represents the layouts of 9 bus interchanges. Since this research aimed to estimate how much bus exhaust gas could deteriorate indoor air quality at bus interchanges, ventilation systems of the indoor platforms and bus parks had to be taken into account. All bus interchanges managed indoor air quality by means of central air conditioning systems. However, air quality at bus parks highly depended on natural ventilation. Since the bus park structures, which may affect the migration of air pollutants, were all different depending on distinctive building structures, the bus interchanges were classified into two categories: (1) “open/partially enclosed”, where the bus park is either fully open-air or has a partially sheltered overhead such that air can exchange freely with the outdoors; and (2) “fully sheltered”, means the bus park’s overhead is completely covered. Then, the classification will be used to check if the bus park envelope can restrict the dilution and dispersion of air pollutants. Indoor/Outdoor (I/O) ratio, which is a widely-used concept for tracking the sources of air pollutants by comparing the pollutant concentration of indoor and outdoor, could not be used in this research, but a novel alternative concept, Indoor/Bus Park (I/B) ratio would be used for data analysis.

5.2 Sampling Procedures

Two indoor sampling points and a bus park sampling point were set up for both VOCs and gPAHs in each bus interchange except for #9. A single indoor point was available at #9 bus interchange. For pPAHs sampling, an indoor sampling point and a bus park sampling point were specified. To assess exposure, samplings were conducted for a day from 6 am to 10 pm during operational hours on regular weekday. For each sampling sites, three time slots (starting at 7 a.m., 2 p.m., and 6 p.m.; respectively) were chosen for VOCs detection; two time slots (starting at 6 a.m. and 2 p.m.; respectively) were chosen for gPAHs detection; one time slot (starting at 6 a.m.) was chosen for pPAHs detection. Thus, triplicate VOC samples and duplicate gPAH samples were collected from one sample location per facility to reduce the uncertainties of sampling. A total of 75 VOC samples, 49 gPAHs samples and 50 pPAHs samples were collected from 9 bus interchange (3 VOC samples, 6 gPAHs samples and 4 pPAH samples missed). The data reported in this study were the average values of those replicate measurements. For each batch of samples, a field and laboratory blank were employed.

Portable sampling pumps (GilAir Plus, Sensidyne, LP) with preconditioned Tenax TA sorbent tubes (Markes International, Llantrisant, U.K.) were used to collect VOCs with pumps flow rate at 100 cm³/min for 60 min. Gaseous PAHs (gPAHs) were sampled by

PAH sorbent tubes (Markes International, Llantrisant, U.K.) with pump flow rate at 300 cm³/min for 450 min. After the collection of air samples, all tubes were sealed with long-term storage caps, kept at room temperature, and analyzed within 24 hours. To assess particulate PAHs (pPAHs) concentrations, customized air filter samplers (PTFE membrane disc filter, 47 mm, 1 µm) were used to collect PM_{2.5} and PM₁₀ separately for at least 15 hours at the “Indoor” and bus park and processed using the standard gravimetric method. Every filter used in PM sampling was conditioned and weighed, before and after sampling to determine the mass of PM collected from the air. Then, the filters were packed to preconditioned stainless steel TD tubes (Markes International, Llantrisant, U.K.) and fixed with glass wool and fixed with sorbent-retaining gauze. Figure 3 illustrates the components of the prepared TD tubes (Chen, A., 2015).

Figure 3. Fitting of sorbent tubes for pPAHs analysis (Chen, A., 2015)

Additionally, air temperature and relative humidity (RH) levels were continuously measured for the whole period using TSI VelociCalc® Multi-Function Ventilation Meter Model 9565-P with TSI Indoor Air Quality Probe 982. Temperature and RH measurements were averaged over the entire sampling time for data analysis.

5.3 Analytical Procedures

The sampled and prepared tubes were desorbed with automated thermal desorber (Markes TD100) and analyzed using gas chromatograph (GC, Agilent 7890A) coupled with a mass selective detector (MSD, Agilent 5975C). For VOCs, the temperature program of the GC oven was: 60 °C holds for 2 min, then ramps to 240 °C and holds for 10 min at a rate of 20 °C/min, finally the temperature ramps to 280 °C at a rate of 20 °C/min and holds for 5 min. The flow rate of the carrier gas (helium) was 1.5 mL/min. For PAHs, the GC setting was: 50 °C holds for 1 min, then ramps to 100 °C at a rate of 15 °C/min, 240 °C at a rate of 20 °C/min and to 260 °C at a rate of 10 °C/min. 260 °C holds for 7 min, and finally the temperature ramps to 310 °C at a rate of 40 °C/min and hold for 5 min. The flow rates of

the carrier gas (helium) were 1.5 mL/min for VOCs and 3.0 mL/min for PAHs, individually. The GC column used for separation was DB-5MS (30 m×0.25 mm×0.25 µm) (Agilent Technology Inc., Singapore). The MS worked in a positive ionization model was operated in single ion monitoring (SIM) model to detect and quantify analytes. The target VOCs were quantified by selected ions: $m/z=78$ for benzene, $m/z=92$ for toluene, $m/z=106$ for ethylbenzene and xylene, $m/z=104$ for styrene, $m/z=147$ for 1,4-dichlorobenzene, $m/z=128$ for naphthalene. The target PAHs were detected and quantified by SIM: $m/z=166.6$ for fluorene, $m/z=178.6$ for phenanthrene and anthracene, $m/z=202.8$ for fluoranthene and pyrene, $m/z=228.8$ for benz[a]anthracene $m/z=229$ for chrysene, and $m/z=253$ for benzo[b]fluoranthene and benzo[a]pyrene. Due to the diverse characteristics of target compounds, VOCs and PAHs had different settings of TD-GC-MS. The settings of analytical instrument are detailed in appendix A.

5.4 Chemical and Reagents

Twelve target VOCs including benzene, toluene, tetrachloroethylene, ethylbenzene, m-xylene, p-xylene, o-xylene, styrene, 1,4-dichlorobenzene, 1,2-dichlorobenzene, 1,3-dichlorobenzene and naphthalene, and 16 EPA priority pollutant PAHs, acenaphthene (Ace), acenaphthylene (Acy), fluorene (Fl), phenanthrene (Phe), anthracene (Ant), fluoranthene (Fla), pyrene (Pyr), benz[a]anthracene (BaA), chrysene (Chr), benzo[b]fluoranthene (BbF), benzo[k]fluoranthene (BkF), benzo[a]pyrene (BaP), dibenz[a,h]anthracene (DBaA), indeno[1,2,3-cd]pyrene (Ind), and benzo[g,h,i]perylene (BghiP), were selected for sampling due to their toxicities and relatively high possibility of exposure. A VOCs standard mixture (8260B MegaMix, 2 mg/mL of each compound in methanol) containing 12 target VOCs was purchased from Accustandard (NH, U.S.). A PAHs standard mixture (10 µg/mL each component in acetonitrile) including 16 PAHs was also obtained from Sigma Aldrich (St. Louis, Missouri, USA).

5.5 Quality Control

To estimate how much air pollutants were in the collected samples, TD-GC-MS was applied to analyze air samples. Before the analysis, six concentration levels (0.001, 0.01, 0.1, 1, 10, 100 µg/mL) of liquid calibration standards for both VOCs and PAHs were prepared and 2 µL of each standard was spiked into the Tenax TA adsorbent tube using a standard Agilent GC syringe. A calibration solution loading rig (CSLRTM, Markes International, Llantrisant, U.K.) was also used for injecting the standard samples to replicate real samples, and nitrogen gas (Air Liquids Inc, Singapore) was applied in the

rig as a carrier gas with a purity of 99.9% at a flow rate of 100mL/min for VOCs and 300mL/min for PAHs. While spiking standards with GC syringe, the needle was left on the injection port for 20 second to make sure that the standard compounds were transferred to the sorbent tube from the needle. After removing the needle, additional 2 min was required for purging solvents used in standards, and then the sorbent tubes were removed from the rig and loaded on TD-GC-MS for analysis. All sorbent tubes used in both sampling and calibration were properly conditioned at 330°C for 2 h with nitrogen gas and left to cool down for 1 h. The conditioned tubes were tightly capped with long-term storage brass caps and stored at room temperature.

The Method Detection Limits (MDLs) were developed by the sum of the mean concentrations of blank sorbent tubes (n=6) and three times of the standard deviations (Appendix B). Field blanks and laboratory blanks were analyzed with the same batch of samples. All the final calculated concentrations were blank corrected. VOCs and PAHs with measured values lower than their MDLs were assigned to a value of the MDL/2 for the data analysis. Moreover, if the substituted values with MDL/2 accounted for over 60% of total data, the compounds were excluded in data analysis.

5.6 Statistical Analysis

The result data from TD-GC-MS were obtained and analyzed through two application programs, MassHunter Qualitative Analysis and MassHunter Quantitative Analysis (Agilent). GraphPad Prism Version 8 (GraphPad Software Company), was also used for data analysis. Wilcoxon signed rank test, Mann-Whitney test and Pearson or Spearman correlation analysis were conducted through the software and various types of embedded drawing technics were utilized to present the characteristics of pollutants in the bus interchange.

5.7 Health Risk Assessment

According to the latest U.S. EPA's guidelines for human exposure assessment, human health risk assessment has been conducted following National Research Council (NRC)'s four-step process: hazard identification, dose-response assessment, exposure assessment and risk characterization (U.S. EPA, 2019).

For the hazard identification part, the characteristics of VOCs and PAHs, their sources in ambient atmosphere, their fates, the exposure pathways and their potential risks to humans were gathered and stated in previous chapters.

As to the dose-response assessment, the toxicological data of each chemical compound were collected from California Office of Environmental Health Hazard Assessment (OEHHA) and EPA's IRIS to evaluate human health risks.

For the exposure assessment, the daily dose via inhalation, a major pathway for exposure to air pollutants, was calculated, and the dose via dermal absorption was additionally estimated due to the lipophilic attributes of pollutants. Daily doses via inhalation were calculated by following ATSDR's *Public Health Assessment GUIDANCE MANUAL* (ATSDR, 2005) and the calculation of daily doses via dermal absorption of airborne pollutants were followed few studies (Gaspar, F. W., et al., 2014; Weschler, C. J., & Nazaroff, W. W., 2012; Weschler, C. J., & Nazaroff, W. W., 2010). Subsequently, Monte Carlo method was applied to simulate the exposure uncertainty and variability from assigned exposure factors and measured concentrations of pollutants (EPA, 1997). The Monte Carlo simulation was performed 100,000 times, then daily dose via inhalation and dermal absorption were estimated. Further description of dose calculations and Monte Carlo simulations would be delineated later in this chapter.

Lastly, risk characterization, a step to examine whether the emission could cause harm to people or not, was proceeded for each chemical compound. Due to the lack of guidelines and regulations for indoor air quality regarding VOCs and PAHs, the evaluation of carcinogenic and noncarcinogenic risk for the compounds had to be followed with several references. For VOCs, the daily inhalation doses for each compound were compared to OEHHA's standards, and the actual concentration levels at bus interchanges were compared to EPA's reference concentration (RfC). Additionally, as Singapore has a guideline for TVOC (SPRING Singapore, 2016) at indoor spaces, the concentration value was also compared to real samples. For PAHs, a carcinogenic risk was estimated through Incremental Lifetime Cancer Risk (ILCR) by following the EPA's risk assessment guidelines (EPA, 1989). Since all detected PAHs except for benzo[a]pyrene have yet to be evaluated their carcinogenicities, their carcinogenic risks had to be estimated as benzo[a]pyrene using toxic equivalent factors (LaGoy, P.K., & Nisbet, I.C.T., 1992). To quantify the risk with consideration of uncertainty and variability, Monte Carlo method was applied in risk calculation. For evaluating noncarcinogenic risk, the actual concentration levels of benzo[a]pyrene at bus interchanges were compared to EPA's reference concentration.

5.7.1 Dose Calculation

Daily doses of air pollutants were estimated for commuters who regularly use bus interchanges. Inhalation and dermal doses were considered as main exposure routes of the air pollutants, and the doses were calculated based on the indoor concentrations of the pollutants at bus interchanges.

$$D_{total} = D_{inh} + D_{derm} \quad (1)$$

D_{total} is the total dose that a commuter receives while using bus interchanges ($\mu\text{g/kg/day}$), D_{inh} is the inhalation dose, and D_{derm} is the dose through dermal absorption. The equations for the calculations of doses are shown in Table 8 (Gaspar, F. W. et al., 2014).

Table 8. Equations for the calculations of inhalation and dermal absorption dose

Pathway	Equation	Description	No.
Inhalation	$D_{inh} = \frac{C_g \times IR \times EF}{BW}$	D_{inh} : inhalation dose ($\mu\text{g/kg/day}$) C_g : contaminant concentration ($\mu\text{g/m}^3$) IR : daily inhalation rate (m^3/day) EF : exposure factor (unitless) BW : body weight (kg)	(2)
Dermal absorption	$D_{derm} = \frac{C_g \times K_g \times SA \times EF}{BW}$	D_{derm} : dermal dose ($\mu\text{g/kg/day}$) K_g : indoor air transdermal coefficient (m/day) SA : body surface area (m^2/kg) EF : exposure factor (unitless)	(3)

Inhalation rates (IR), body surface areas (SA) and body weights (BW)

Inhalation rates (IR), body surface areas (SA) and body weights (BW) were collected from U.S. Environmental Protection Agency's Exposure Factors Handbook (EPA, 2011), and presented in Table 9. Some of the age groups were merged for further calculation in Monte Carlo simulation, and their standard deviations that the handbook does not provide were estimated with mean values and 95%ile values.

Table 9. The mean values and standard deviations of inhalation rate (IR), body surface area (SA) and body weight (BW) used for dose calculations

Age group	IR (m ³ /day)	SA (m ²)	BW (kg)
Birth to <1 year	8.64 (2.45)	0.40 (0.07)	7.85 (1.90)
1 to <2 years	13.41 (2.92)	0.53 (0.05)	11.4 (1.58)
2 to <3 years	12.99 (2.46)	0.61 (0.05)	13.8 (2.01)
3 to <6 years	12.40 (1.68)	0.76 (0.12)	18.60 (4.62)
6 to <11 years	12.93 (2.50)	1.08 (0.24)	31.80 (12.58)
11 to <16 years	14.34 (2.97)	1.59 (0.29)	56.80 (19.45)
16 to <21 years	15.44 (3.31)	1.84 (0.30)	71.60 (22.13)
21 to <41 years	16.83 (4.35)	1.95 (0.29)	79.59 (23.39)
41 to <61 years	18.55 (3.91)	2.01 (0.30)	83.51 (22.88)
61 to <81 years	14.94 (2.53)	1.94 (0.27)	79.98 (19.60)
≥81 years	12.97 (1.90)	1.77 (0.21)	68.50 (15.26)

Calculation of the pollutant concentrations in air (C_g)

Since some concentration values of PAHs were only available in dust samples, the concentration in air (C_g) had to be estimated from the concentration in dust (C_p). Their relation between two concentrations is represented as equations below.

Table 10. Equations for the calculation of pollutant concentration in air

Equation	Description	No.
$C_g = \frac{C_p}{TSP \times k_{pg}}$	<p>C_g : concentration in air (ng/m³)</p> <p>C_p : concentration in dust (ng/m³)</p> <p>TSP : mass concentration of airborne particles (μg/m³)</p> <p>k_{pg} : particle-air partition coefficient (m³/μg)</p>	(4)
$k_{pg} = \frac{f_{om_part} \times K_{oa}}{\rho_{part}}$	<p>f_{om_part} : volume fraction of organic matter associated with airborne particles (unitless)</p> <p>K_{oa} : octanol-air partition coefficient (unitless)</p> <p>ρ_{part} : density of airborne particles (μg/m³)</p>	(5)

The variables are collected in a reference (Weschler, C. J., & Nazaroff, W. W., 2010) except for the TSP, total suspended particles (μg/m³). The value of TSP, assumed as 20 μg/m³ for indoor SVOCs in the reference, is replaced by the concentration of PM₁₀ samples (mean: 22.3 μg/m³), since the TSP is lower than the concentration of PM₁₀. k_{pg} is partitioning coefficient between gas phase and airborne particles (m³/μg) and it was estimated from Equation (5). f_{om_part} is the volume fraction of organic matter in airborne particles, assumed to 0.4 (unitless). ρ_{part} is the density of airborne particles, assumed to 10⁻¹² μg/m³. K_{oa} is octanol-air partition coefficient (unitless) summarized in Table 12 by chemical compounds.

Calculation of indoor air transdermal coefficient (K_g)

Indoor air transdermal coefficients (K_g), an indicator of absorption capacity of human skin, were also calculated for estimation of dermal dose. The equations applied for K_g are summarized in Table 11 (Weschler, C. J., & Nazaroff, W. W., 2012).

Table 11. Equations applied to estimate indoor air transdermal coefficients

Pathway	Equation	Description	No.
Through skin surface to stratum corneum, $k_{p_{cw}}$ (cm/s)	$\log(k_{p_{cw}}) = 0.7 \log(K_{ow}) - 0.722MW^{\frac{2}{3}} - 5.252$	$k_{p_{cw}}$: permeability coefficient of the layer between skin surfaces and stratum corneum (cm/s) K_{ow} : octanol-water partition coefficient (unitless) MW : molecular weight (g/mol)	(6)
Through stratum corneum to viable epidermis, k_{p_w} (cm/s)	$k_{p_w} = \frac{k_{p_{cw}}}{1 + (k_{p_{cw}} \times \sqrt{MW})/2.6}$	k_{p_w} : permeability coefficient of the layer between stratum corneum and viable epidermis (cm/s)	(7)
Through viable epidermis to dermal capillaries, k_{p_b} (cm/s)	$k_{p_b} = k_{p_w} \times K_{wa}$	k_{p_b} : permeability coefficient of the layer between viable epidermis and dermal capillaries (cm/s) K_{wa} : water-air partition coefficient (unitless)	(8)
Overall bulk air to dermal capillaries, K_g (m/day)	$K_g = \frac{864}{\frac{1}{\gamma_d} + \frac{1}{k_{p_b}}}$	K_g : indoor air transdermal coefficient (m/day) γ_d : mass-transfer coefficient from bulk air to skin surface (cm/s)	(9)

The permeability coefficients, $k_{p_{cw}}$, k_{p_w} and k_{p_b} , signify gaseous pollutant's capacity to penetrate each skin layer, a gateway to blood capillaries. $k_{p_{cw}}$ is a permeability coefficient of the layer between skin surfaces and stratum corneum (cm/s), k_{p_w} is permeability coefficient of the layer between stratum corneum and viable epidermis (cm/s), and k_{p_b} is a permeability coefficient of the layer between viable epidermis and dermal capillaries (cm/s). K_{ow} is octanol-water partition coefficient (unitless) and K_{wa} is water-air partition

coefficient (unitless). Those coefficients are summarized in Table 12 by chemical compounds. γ_d is a mass-transfer coefficient from bulk air to skin surface, assumed 0.167 cm/s.

Table 12. Partition coefficients of selective VOCs and PAHs

Analyte	MW	$\log K_{ow}$	$\log K_{oa}$	$\log K_{wa}$
Benzene	78.11	2.130	2.774	0.644
Toluene	92.14	2.730	3.296	0.566
Ethylbenzene	106.17	3.150	3.642	0.492
Xylene(m,p)	106.17	3.175	3.716	0.541
Xylene(o)	106.17	3.160	3.910	0.674
Styrene	104.15	2.950	3.899	0.949
1,2-dichlorobenzene	147	3.430	4.535	1.105
Naphthalene	128.18	3.300	5.045	1.745
Fluorene	166.22	4.180	6.585	2.405
Phenanthrene	178.24	4.460	7.222	2.762
Anthracene	178.24	4.450	7.093	2.643
Fluoranthene	202.26	5.160	8.601	3.441
Pyrene	202.26	4.880	8.193	3.313
Benz[a]anthracene	228.3	5.760	9.069	3.309
Chrysene	228.3	5.810	9.480	3.670
benzo[b]fluoranthene	252.32	5.780	10.351	4.571
Benzo[a]pyrene	252.32	6.130	10.869	4.729

* Partition coefficients were estimated from U.S. EPA's EPI Suite program.

Exposure Factor Calculations (EF)

Exposure Factor (EF), a correction factor based on the frequency of exposure to certain chemicals, was calculated for commuters who use the bus interchanges. For the estimation of the number of hours the people spent in bus interchanges, a lower bound of 0.15 hour per day and an upper bound of 0.58 hour per day were assigned by referring to a document about Singapore's bus operations (Menon, G., & Kuang, L. C., 2006), and the values were doubled for a round trip. It was assumed that the commuters used the sites five days per week and 50 weeks per year with the exclusion of the period of annual leave. The EF is calculated:

$$EF = \frac{2 \text{ (times)} \times \frac{F \text{ (varies)}}{24 \text{ hour}} \times \frac{24 \text{ hour}}{1 \text{ day}} \times \frac{5 \text{ day}}{1 \text{ week}} \times \frac{50 \text{ week}}{1 \text{ year}}}{\frac{365 \text{ day}}{1 \text{ year}}} \quad (10)$$

Where, F is the time of exposure in bus interchange per day (hour).

Monte Carlo Simulation

Uncertainty and variability in the dose estimation were considered and quantified via a Monte Carlo simulation, where input parameters are randomly drawn from their probability distribution. Except for the actual air pollutant's concentrations, which were log-normally distributed, other parameters were assumed to be normally distributed. The simulations were conducted 100,000 times to obtain stable dose results. For input parameters that their distributions were unknown, such as EF, K_{oa} and K_g , a concept of coefficient of variation (CV), a ratio of the arithmetic standard deviation to the arithmetic mean, was used, and the value of 1 was assigned. For the input parameters that their CVs were stated in the reference paper (Gaspar, F. W., et al., 2014), such as f_{om_part} (CV = 0.3) and ρ_{part} (CV = 0.25), the CVs were extracted and applied for dose calculation. For other parameters, arithmetic mean values and arithmetic standard deviations were used for dose calculation.

5.7.2 Risk Characterization

Volatile Organic Compounds (VOCs)

Firstly, the daily doses of VOCs were compared to OEHHA's standards. Under California's Proposition 65, OEHHA provides "Safe Harbor Levels", which include No Significant Risk Levels (NSRLs) for carcinogens and Maximum Allowable Dose Levels (MADLs) for chemicals causing reproductive toxicity. As the dose calculations were conducted for diverse age groups, their risk levels were properly adjusted. Since NSRLs were created based on an adult body weight of 70 kg, the adjustments were followed in accordance with each average weight of the age group (Table 9). Additionally, Age-specific Cancer Risk Factor (ACRF) of 10 for children below the age of two years, the factor of 3 for children between the ages of two and fifteen years, and the factor of 1 for other groups were applied following OEHHA's guideline (OEHHA, 2009):

$$NSRL_{adjusted} = NSRL_{standard} \times \frac{BW_{specific} (kg)}{BW_{standard} (70 kg)} \times \frac{1}{ACRF} \quad (11)$$

Similarly, the values of MADLs were adjusted, since MADLs was calculated based on a human female body weight of 58 kg:

$$MADL_{adjusted} = MADL_{standard} \times \frac{BW_{specific} (kg)}{BW_{standard} (58 kg)} \quad (12)$$

Table 13 is a summary of NSRL and MADL by age groups. The table only contains NSRL and MADL of few analytes, since the safe harbor levels of other compounds were not available.

Table 13. A summary of adjusted NSRL and MADL of analytes

Age Group	NSRL _{adjusted} (µg/day)				MADL _{adjusted} (µg/day)	
	Benzene	Ethylbenzene	Styrene	Naphthalene	Benzene	Toluene
Birth to <1 year	0.15	0.61	0.30	0.07	6.63	1758.96
1 to <2 years	0.21	0.88	0.44	0.09	9.63	2555.17
2 to <3 years	0.85	3.55	1.77	0.38	11.66	3093.10
3 to <6 years	1.15	4.78	2.39	0.51	15.71	4168.97
6 to <11 years	1.97	8.18	4.09	0.88	26.87	7127.59
11 to <16 years	3.52	14.61	7.30	1.57	47.99	12731.03
16 to <21 years	13.30	55.23	27.62	5.93	60.49	16048.28
21 to <41 years	14.78	61.40	30.70	6.59	67.24	17839.03
41 to <61 years	15.51	64.42	32.21	6.92	70.55	18718.50
61 to <81 years	14.85	61.70	30.85	6.63	67.57	17927.05
≥81 years	12.72	52.84	26.42	5.68	57.87	15353.45

Then, Hazard Quotient (HQ), a hazard index of exposure to certain chemicals, was calculated for each analytes with the following equation:

$$HQ = \frac{\text{estimated daily dose}}{\text{reference dose}} = \frac{D_{inh} (\mu\text{g/kg/day}) \times BW (kg)}{MADL \text{ or } NSRL (\mu\text{g/day})} \quad (13)$$

If the HQ is less than or equal to 1, adverse health effects are not likely to occur, so it can be considered to have a negligible hazard. Conversely, if the HQ is greater than 1, the adverse health issues from chemical exposure could possibly arise. Only the inhalation dose was used in the estimation because there was no reference dose for dermal absorption.

Furthermore, the actual concentration levels at bus interchanges were compared to EPA's reference concentration. Since few VOCs have been assessed by EPA, the RfC of each VOC from noncancer assessment was collected and summarized in Table 14.

Table 14. EPA's Reference Concentration (RfC) for selective VOCs

Analytes	Benzene	Toluene	Ethylbenzene	Xylenes	Styrene	Naphthalene
RfC (µg/m ³)	30	5000	1000	100	1000	3

Similarly, the HQs were calculated for each analytes with the following equation:

$$HQ = \frac{\text{exposure concentration}}{\text{reference concentration}} = \frac{C_g (\mu\text{g/m}^3)}{RfC (mg/m^3)} \quad (14)$$

Lastly, the concentration levels of TVOC at bus interchanges were also compared to Singapore's guideline, 1000 ppb (SPRING Singapore, 2016).

Polycyclic Aromatic Compounds (PAHs)

For the carcinogenic risks from the PAHs exposure, the incremental possibility of an individual developing cancer over a lifetime called Incremental Lifetime Cancer Risk (ILCR) was estimated by following the EPA's risk assessment guidelines (EPA, 1989). Firstly, all concentration values of each PAH except for benzo[a]pyrene were converted to the values of benzo[a]pyrene with their toxic equivalent factors (LaGoy, P.K., & Nisbet, I.C.T., 1992), since their toxicities have yet to be evaluated. The relation between benzo[a]pyrene and the other PAHs are shown as Equation (15).

$$\sum C_{BaP(eq)} = \sum C_i \times TEF_i \quad (15)$$

Where i represents PAH type, C is the PAH concentration acquired from PM_{10} dust samples, Toxic Equivalent Factor (TEF) is the factor, which expresses the toxicity of chemical as benzo[a]pyrene, and $C_{BaP(eq)}$ is the PAH concentration after the conversion. Table 15 contains TEF of each PAH.

Table 15. Toxic Equivalent Factors (TEFs) of selective PAHs

Compounds (i)	Fluoranthene	Pyrene	Benz[a]anthracene	Chrysene	Benzo[b]fluoranthene	Benzo[a]pyrene
TEF_i	0.001	0.001	0.1	0.01	0.1	1

Then, the ILCR for the exposure to PAHs was estimated by following Equation (16) in Table 16 (EPA, 1989; Miwei, S. et al., 2020). The total concentration ($\sum C_{BaP(eq)}$) was used in estimating ILCR. Since the ILCR is the risk that an individual could face during his or her lifetime, the ILCR was not able to be estimated by age groups. Instead, the age group was specified for the economically active age group (age 20 to 60 years), and the physiological attributes of the group, such as IR and BW, were adjusted for the group by combining two selective groups in Table 9 (21 to < 41 years and 41 to < 61 years). EF is the exposure factor mentioned in Equation (10), ED is exposure duration (40 years, 21 to 61 years), SF is the slope factor of benzo[a]pyrene gathered from OEHHHA, $0.0039 (\mu\text{g}/\text{kg}/\text{d})^{-1}$, and LT is lifetime assuming 70 years. Subsequently, Monte Carlo simulation was also conducted considering the uncertainty and variability, which would come from a broad range of age.

Table 16. An equation of ILCR for PAH risk assessment

Equation	Description	No.
$ILCR = \frac{C \times IR \times EF \times ED \times SF}{BW \times LT}$	<i>C</i> : contaminant concentration (ng/m ³)	(16)
	<i>IR</i> : daily inhalation rate (m ³ /day)	
	<i>EF</i> : exposure factor (unitless)	
	<i>ED</i> : exposure duration (yr)	
	<i>SF</i> : slope factor (µg /kg/d) ⁻¹	
	<i>BW</i> : body weight (kg)	
	<i>LT</i> : lifetime (yr)	

Besides, the actual concentration levels of PAHs at bus interchanges were compared to EPA's reference concentrations for estimating the noncarcinogenic risk of PAHs. However, the RfC was only available to benzo[a]pyrene. As a result, the estimation of noncarcinogenic risks of PAHs was solely conducted for benzo[a]pyrene with actual indoor concentration levels at bus interchanges and its RfC (2 ng/m³) by following Equation (14).

6. CHAPTER 4 – RESULTS AND DISCUSSION

6.1 Volatile Organic Compounds

6.1.1 VOC concentrations at Indoor and Bus Park

Benzene, toluene, ethylbenzene, xylene (m,p), xylene (o), styrene, 1,2-dichlorobenzene and naphthalene were well detected in samples, and Total Volatile Organic Compounds (TVOCs) was used for representing cumulative concentrations of 9 analyzed VOCs. The VOCs concentrations in indoors and bus parks at 9 bus interchanges are summarized in Table 17.

Table 17. Summary of VOCs concentrations in Singapore's 9 bus interchanges ($\mu\text{g}/\text{m}^3$)

Compounds	Indoor (n=49)				Bus park (n=26)			
	AVG	STD	MED	Range	AVG	STD	MED	Range
Benzene	2.94	2.56	3.33	0.11 - 7.62	2.62	2.75	0.67	0.11 - 7.79
Toluene	15.11	15.95	8.95	1.42 - 68.99	12.65	14.06	8.71	1.20 - 58.74
Ethylbenzene	3.56	3.05	3.14	0.09 - 11.86	3.57	3.37	2.79	0.24 - 12.94
Xylene (m,p)	3.62	3.34	3.19	0.10 - 14.07	4.17	5.36	2.35	0.23 - 23.68
Xylene (o)	2.55	2.09	1.76	0.10 - 7.30	2.73	2.56	1.55	0.23 - 9.52
Styrene	2.03	1.77	1.61	0.06 - 5.94	2.01	1.84	1.08	0.06 - 5.46
1,2-dichlorobenzene	1.90	1.83	1.37	0.09 - 5.92	1.44	1.85	0.28	0.09 - 5.50
Naphthalene	2.69	1.85	3.15	0.07 - 7.09	1.75	1.70	0.74	0.07 - 4.55
TVOCs	34.40	28.64	30.42	2.10 - 112.20	30.94	29.70	20.24	2.69 - 115.33

AVG: Arithmetic mean value; STD: Standard deviation; MED: Median value

For individual VOC, the median concentrations of indoor samples were higher than bus park. Similarly, the median concentration of TVOCs in indoor areas (median: $30.42 \mu\text{g}/\text{m}^3$; range: $2.10\text{-}112.20 \mu\text{g}/\text{m}^3$) was higher than bus parks (median: $20.24 \mu\text{g}/\text{m}^3$; range: $2.69\text{-}115.33 \mu\text{g}/\text{m}^3$). Among the VOCs, toluene was the highest concentrated VOCs in indoor samples (median: $8.95 \mu\text{g}/\text{m}^3$; range: $1.42\text{-}68.99 \mu\text{g}/\text{m}^3$), and also in bus park samples (median: $8.71 \mu\text{g}/\text{m}^3$; range: $1.20\text{-}58.74 \mu\text{g}/\text{m}^3$). This trend parallels findings in other studies on vehicle exhaust; VOC studies in bus stations (Lin, C. et al., 2020; Kim, Y.M. et al., 2001), in buses (Parra, M. et al., 2008; Lau, W.-L., & Chan, L.-Y., 2003), on roads with high traffic (Yu, C. H. et al., 2014; Ho, K., & Lee, S., 2002), and indoor car parks (Castro, B. P. et al., 2015) reported toluene as the most abundant VOCs. Furthermore,

numerous research projects on elemental analysis of exhaust gases from various vehicles also show the consistent trend (Watson, J. G. et al., 2001; Schauer, J.J. et al., 1999).

The higher indoor VOCs concentrations than bus parks could be due to the indoor VOCs sources, as shown in other indoor microenvironments, such as offices (Zuraimi, M. et al., 2006; Daisey, J. et al., 1994), residents (Lee, J. Y. et al., 2018; Yamaguchi, T. et al., 2006) and shopping malls (Tang, J. et al., 2005). The accumulation of VOCs could also be plausible explanation for the higher indoor VOCs. Natural ventilation originated from the structural openings of bus parks could reduce the bus park VOCs levels, whereas the ventilation system for the indoor spaces could not do as well (Riain, C. N. et al., 2003). Higher indoor air pollutants levels have also been reported in another bus interchange study (Lin, C. et al., 2020), and shown in the computational fluid dynamics model study, which simulated the environment where a building was close to busy roads (Tong, Z. et al., 2016). Moreover, the reason why toluene was the highest concentrated VOC in the bus interchanges would be due to the universal usage of toluene mixed in paints, adhesives and fragrances or it could be due to the emissions from buses.

6.1.2 Effect of Bus Interchange Structure on VOCs migration

Since the bus park structures could affect the migration of VOCs in the bus interchanges depending on how much the sites are exposed to outdoor environment, all VOCs samples were divided into the two categories (fully sheltered and open/partially enclosed bus interchanges) and analyzed by comparing the bus park VOCs concentrations to indoor

Figure 4. Box plot of indoor and bus park VOC concentrations at bus interchanges categorized as fully sheltered (a) and open/partially enclosed (b). “*”, “” and “***” represents statistical significance “ $p < 0.05$ ”, “ $p < 0.01$ ” and “ $p < 0.001$ ” from comparison between indoor and bus park concentration levels by Wilcoxon signed rank test; respectively.**

VOCs concentrations. The median concentration of VOCs and their distributions are shown in Figure 4.

For individual VOC, no significant concentration difference was found between indoor and bus park samples, except 1,2-dichlorobenzene and naphthalene in both types of bus interchanges. Those compounds showed higher indoor concentrations than bus parks throughout all bus interchanges. For the open/partially enclosed bus interchanges (Figure 4 (b)), the median concentrations of VOCs at indoors were slightly higher than at bus parks in general, whereas there was no significant difference between indoor and bus park in fully sheltered samples except for 1,2-dichlorobenzene and naphthalene (Figure 4 (a)).

Furthermore, the correlation analysis between indoor VOC samples and bus park VOC samples was conducted using the TVOC concentration values. The purpose of this analysis was to investigate how much pollutants at bus park would affect indoor environment and vice versa. Spearman Correlation analysis was conducted ($n=26$), and Figure 5 is the result of the correlation analysis.

Figure 5. Plot of TVOCs concentration levels from indoor and bus park air samples ($n=26$). “*” represents the statistical significance “ $p < 0.001$ ” from comparison between indoor and bus park by Spearman correlation.**

The VOCs concentration values of indoor samples and bus park samples were highly correlated with r value 0.90 ($p < 0.001$). This result suggests the indoor space and bus park are closely related, so VOCs from indoor or bus park could affect the other space.

Moreover, Pearson correlation analysis was conducted ($n=76$) to identify what was the dominant VOCs contributor at bus interchanges. Both indoor samples and bus park samples were collected, and the correlation analysis between VOCs components was conducted. Since all VOCs except for 1,2-dichlorobenzene were the constituents of diesel exhaust (Watson, J. G. et al., 2001; Schauer, J.J. et al., 1999), if the bus exhaust was a

dominant VOCs source at bus interchanges, the correlation between those VOC components would be highly correlated. On the contrary, if there were other VOCs contributors at bus interchanges, such as emissions from connected shopping mall, smoking and cooking, they would make the correlation between compounds weaken. Table 18 represents the correlations between compounds at bus interchanges.

Table 18. Pearson Correlation Coefficients between VOCs components (n=75)

Compounds	1	2	3	4	5	6	7	8
1. Benzene	1.0000							
2. Toluene	0.7102***	1.0000						
3. Ethylbenzene	0.8719***	0.7351***	1.0000					
4. Xylene (m,p)	0.7910***	0.8412***	0.8648***	1.0000				
5. Xylene (o)	0.8164***	0.6760***	0.9439***	0.9003***	1.0000			
6. Styrene	0.8157***	0.5245***	0.8975***	0.7678***	0.9459***	1.0000		
7. 1,2-dichlorobenzene	0.6928***	0.2863*	0.7617***	0.4882***	0.7686***	0.9123***	1.0000	
8. Naphthalene	0.6899***	0.3873***	0.6899***	0.5196***	0.7196***	0.8513***	0.8719***	1.0000

* Correlation is significant at the 0.05 level

*** Correlation is significant at the 0.001 level

The VOCs were correlated well with each other, and it was predictable that those VOCs components would be highly and positively correlated with each other, since the dominant source of VOCs in bus interchanges was considered as bus exhaust from bus parks. The high correlations between VOCs components were also reported in other studies, such as VOCs sampling in the public buses (Parra, M. et al., 2008).

The findings from the correlation analyses imply the bus exhaust would be the dominant VOC source in bus interchanges and the VOCs can be readily transported from bus parks to indoors and vice versa. Then, the result that most VOC concentrations from indoor samples and bus parks were not significantly different (Figure 4) could be interpreted the air exchange between the spaces was frequently occurred whenever passengers get on and off their buses. The dominant VOCs source seems to be bus exhaust, but other strong sources of 1,2-dichlorobenzene and naphthalene might exist in indoor spaces, as the relatively high indoor concentrations suggest. Therefore, the higher indoor VOCs concentrations than bus park observed in open/partially enclosed bus interchanges (Figure 4 (b)) might not imply the other indoor VOC sources but the strong natural ventilation originated from the structural openings of bus parks. If there were other strong VOC sources in indoor areas, the trend should be shown in fully sheltered (Figure 4 (a)).

6.2 Gaseous Polycyclic Aromatic Hydrocarbons

6.2.1 gPAHs concentrations at Indoor and Bus park

Among the target PAHs, 7 types of PAHs, fluorene, phenanthrene, anthracene, fluoranthene, pyrene, benz[a]anthracene and chrysene, were well detected in gPAHs samples and Total gaseous Polycyclic Aromatic Hydrocarbons (TgPAHs) was used for representing cumulative concentrations of the 7 analyzed gPAHs. The gPAHs concentrations in indoors and bus parks at 9 bus interchanges are summarized in Table 19.

Table 19. Summary of gPAHs concentrations in Singapore's 9 bus interchanges (ng/m³)

Compounds	Indoor (n=32)				Bus park (n=17)			
	AVG	STD	MED	Range	AVG	STD	MED	Range
Fluorene	2.81	3.27	0.80	0.12-10.66	4.39	5.97	1.47	0.10-17.85
Phenanthrene	2.51	3.34	1.16	0.01-11.44	1.18	1.64	0.49	0.01-6.68
Anthracene	1.23	1.43	0.93	0.01-6.47	0.79	1.03	0.61	0.01-3.33
Fluoranthene	1.68	1.92	0.45	0.02-6.81	1.43	1.33	0.50	0.18-3.75
Pyrene	2.60	4.27	0.62	0.00-19.70	4.98	9.33	0.38	0.00-25.92
Benz[a]anthracene	4.09	5.21	0.64	0.03-14.22	4.06	4.76	0.95	0.03-11.47
Chrysene	5.39	4.44	7.53	0.12-12.96	5.01	4.32	7.55	0.12-9.74
TgPAHs	20.30	17.85	18.99	1.42-59.86	21.85	19.24	24.26	1.69-56.95

AVG: Arithmetic mean value; STD: Standard deviation; MED: Median value

For individual gPAH, there was no remarkable difference in median concentration between bus park and indoor samples. Some concentration levels of gPAHs at bus park were higher than indoor concentrations, and other gPAHs had the opposite results. The median concentration of TgPAHs at bus parks (median: 24.26 ng/m³; 1.69-56.95 ng/m³) was higher than the concentration of indoors (median: 18.99 ng/m³; range: 1.42-59.86 ng/m³). The most abundant gPAH was chrysene (median: 7.53 ng/m³; range: 0.12-12.96 ng/m³) in indoors, and benz[a]anthracene (median: 0.95 ng/m³; range: 0.03-11.47 ng/m³) in bus parks.

Similar to the result of VOCs, the concentration levels of gPAHs in indoor air samples were comparable to the bus park. According to other studies about PAH measurement in diverse indoor microenvironments, exhaust gases from outdoor environment and indoor smoking were the major sources of PAHs in industrialized countries (WHO, 2010). Since smoking was strictly prohibited in indoor space of Singapore's bus interchanges, it can be reasonably considered that the bus exhaust was the dominant contributor of PAHs at bus interchanges. Then, the similar levels of gPAHs concentrations between bus parks and adjacent indoor spaces might be caused by the migration of the pollutants from bus parks to indoor areas.

6.2.2 Effect of Bus Interchange Structure on gPAHs migration

Since the bus park structures could affect the migration of gPAHs in the bus interchanges depending on how much the sites are exposed to outdoor environment, all gPAHs samples were divided into the two categories (fully sheltered and open/partially enclosed bus interchanges) and analyzed by comparing the bus park gPAHs concentrations to indoor gPAHs concentrations. The median concentration of gPAHs and their distributions are shown in Figure 6.

Figure 6. Box plot of indoor and bus park gPAHs concentrations at bus interchanges categorized as fully sheltered (a) and open/partially enclosed (b). “*” and “**” represents statistical significance “ $p < 0.05$ ” and “ $p < 0.01$ ” from comparison between indoor and bus park concentration levels by Wilcoxon signed rank test; respectively.

Generally, there was no significant concentration difference between indoor and bus park samples for most of the gPAHs (Figure 6 (a) and (b)), but slightly higher median values of indoor gPAHs than bus park were found in “Open/Partially Enclosed” samples (Figure 6 (b)), as represented in VOCs samples (Figure 4 (b)).

Furthermore, correlation analysis between indoor gPAHs samples and bus park gPAHs samples was conducted using the TgPAH concentration values ($n=17$). Figure 7 illustrates the correlation of TgPAHs between indoor samples and bus park samples. The

concentration values were highly correlated with r value 0.75 ($p < 0.001$), and the result implies gPAHs could migrate between the spaces.

Moreover, Pearson Correlation analysis between gPAHs components was conducted

Figure 7. Plot of TgPAHs concentration levels from indoor and bus park air samples (n=17). “*” represents the statistical significance “ $p < 0.001$ ” from comparison between indoor and bus park by Spearman correlation.**

using all gPAHs samples at bus interchange (n=55). Table 20 represents the correlations between gPAHs components.

Table 20. Pearson Correlation Coefficients between gPAHs components (n=49)

Compounds	1	2	3	4	5	6	7
1. Fluorene	1.0000						
2. Phenanthrene	0.6009***	1.0000					
3. Anthracene	0.5832***	0.5986***	1.0000				
4. Fluoranthene	0.4176**	0.6931***	0.1654	1.0000			
5. Pyrene	0.6355***	0.0627	0.4113**	-0.1968	1.0000		
6. Benz[a]anthracene	0.3407*	0.5629***	0.0508	0.9233***	-0.2712	1.0000	
7. Chrysene	0.5882***	0.5510***	0.4093**	0.7244***	0.2496	0.7346***	1.0000

* Correlation is significant at the 0.05 level

** Correlation is significant at the 0.01 level

*** Correlation is significant at the 0.001 level

Most of gPAHs were well correlated with each other except for few cases. Fluorene, phenanthrene, anthracene and chrysene, typical tracers of primary emission from diesel exhaust (Tavares, M. et al., 2004), presented highly positive correlations between each other, and the result would suggest the main source of gPAHs at bus interchanges was diesel exhaust.

Similar to the findings in the result of VOCs, open/partially enclosed bus interchanges (Figure 6 (b)) had higher indoor gPAHs concentrations than bus parks, whereas no significant concentration difference was found between indoor and bus park samples at

fully sheltered bus interchanges (Figure 6 (a)). Considering that bus exhaust is regarded as the major gPAH source and the pollutant could be readily transported from bus parks to indoor spaces, the lower gPAHs concentrations at bus parks than indoors observed in open/partially enclosed bus interchanges would be due to the strong natural ventilation originated from the structural openings of the bus parks.

6.3 Particulate Polycyclic Aromatic Hydrocarbons

6.3.1 Indoor and Bus park pPAHs concentrations

Among the target PAHs, 6 types of PAHs, fluoranthene, pyrene, benz[a]anthracene, chrysene, benzo[b]fluoranthene and benzo[a]pyrene, were well detected in particulate matters. The purpose of collecting pPAHs in different sizes of particulate matter was to check whether pPAHs would be selectively localized in a specific size of dust or not. As no significant trend was found between pPAHs and particle size PM_{10} and $PM_{2.5}$, similarly observed in other previous studies (Fang, X. et al., 2019; Zhu, J. et al., 2019; Murillo, J. H. et al., 2017; Kaushik, C. P. et al., 2012), pPAH samples from different particle sizes were pooled for the data analysis. Table 21 represents the concentrations of pPAHs in indoor and bus park at 9 bus interchanges.

Table 21. Summary of pPAHs concentrations in Singapore's 9 bus interchanges (ng/m³)

Compounds	Indoor (n=25)				Bus park (n=25)			
	AVG	STD	MED	Range	AVG	STD	MED	Range
Fluoranthene	0.13	0.14	0.18	0.01-0.60	0.15	0.16	0.19	0.01-0.63
Pyrene	0.21	0.23	0.14	0.00-0.89	0.30	0.49	0.11	0.01-2.14
Benz[a]anthracene	0.38	0.40	0.24	0.03-1.42	0.69	0.64	0.71	0.03-3.43
Chrysene	0.42	0.42	0.21	0.03-1.53	0.43	0.39	0.30	0.03-1.29
Benzo[b]fluoranthene	0.17	0.20	0.12	0.01-0.57	0.25	0.27	0.14	0.01-0.81
Benzo[a]pyrene	0.26	0.64	0.11	0.00-3.35	0.42	0.64	0.15	0.00-2.17
TpPAHs	1.57	1.01	1.38	0.33-4.60	2.24	1.31	1.94	0.47-5.02

AVG: Arithmetic mean value; STD: Standard deviation; MED: Median value

In general, individual pPAH concentrations and TpPAHs concentrations of indoor samples were lower than bus park samples. The most abundant pPAH was chrysene in indoors (median: 0.21 ng/m³; range: 0.03-1.53 ng/m³), and benz[a]anthracene in bus parks (median: 0.71 ng/m³; range: 0.03-3.43 ng/m³). The median TpPAHs concentration in indoors (median: 1.38 ng/m³; range: 0.33-4.60 ng/m³) was lower than the concentration at bus parks (median: 1.94 ng/m³; range: 0.47-5.02 ng/m³).

Interestingly, the pPAHs concentrations in indoors were lower than at bus parks in contrast with the result of VOCs and gPAHs. Considering the source of PAHs was limited and the major source was considered as bus exhaust in the bus interchanges, the result is more intuitive compared to the result of gPAHs. However, contrary to the gaseous pollutants strongly affected by air circulation, the migration of particulate contaminants from bus parks to indoor spaces might be more tardy than gaseous contaminants (Hussein, T. et al., 2004). Then, the pPAH could not be accumulated much in indoor areas. Another possible reason could be due to the other indoor sources of dusts. In the bus interchanges, it was ambiguous that the collected dust was solely originated from bus exhaust, although the bus exhaust contributed to generate a large amount of dust. There were still other possible emission sources of dust, such as many commuters, connected shopping malls and subway stations. The dust from various sources could be flown into dust collectors during sample collection, and they might lower the concentration levels of pPAHs in some samples.

6.3.2 Effect of Bus Interchange Structure on pPAHs migration

Since the bus park structures could affect the migration of pPAHs in the bus interchanges depending on how much the sites are exposed to outdoor environment, all pPAHs samples were divided into the two categories (fully sheltered and open/partially enclosed bus interchanges) and analyzed by comparing the bus park pPAHs concentrations to indoor pPAHs concentrations. The median concentration of pPAHs and their distributions are shown in Figure 8.

Generally, the pPAHs concentration levels of indoors and bus parks are not significantly different (Figure 8), similar to the result of both VOCs (Figure 4) and gPAHs (Figure 6). However, lower median values of indoor pPAHs samples than bus parks were observed in both types of bus interchanges (Figure 8 (a) and (b)).

Figure 8. Box plot of indoor and bus park pPAHs concentrations at bus interchanges categorized as fully sheltered (a) and open/partially enclosed (b). “*” and “***” represents statistical significance “ $p < 0.05$ ” and “ $p < 0.001$ ” from comparison between indoor and bus park concentration levels by Wilcoxon signed rank test; respectively.

Furthermore, Spearman correlation analysis between indoor pPAHs samples and bus park pPAH samples was conducted using the TpPAH concentration values ($n=25$). Figure 9 illustrates the correlation of TpPAHs and those concentration values were moderately correlated with r value 0.39 ($p < 0.05$). The correlation was somewhat weaker than the result of TgPAHs ($r=0.75$; $p < 0.001$) suggesting the migration of particulate pollutants was not as quick as the transportation of the gaseous pollutants.

Figure 9. Plot of TpPAHs levels from indoor and bus park air samples ($n=25$). “*” represents the statistical significance “ $p < 0.05$ ” from comparison between indoor and bus park by Spearman correlation.

Pearson Correlation analysis was conducted (n=50) to check if the major contributor of pPAHs was diesel exhaust gases from bus park. Table 22 represents the correlations between pPAHs components at bus interchanges.

Table 22. Pearson Correlation Coefficients between pPAHs components (n=50)

Compounds	1	2	3	4	5	6
1. Fluoranthene	1.0000					
2. Pyrene	0.2979*	1.0000				
3. Benz[a]anthracene	-0.1225	0.1226	1.0000			
4. Chrysene	0.3430*	0.3322*	-0.1761	1.0000		
5. Benzo[b]fluoranthene	0.2112	-0.0964	0.1695	-0.3392	1.0000	
6. Benzo[a]pyrene	0.1203	0.2403	-0.0575	0.3918**	-0.1518	1.0000

* Correlation is significant at the 0.05 level

** Correlation is significant at the 0.001 level

Most of pPAHs were not correlated with each other except for few cases. Due to the weak correlations between pPAHs components, it was not possible to confirm that the main source of pPAHs in bus interchanges was diesel exhaust. Besides, it was anticipated that the existence of other pPAH sources around bus interchanges or the occurrence of dust irrelevant to pPAHs might lead to an unexpected result.

In the case of tunnel study, the correlations between pPAHs components were positively high because the sources of pPAHs were apparent and the space was simple (Fang, X. et al., 2019). However, in the case of roadside study, when the roads were with lighter traffic, the correlations of pPAHs between components were weaker (Ho, K., & Lee, S., 2002). Another PAH study in various microenvironments also implies the high correlations between PAHs components were derived from the homogeneity of space (Li, Z., 2009). In consideration of other studies, the weak correlations might be due to the complexity of bus interchanges and their diverse building structures.

Overall, the pPAHs concentration levels between indoors and bus parks were not significantly different, as the result of both VOCs and gPAHs showed. However, a distinctive trend was observed in the pPAHs samples, which concentration levels at bus parks were higher than indoors throughout all bus interchanges. Although the diffusion and dispersion of pPAHs were not strong compared to gaseous pollutants, the infiltration of pPAHs was observed through correlation analysis between indoor and bus park samples ($r=0.39$; $p < 0.05$). Moreover, since the pPAHs samples were collected from the dusts, other possible sources of dusts could affect the concentrations of pPAHs, as the correlation analysis between pPAHs components suggested. Therefore, the relatively high bus park concentrations of pPAHs may be caused by tardy migration of pPAHs or the

intervention of other dust sources, which are not relevant to bus emission. The effect of bus interchange structure on pPAH migration would not be shown, but it will be further analyzed in the next chapter with an new indicator, I/B ratio.

6.4 Effect of Bus Park Structure on I/B Air Pollutants Ratios

Indoor to bus park (I/B) ratio, a novel concept, was used to normalize the comparison of pollutant levels in indoor and bus park, since the concentration levels could be varied depending on the operation characteristics of bus interchanges, such as size of operation and the number of commuter. Owing to the varied exposure levels of bus parks to outdoor environments, I/B ratio had to be used instead of indoor to outdoor concentration ratio (I/O ratio), a widely used concept for representing the relationship between indoor and outdoor concentration of air pollutants. The I/B ratios of TVOCs, TgPAHs and TpPAHs were calculated and classified into the two categories of bus interchanges (fully sheltered and open/partially enclosed) to assess the effect of bus park structure on the migration of pollutants (Figure 10).

Figure 10. Box plot of I/B ratios of TVOCs, TgPAHs and TpPAHs by bus interchanges classified by two categories including fully sheltered and open/partially enclosed. “*” represents statistical significance “ $p < 0.05$ ” from comparison of I/B ratios between three bus park types by Mann-Whitney test.

For TVOCs, the median I/B ratio of fully sheltered bus interchanges was 0.98, lower than open/partially enclosed bus interchanges ratio of 1.28. Regarding the TgPAHs, the median I/B ratio of fully sheltered bus interchanges was 0.76, lower than open/partially enclosed bus interchanges ratio of 1.31. As for TpPAHs, the median I/B ratio of fully sheltered bus interchanges was 0.71, which was lower than open/partially enclosed bus interchanges ratio of 0.90.

Overall, the I/B ratios of TVOCs, TgPAHs and TpPAHs in bus interchanges with fully sheltered bus parks and open/partially enclosed bus parks were noted to be significantly different, with lower I/B ratios observed for fully sheltered bus interchanges. Considering the positive correlation between indoor and bus park air pollutants (Figure 5, 7 and 9), the lower I/B ratios observed in fully sheltered bus interchanges indicate the air pollutants at bus parks could not be diluted well. In contrast, the higher I/B ratios shown in open/partially bus interchanges represent the pollutants at bus parks were naturally ventilated well. Therefore, this suggests a lower risk potential of bus park pollutants in open/partially enclosed bus interchanges than fully sheltered bus interchanges, that could likely be contributed by natural air exchanges through bus park's overhead openings.

6.5 Risk Assessment for VOCs and PAHs exposure

6.5.1 Dose Calculation

The amount of air pollutant that an individual could take in a day at bus interchanges was estimated for both VOCs and PAHs. The daily doses via inhalation and dermal absorption were considered, and estimated with Monte Carlo simulation. Then, the total doses, the sum of two routes of doses, were calculated and summarized in Table 23 and Appendix C. It was found that the amount of total dose decreases with age, and the trend was consistent in both inhalation dose and dermal dose. This is because the younger people relatively intake more pollutants via inhalation and dermal absorption in comparison to their body weights (IR/BW and SA/BW presented in Table 9 for dose calculation). This result corresponds to the general knowledge of toxicology that infants and children could be more susceptible to toxic chemicals (Rauh, V. A. et al., 2010).

Table 23. The mean (SD) doses of VOCs ($\mu\text{g/kg/day}$) and PAHs (ng/kg/day) by age groups.

Compounds	Birth to <1 year	1 to <2 years	2 to <3 years	3 to <6 years	6 to <11 years	11 to <16 years	16 to <21 years	21 to <41 years	41 to <61 years	61 to <81 years	≥ 81 years
VOCs ($\mu\text{g/kg/day}$)											
Benzene	2.30 (9.11)	2.30 (8.3)	1.88 (7.18)	1.40 (5.17)	1.06 (6.88)	0.57 (2.29)	0.47 (1.78)	0.45 (1.83)	0.47 (1.8)	0.39 (1.36)	0.39 (1.63)
Toluene	7.62 (12.39)	7.78 (11.39)	6.26 (9.18)	4.60 (6.88)	3.50 (7.86)	1.91 (3.23)	1.57 (2.44)	1.53 (2.44)	1.58 (2.48)	1.29 (1.98)	1.31 (1.95)
Ethylbenzene	2.16 (4.62)	2.18 (4.38)	1.77 (3.63)	1.31 (2.67)	0.98 (2.68)	0.54 (1.25)	0.45 (1.03)	0.43 (0.91)	0.44 (0.92)	0.37 (0.72)	0.37 (0.73)
Xylene (m,p)	2.18 (4.8)	2.23 (4.69)	1.81 (4.19)	1.32 (2.89)	1.02 (3.54)	0.55 (1.3)	0.45 (0.95)	0.43 (0.97)	0.45 (1.01)	0.37 (0.76)	0.37 (0.83)
Xylene (o)	1.49 (2.91)	1.53 (2.81)	1.22 (2.22)	0.90 (1.7)	0.69 (1.86)	0.38 (0.78)	0.31 (0.6)	0.30 (0.6)	0.31 (0.62)	0.26 (0.51)	0.26 (0.5)
Styrene	1.42 (3.98)	1.48 (4.31)	1.15 (3.65)	0.87 (2.86)	0.65 (2.57)	0.36 (1.59)	0.30 (0.84)	0.30 (0.95)	0.30 (0.89)	0.25 (0.72)	0.24 (0.7)
1,2-dichlorobenzene	1.27 (3.9)	1.29 (3.66)	1.04 (3.14)	0.77 (2.72)	0.59 (2.62)	0.33 (1.18)	0.26 (0.84)	0.26 (0.8)	0.27 (0.84)	0.22 (0.67)	0.22 (0.67)
Naphthalene	1.79 (3.87)	1.81 (3.71)	1.48 (2.96)	1.11 (2.42)	0.86 (2.46)	0.47 (1.05)	0.39 (0.8)	0.38 (0.79)	0.39 (0.78)	0.33 (0.66)	0.33 (0.59)
PAHs (ng/kg/day)											

Fluorene	2.04 (4.51)	1.99 (4.50)	1.65 (3.45)	1.32 (2.99)	1.09 (3.27)	0.67 (1.59)	0.58 (1.31)	0.54 (1.16)	0.55 (1.17)	0.49 (1.03)	0.50 (1.06)
Phenanthrene	3.51 (12.15)	3.39 (11.19)	2.90 (9.80)	2.41 (8.61)	2.14 (8.98)	1.43 (10.00)	1.23 (4.85)	1.17 (6.60)	1.15 (4.46)	1.09 (3.98)	1.12 (4.60)
Anthracene	2.49 (14.83)	2.36 (16.60)	2.10 (12.88)	1.71 (10.69)	1.49 (9.78)	0.93 (4.88)	0.81 (5.34)	0.80 (5.43)	0.76 (4.51)	0.71 (3.99)	0.72 (3.71)
Fluoranthene	4.72 (15.07)	4.31 (11.51)	3.97 (10.77)	3.58 (9.41)	3.50 (14.00)	2.49 (8.09)	2.18 (6.43)	2.04 (5.99)	2.02 (6.36)	1.93 (5.47)	2.03 (6.07)
Pyrene	11.83 (100.15)	12.58 (358.49)	9.76 (86.03)	8.61 (81.31)	8.51 (87.96)	6.14 (80.60)	5.51 (104.93)	5.02 (88.16)	4.79 (52.89)	4.72 (43.37)	5.03 (83.01)
Benz[a]anthracene	26.09 (459.96)	22.12 (260.40)	19.32 (147.62)	18.74 (238.23)	17.78 (242.29)	13.11 (163.00)	12.01 (306.60)	9.70 (98.60)	10.09 (89.76)	10.30 (143.05)	11.67 (191.09)
Chrysene	27.77 (115.67)	24.03 (81.43)	22.39 (80.80)	21.10 (81.76)	21.53 (132.54)	14.92 (67.57)	13.62 (72.30)	12.32 (52.22)	12.08 (57.24)	11.58 (43.75)	12.42 (54.47)
Benzo[b]fluoranthene	8.77 (43.41)	7.78 (22.16)	7.31 (21.06)	6.81 (21.98)	6.77 (26.05)	4.73 (16.07)	4.29 (14.26)	4.01 (13.55)	3.88 (12.93)	3.80 (11.73)	3.98 (12.49)
Benzo[a]pyrene	3.30 (9.43)	2.92 (6.87)	2.73 (6.79)	2.56 (6.77)	2.46 (8.04)	1.84 (7.89)	1.59 (4.56)	1.50 (4.35)	1.45 (4.09)	1.44 (4.04)	1.49 (4.26)

Moreover, the relative contribution of chemical doses of inhalation and dermal absorption are visualized in Figure 11 by using median values of doses. Each stacked bar, a constituent of the column labeled with an analyte, represents the dose contribution of specific age group in ascending order of age from left side. For VOCs, inhalation was the dominant exposure route with a median percent of 99.43 % (range: 91.88-99.90 %). On the other hands, dermal absorption was the dominant route for PAHs with a median percent of 81.65 % (range: 17.56-93.94 %). These results are consistent with other papers in that semi-volatile hydrocarbons, which have relatively high octanol-air coefficients and vapor pressures, tend to attach to organic matters rather than hover in the air (Weschler, C. J., & Nazaroff, W. W., 2008; Abdel-Shafy, H. I., & Mansour, M. S., 2016; Jia, S. et al., 2019).

Figure 11. Relative contribution of VOCs and PAHs dose from inhalation and dermal absorption (each black line inside the column represent one age group)

6.5.2 Risk Characterization

Volatile Organic Compounds (VOCs)

To estimate the inhalation risk of VOCs, two types of HQ introduced in Equation (13) and (14) were used. The one was based on OEHHA's Safe Harbor Levels, and the other was originated from EPA's reference concentration. The former HQs are presented in Table 24 and the later HQs are shown in Table 25.

Table 24. Summary of Hazard Quotients (HQ) for VOCs based on OEHHA's guidelines

Safe Levels	Analyte	Age Group	% HQ > 1*	Mean (STD)	5 th %	25 th %	50 th %	75 th %	95 th %
NSRL	Benzene	Birth to <1 year	95.40	123.82 (9.11)	1.09	7.76	26.95	90.23	480.40
		1 to <2 years	95.84	123.81 (8.30)	1.22	8.49	28.96	95.31	495.04
		2 to <3 years	85.18	30.33 (7.18)	0.29	2.02	6.89	22.48	117.64
		3 to <6 years	81.24	22.58 (5.17)	0.21	1.50	5.17	16.76	89.46
		6 to <11 years	73.54	17.01 (6.88)	0.13	0.92	3.22	11.12	63.89
		11 to <16 years	64.53	9.23 (2.29)	0.08	0.57	1.99	6.61	36.07
		16 to <21 years	36.82	2.54 (1.78)	0.02	0.16	0.55	1.83	10.04
		21 to <41 years	36.35	2.42 (1.83)	0.02	0.15	0.53	1.79	9.53
		41 to <61 years	37.13	2.53 (1.80)	0.02	0.16	0.56	1.85	9.84
		61 to <81 years	33.49	2.07 (1.36)	0.02	0.13	0.47	1.55	8.12
		≥81 years	33.84	2.11 (1.63)	0.02	0.14	0.48	1.56	8.28
	Ethylbenzene	Birth to <1 year	93.38	27.88 (4.62)	0.76	4.15	11.25	28.61	104.92
		1 to <2 years	94.00	28.26 (4.38)	0.83	4.46	11.96	30.00	105.24
		2 to <3 years	76.66	6.88 (3.63)	0.20	1.08	2.86	7.18	25.46
		3 to <6 years	69.93	5.08 (2.67)	0.15	0.80	2.13	5.31	18.82
		6 to <11 years	57.98	3.80 (2.68)	0.09	0.48	1.35	3.52	14.05
		11 to <16 years	44.04	2.09 (1.25)	0.05	0.29	0.80	2.08	7.84
		16 to <21 years	14.35	0.58 (1.03)	0.02	0.08	0.23	0.59	2.19
		21 to <41 years	13.79	0.56 (0.91)	0.01	0.08	0.22	0.57	2.11
		41 to <61 years	14.26	0.57 (0.92)	0.02	0.09	0.23	0.59	2.14
		61 to <81 years	11.38	0.47 (0.72)	0.01	0.07	0.19	0.49	1.79
		≥81 years	11.36	0.47 (0.73)	0.01	0.07	0.20	0.50	1.75
	Styrene	Birth to <1 year	90.94	36.55 (3.98)	0.53	3.38	10.58	31.58	145.63
		1 to <2 years	91.66	38.21 (4.31)	0.59	3.65	11.37	33.44	149.89
		2 to <3 years	72.50	8.93 (3.65)	0.14	0.88	2.73	8.02	35.12
		3 to <6 years	66.04	6.73 (2.86)	0.10	0.64	1.98	5.77	25.50
		6 to <11 years	55.44	5.04 (2.57)	0.06	0.39	1.26	3.84	19.34
		11 to <16 years	43.64	2.77 (1.59)	0.04	0.24	0.77	2.31	10.75
		16 to <21 years	17.14	0.76 (0.84)	0.01	0.07	0.22	0.65	2.97
		21 to <41 years	16.65	0.75 (0.95)	0.01	0.07	0.21	0.62	2.94
		41 to <61 years	17.27	0.76 (0.89)	0.01	0.07	0.22	0.65	3.08
		61 to <81 years	14.49	0.63 (0.72)	0.01	0.06	0.18	0.55	2.47
		≥81 years	14.50	0.62 (0.70)	0.01	0.06	0.19	0.54	2.43

	Naphthalene	Birth to <1 year	99.20	208.71 (3.86)	5.63	30.45	83.41	213.72	781.92
		1 to <2 years	99.26	212.24 (3.70)	6.15	32.99	88.29	222.65	790.55
		2 to <3 years	96.71	51.49 (2.96)	1.52	8.07	21.47	54.29	193.08
		3 to <6 years	95.35	38.32 (2.42)	1.07	5.74	15.50	39.40	143.94
		6 to <11 years	92.26	29.07 (2.45)	0.65	3.55	9.95	26.59	107.06
		11 to <16 years	87.45	15.65 (1.05)	0.40	2.19	6.00	15.72	59.00
		16 to <21 years	65.27	4.29 (0.80)	0.12	0.62	1.71	4.39	16.09
		21 to <41 years	63.02	4.20 (0.79)	0.11	0.59	1.63	4.23	15.93
		41 to <61 years	64.57	4.28 (0.78)	0.12	0.63	1.72	4.41	16.00
		61 to <81 years	60.40	3.61 (0.66)	0.10	0.54	1.47	3.72	13.50
		≥81 years	60.83	3.54 (0.59)	0.10	0.55	1.47	3.70	13.35
MADL	Benzene	Birth to <1 year	38.63	2.72 (9.11)	0.02	0.17	0.59	1.98	10.56
		1 to <2 years	40.11	2.72 (8.30)	0.03	0.19	0.64	2.10	10.88
		2 to <3 years	34.90	2.22 (7.18)	0.02	0.15	0.51	1.65	8.62
		3 to <6 years	29.01	1.65 (5.17)	0.02	0.11	0.38	1.23	6.56
		6 to <11 years	21.50	1.25 (6.88)	0.01	0.07	0.24	0.81	4.68
		11 to <16 years	13.80	0.68 (2.29)	0.01	0.04	0.15	0.48	2.64
		16 to <21 years	11.68	0.56 (1.78)	0.00	0.03	0.12	0.40	2.21
		21 to <41 years	11.35	0.53 (1.83)	0.00	0.03	0.12	0.39	2.09
		41 to <61 years	11.80	0.56 (1.80)	0.01	0.04	0.12	0.41	2.16
		61 to <81 years	9.74	0.45 (1.36)	0.00	0.03	0.10	0.34	1.78
		≥81 years	9.76	0.46 (1.63)	0.00	0.03	0.11	0.34	1.82
	Toluene	Birth to <1 year	0.02	0.03 (12.38)	0.00	0.01	0.02	0.04	0.12
		1 to <2 years	0.01	0.03 (11.39)	0.00	0.01	0.02	0.04	0.12
		2 to <3 years	0.00	0.03 (9.18)	0.00	0.01	0.02	0.03	0.09
		3 to <6 years	0.00	0.02 (6.88)	0.00	0.00	0.01	0.02	0.07
		6 to <11 years	0.02	0.02 (7.86)	0.00	0.00	0.01	0.02	0.05
		11 to <16 years	0.00	0.01 (3.23)	0.00	0.00	0.00	0.01	0.03
		16 to <21 years	0.00	0.01 (2.44)	0.00	0.00	0.00	0.01	0.02
		21 to <41 years	0.00	0.01 (2.44)	0.00	0.00	0.00	0.01	0.02
		41 to <61 years	0.00	0.01 (2.48)	0.00	0.00	0.00	0.01	0.02
		61 to <81 years	0.00	0.01 (1.98)	0.00	0.00	0.00	0.01	0.02
		≥81 years	0.00	0.01 (1.95)	0.00	0.00	0.00	0.01	0.02

* “% HQ > 1” represents the percent of exposures higher than OEHHA’s Safe Harbor Levels

Since not all VOCs had their risk safety benchmarks, only available chemicals were listed on Table 24 and subsequently categorized by two criteria, NSRL for cancer risk and MADL for reproductive risk. In general, the carcinogenic risks of infants (birth to <2 years) were estimated relatively high for all VOCs compounds. The proportion of benzene, which exceed the safe level, was approximately over 95 %, ethylbenzene (93 %), styrene (91 %) and naphthalene (99 %), respectively. Moreover, for the age groups from 2 years to 15 years, the 76 % of simulation data of benzene exceeded the safety level, ethylbenzene (62 %), styrene (59%) and naphthalene (93%), respectively. The age groups over 15 years old had relatively low HQs except for the naphthalene. The proportion of

benzene, which exceed the safe level was about 35%, ethylbenzene (13%), and styrene (16%), respectively. Only the proportion of HQ for naphthalene, which was over 1, accounted for about 63 %. It could be apprehensive that the results imply the VOC concentrations at bus interchanges are problematic. However, the doses could be overestimated in several calculation. The ACRFs, which are the safety factors for young people as mentioned in Equation (11) and used in the dose estimation, would make their HQs raised considerably. Moreover, since the exposure time (0.15-0.58 hour) assumed in the dose calculation was for the commuter, the carcinogenic risks that the younger age groups will take would be lower than the estimation. Furthermore, the OEHHHA's safe harbor levels are very strict guidelines, so many actual cases could not meet the criteria (Jia, S. et al., 2019; Hoang, T. et al., 2016; Kira, O. et al., 2019). Nevertheless, the carcinogenic risks should be taken seriously because the employees working at the bus interchanges could be exposed to the pollutants for longer time than commuters. Otherwise, the HQs from MADLs were relative lower when they were compared to HQs from NSRLs. Approximately, 80% of the population might apart from reproductive risk of the exposure to benzene, and almost 100% of the population might have no risk of the exposure to toluene.

In Table 25, the HQs from the Equation (14) were calculated for particular 6 VOCs and summarized. As the HQs were originated from the RfC in EPA's IRIS, the risks were about non-cancer diseases.

Table 25. Summary of Hazard Quotients (HQ) for VOCs based on EPA's guidelines

Analytes	RfC	C _g ^b	% HQ > 1 ^c	HQ				
				MIN	25 th %	50 th %	75 th %	Max
Benzene^a	30	2.79 (2.54)	0	0.00	0.01	0.11	0.15	0.24
Toluene^a	5000	14.50 (15.20)	0	0.00	0.00	0.00	0.00	0.01
Ethylbenzene^a	1000	3.41 (2.99)	0	0.00	0.00	0.00	0.00	0.01
Xylenes^a	100	5.89 (5.21)	0	0.00	0.02	0.04	0.07	0.16
Styrene^a	1000	1.92 (1.77)	0	0.00	0.00	0.00	0.00	0.01
Naphthalene^a	3	2.55 (1.80)	49.81	0.02	0.33	0.98	1.32	1.95

a. $\mu\text{g}/\text{m}^3$, b. Arithmetic mean value and standard deviation.

c. “%HQ > 1” represents the percent of exposures higher than EPA's RfC.

In general, the indoor concentration levels of the compounds were lower than their RfCs, but some of naphthalene slightly exceeded the RfC. Approximately, 50% of naphthalene

was higher than its RfC, but did not exceed twice the value. However, the average indoor naphthalene level (mean: $2.55 \mu\text{g}/\text{m}^3$) in the bus interchanges is not observed to be high, when compared to other indoor spaces such as shopping center (mean: $19.9 \mu\text{g}/\text{m}^3$), supermarket (mean: $14.2 \mu\text{g}/\text{m}^3$), hotel (mean: $16.3 \mu\text{g}/\text{m}^3$) and office (mean: $310.8 \mu\text{g}/\text{m}^3$) (Lu, H. et al., 2008; Zuraimi, M. et al., 2006). Nevertheless, the HQ based on RfC is an index which only take account the chemical exposure through inhalation. As shown in Figure 11, the dermal adsorption might be the dominant exposure route for lipophilic compounds, so systematic approaches on the exposure pathways of air pollutants are required in future studies to prevent underestimating the amount of pollutant intake through dermal adsorption and ingestion. The concentration levels of VOCs in various microenvironments from other studies are summarized in Appendix D for reference.

Lastly, the sum of concentrations of VOCs, TVOCs, was compared to the reference concentration of indoor air quality from Singapore guideline. The TVOCs concentration levels in indoors (median: 30.42 ppb; max: 112.20 ppb) was within the acceptable limit of 1000 ppb.

Overall, the risk assessment for VOCs shows that the concentration of naphthalene at bus interchanges may cause health problems, both cancer and non-cancer. Furthermore, the other concentrations of VOCs, benzene, ethylbenzene and styrene, may cause carcinogenic problems, when people would be exposed to the pollutants for a long time.

Polycyclic Aromatic Compounds (PAHs)

The carcinogenic risks of PAHs at bus interchanges were estimated as an incremental lifetime cancer risk, the possibility of an individual developing cancer over a lifetime. Since all detected PAHs had to be converted as benzo[a]pyrene for the estimation, the result of this risk assessment represented overall PAHs at bus interchanges. Moreover, the uncertainties and variabilities originated from physiological attributes of people, exposure duration and various PAH concentrations in the sites were also considered in Monte Carlo simulation.

Table 26. Incremental lifetime cancer risk (ILCR) of PAHs at bus interchanges

Analyte	Age group	ILCR					
		Mean (STD)	MIN	25 th %	50 th %	75 th %	MAX
PAHs	Adult (21 to <61 years)	9.30×10^{-8} (0.00)	5.56×10^{-13}	2.41×10^{-9}	1.06×10^{-8}	4.40×10^{-8}	9.81×10^{-5}

Table 26 represents the carcinogenic risk of total PAHs. The median value is 1.06×10^{-8} with the maximum value of 9.81×10^{-5} . It might be obscure to assess the results because the air pollutants do not have clear risk levels that represent acceptable or unacceptable

thresholds. However, U.S.EPA suggests the upper limit of acceptable ILCR at about 10^{-4} and the target level for the public health at about 10^{-5} (U.S.EPA, 2018). Then, the result in Table 26 could be interpreted the concentration levels of PAHs in Singapore's bus interchanges may not cause cancer.

Moreover, the characterization of non-carcinogenic risk for PAHs was conducted. However, the estimation was only possible for benzo[a]pyrene owing to the limited toxicological data in EPA's IRIS.

Table 27. Summary of Hazard Quotients (HQ) for PAHs based on EPA's guidelines

Analytes	RfC	C _g ^b	% HQ > 1 ^c	HQ				
				MIN	25 th %	50 th %	75 th %	Max
Benzo[a]pyrene^a	2	0.26 (0.64)	1.89	0.00	0.01	0.05	0.13	1.68

a. ng/m³, b. Arithmetic mean value and standard deviation.

c. "%HQ > 1" represents the percent of exposures higher than EPA's RfC.

The HQs in Table 27 were calculated for benzo[a]pyrene with Equation (14). Approximately, 2% of benzo[a]pyrene was exceeded the safety level, but it did not reach twice the value of the level (Max: 1.68). However, the average indoor particulate benzo[a]pyrene level (mean: 0.26 ng/m³) is comparable to other microenvironments (Zhu, J. et al., 2019; Oliveira, M. et al., 2014), such as school (mean: 0.26 ng/m³) and childcare center (mean: 0.36 ng/m³). The concentration levels of PAHs in various microenvironments from other studies are summarized in Appendix 4 for reference. Consequently, the concentrations of benzo[a]pyrene in bus interchanges may barely cause some human health problems.

7. CHAPTER 5 – CONCLUSIONS AND RECOMMENDATIONS

7.1 Summary of Findings and Interpretation

This research was planned for measuring the air quality for VOCs and PAHs at 9 bus interchanges in Singapore, and subsequently estimating the human health risks. For VOCs, toluene was the most abundant VOC in both indoor and bus park environments. There was no significant concentration difference between indoor and bus park samples except for 1,2-dichlorobenzene and naphthalene, which showed consistently higher indoor concentrations than bus park throughout all bus interchanges. The bus park VOCs were highly correlated to indoor VOCs ($r=0.90$; $p < 0.001$), and the strong correlations between each individual VOC were also observed. Considering the structure of bus parks, the higher VOC concentrations were observed in indoors than bus parks at open/partially enclosed bus interchanges, whereas there was no significant concentration difference between indoor VOCs and bus park VOCs at fully sheltered bus interchanges. Those results suggest that the fully enclosed structure might decrease the diffusion of VOC after the emission.

For gPAHs, the most abundant gPAHs was chrysene in indoor environments, and benz[a]anthracene in bus parks. There was no significant concentration difference between indoor and bus park samples. Moreover, there was a strong correlation between indoor and bus park concentrations of gPAHs ($r=0.75$; $p < 0.001$). Considering the structure of bus parks, the higher gPAHs concentration were found in indoors than bus parks at open/partially enclosed bus interchanges, whereas there was no significant concentration difference between indoor gPAHs and bus park gPAHs at fully sheltered bus interchanges, which is consistent to the result of VOCs and indicates retarded diffusion in the enclosed structure with less air exchange.

For pPAHs, two sizes of particles, PM_{10} and $PM_{2.5}$ were collected to check the localization of PAHs. The most abundant pPAHs was chrysene in indoor environments, and benz[a]anthracene in bus parks. Interestingly, lower indoor pPAHs concentrations than bus parks were observed in all bus interchanges, which was not consistent with the result of VOCs and gPAHs. The bus park pPAHs were only slightly correlated to indoor pPAHs ($r=0.39$; $p < 0.05$), which is much less significant than VOCs and gPAH. This suggests the diffusion of particle contaminants might be more retarded than gaseous pollutants.

Furthermore, it was found that the architectural designs of bus parks could affect the migration of air pollutants at bus interchanges. The I/B ratios of TVOCs, TgPAHs and

TpPAHs for open/partially enclosed bus parks were consistently higher than the ratios for fully sheltered bus parks, suggesting the air pollutants at open/partially enclosed bus parks could be easily diluted by natural ventilation. Therefore, the architectural design of bus parks should be taken into consideration in the future construction of transport hub, such as ITH.

For the dose estimation with Monte Carlo Simulation, the amount of total dose for both VOCs and PAHs decreased with age, which implies the infants and children could be more susceptible to the chemical exposure. Moreover, the daily intake of gPAHs by dermal adsorption was estimated 4 times higher than by inhalation. Considering that the inhalation has been considered as a major exposure pathway for air pollutants, additional research studies on the exposure pathway for lipophilic air pollutants would be required in the future.

Regarding to the risk assessment, the concentration levels of VOCs at bus interchanges met Singapore's indoor air quality guidelines, and most VOCs did not exceed RfC values from EPA's IRIS. However, similar to the reports from other countries, benzene, ethylbenzene, styrene and naphthalene could not meet the safe harbor levels of OEHHA. As for the PAH risk assessment, the PAH concentrations of bus interchanges were below the EPA's safety benchmarks.

7.2 Limitations and Recommendations

The major difficulties in this air pollution research came from the sampling procedures. It was challenging to measure the air qualities at the bus interchanges, where the places were always crowded with many people. Air sampling instruments should not impede passenger's way, but at the same time, the samples should reflect the spatial characteristics of the bus interchanges. As a result, a smaller number of samples were collected than expected, and relatively quick methods and the handy instruments had to be set for the measurements. The small size of the samples made it difficult to characterize the migration of pollutants and somewhat raise uncertainty in several analyses.

Moreover, some high concentration values of air pollutants were detected, but they could not be interpreted well in this research. This could be owing to an abnormal rise from unexpected sources, seasonal increase, or other possibilities. Consequently, few samples had to be excluded from data analysis.

Lastly, other limitations were in the dose estimation of air pollutants. We estimated the exposures of gas-phase pollutants via inhalation and additionally considered air-to-skin route by following ATSDR's Public Health Assessment GUIDANCE MANUAL (ATSDR, 2005), but other exposure routes through different media such as airborne particles and through different way of intake like nondietary ingestion still could be estimated. Additionally, the variables used in the dose estimation, such as body surface area and exposure factor, were possibly overestimated depending on the types of clothes that people wear and the working patterns of people.

In sum, obtaining a sufficiently large sample size, applying better sampling instruments and identifying the uncertainty and variability in dose estimation might improve the bus interchange research. In addition, a deeper understanding of aerodynamics in diverse buildings could enhance the value of the research. However, our results still would be valuable for researchers concerning indoor air quality in public spaces, especially for busy transportation hubs.

8. CHAPTER 6 – REFERENCES

- Abdel-Shafy, H. I., & Mansour, M. S. (2016). "A review on polycyclic aromatic hydrocarbons: Source, environmental impact, effect on human health and remediation", Egyptian Journal of Petroleum, Vol. 25, No. 1, pp. 107–123. doi: 10.1016/j.ejpe.2015.03.011
- Agency for Toxic Substances and Disease Registry. (1995). "Toxicological Profile for Polycyclic Aromatic Hydrocarbons". Retrieved from <https://www.atsdr.cdc.gov/ToxProfiles/tp.asp?id=122&tid=25>
- Agency for Toxic Substances and Disease Registry. (2005). "Public Health Assessment GUIDANCE MANUAL(Update)". Retrieved from https://www.atsdr.cdc.gov/publications_HAC.html
- Andersson, J. T., & Achten, C. (2015). "Time to Say Goodbye to the 16 EPA PAHs? Toward an Up-to-Date Use of PACs for Environmental Purposes", Polycyclic Aromatic Compounds, Vol. 35, No. 2-4, pp. 330–354. doi: 10.1080/10406638.2014.991042
- Anthony, D. (1985). "Polycyclic Hydrocarbons and Carcinogenesis" (Vol.283., pp.1-17). Washington (D.C.): American Chemical Society. doi: 10.1021/bk-1985-0283.ch001
- Arey, J., & Atkinson, R. (2003). "PAHs: An Ecotoxicological Perspective" (pp.47-63). Chichester, West Sussex: Wiley. doi: 10.1002/0470867132
- Atkinson, R. (2000). "Atmospheric chemistry of VOCs and NOx", Atmospheric Environment, Vol. 34, No. 12-14, pp. 2063-2101. doi: 10.1016/S1352-2310(99)00460-4
- Atkinson, R., & Arey, J. (2003). "Atmospheric Degradation of Volatile Organic Compounds", Chemical Reviews, Vol. 103, No. 12, pp. 4605–4638. doi: 10.1021/cr0206420
- Binet, P., Portal, J., & Leyval, C. (2000). "Dissipation of 3–6-ring polycyclic aromatic hydrocarbons in the rhizosphere of ryegrass", Soil Biology and Biochemistry, Vol. 32, No. 14, pp. 2011–2017. doi: 10.1016/S0038-0717(00)00100-0
- Boström, C., Gerde, P., Hanberg, A., Jernström, B., Johansson, C., Kyrklund, T., Rannug, A., Törnqvist, M., Victorin, K., & Westerholm, R. (2002). "Cancer Risk Assessment, Indicators, and Guidelines for Polycyclic Aromatic Hydrocarbons in the Ambient Air", Environmental Health Perspectives, Vol. 110, No. S3, pp. 451-489. doi:10.1289/ehp.02110s3451
- Building and Construction Authority. (2010). "Code for Environmental Sustainability of Buildings". Retrieved from https://www.bca.gov.sg/EnvSusLegislation/others/Env_Sus_Code2010.pdf
- California Environmental Protection Agency Office of Environmental Health Hazard Assessment. (2009). "Technical Support Document for Cancer Potency Factors: Methodologies for derivation, listing of available values, and adjustments to allow for early life stage exposures". Retrieved from <https://oehha.ca.gov/media/downloads/cnr/tsdcancerpotency.pdf>
- Canter, L. W., Liu, H.F., Raufer, R. K., & Wagner, C. P. (2000). "Air Pollution" (pp.3-19). Boca Raton, FL: Taylor & Francis Group.
- Castro, B. P., Machado, G. D., Bauerfeldt, G. F., Fortes, J. D., & Martins, E. M. (2015). "Assessment of the BTEX concentrations and reactivity in a confined parking area in Rio de Janeiro, Brazil", Atmospheric Environment, Vol. 104, pp. 22-26. doi:10.1016/j.atmosenv.2015.01.013
- Castro-Jiménez, J., Berrojalbiz, N., Wollgast, J., & Dachs, J. (2012). "Polycyclic aromatic hydrocarbons (PAHs) in the Mediterranean Sea: Atmospheric occurrence, deposition and decoupling with settling fluxes in the water column", Environmental Pollution, Vol. 166, pp. 40–47. doi: 10.1016/j.envpol.2012.03.003
- Castro, D., Slezakova, K., Delerue-Matos, C., Alvim-Ferraz, M. D. C., Morais, S., & Pereira, M. D. C. (2011). "Polycyclic aromatic hydrocarbons in gas and particulate phases of indoor environments influenced by tobacco smoke: Levels, phase distributions, and health risks", Atmospheric Environment, Vol. 45, No. 10, pp. 1799–1808. doi: 10.1016/j.atmosenv.2011.01.018
- Chen, A. (2015). "Occurrence, Human Exposure and Dynamics of Phthalates in ACMV Indoor Environments: Method Development and Field Studies", Doctoral dissertation (42p), Nanyang Technological University, Singapore. Retrieved from <https://dr.ntu.edu.sg/handle/10356/64270>

- Cheng, Y.H., Chang, H.P., & Yan, J.W. (2012). "Temporal Variations in Airborne Particulate Matter Levels at an Indoor Bus Terminal and Exposure Implications for Terminal Workers", Aerosol Air Qual. Res., Vol. 12, No. 1, pp. 30-38. doi:10.4209/aaqr.2011.06.0085
- Chuang, J. C., Mack, G. A., Kuhlman, M. R., & Wilson, N. K. (1991). "Polycyclic aromatic hydrocarbons and their derivatives in indoor and outdoor air in an eight-home study", Atmospheric Environment. Part B. Urban Atmosphere, Vol. 25, No. 3, pp. 369-380. doi: 10.1016/0957-1272(91)90008-3
- Clobes, A. L., Ananth, G. P., Hood, A. L., Schroeder, J. A., & Lee, K. A. (1992). "Human Activities as Sources of Volatile Organic Compounds in Residential Environments", Annals of the New York Academy of Sciences, Vol. 641, pp. 79-86. doi: 10.1111/j.1749-6632.1992.tb16534.x
- Daisey, J., Hodgson, A., Fisk, W., Mendell, M., & Brinke, J. T. (1994). "Volatile organic compounds in twelve California office buildings: Classes, concentrations and sources", Atmospheric Environment, Vol. 28, No. 22, pp. 3557-3562. doi: 10.1016/1352-2310(94)00200-5
- Damato, G., Liccardi, G., Damato, M., & Cazzola, M. (2002). "Outdoor air pollution, climatic changes and allergic bronchial asthma", European Respiratory Journal, Vol. 20, No. 3, pp. 763-776. doi: 10.1183/09031936.02.00401402
- David H. Phillips. (1999). "Polycyclic aromatic hydrocarbons in the diet", Mutation Research, Vol. 443, No. 1-2, pp. 139-147. doi: 10.1016/s1383-5742(99)00016-2
- Dreessen, J., Sullivan, J., & Delgado, R. (2016). "Observations and impacts of transported Canadian wildfire smoke on ozone and aerosol air quality in the Maryland region on June 9-12, 2015", Journal of the Air & Waste Management Association, Vol. 66, No. 9, pp. 842-862. doi: 10.1080/10962247.2016.1161674
- Dundon, R. (2018). "Photos: L.A.'s mid-century smog was so bad, people thought it was a gas attack". Retrieved from <https://timeline.com/la-smog-pollution-4ca4bc0cc95d>
- Environmental Protection Agency. (1989). "Risk Assessment Guidance for Superfund Volume I Human Health Evaluation Manual (Part A). (EPA/540/1-89/002)". Retrieved from https://www.epa.gov/sites/production/files/2015-09/documents/rags_a.pdf
- Environmental Protection Agency. (1997). "Guiding Principles for Monte Carlo Analysis". Retrieved from <https://www.epa.gov/sites/production/files/2014-11/documents/montecar.pdf>
- Environmental Protection Agency. (2002). "Toxicological Review of Benzene". Retrieved from https://cfpub.epa.gov/ncea/iris/iris_documents/documents/toxreviews/0276tr.pdf
- Environmental Protection Agency. (2011). "Exposure Factors Handbook: 2011 Edition". Retrieved from <https://cfpub.epa.gov/ncea/risk/recordisplay.cfm?deid=236252>
- Environmental Protection Agency. (2016). "Health Effects Notebook for Hazardous Air Pollutants: Polycyclic Organic Matter(POM)". Retrieved from <https://www.epa.gov/sites/production/files/2016-09/documents/polycyclic-organic-matter.pdf>
- Environmental Protection Agency. (2017). "Technical Overview of Volatile Organic Compounds". Retrieved from <https://www.epa.gov/indoor-air-quality-iaq/technical-overview-volatile-organic-compounds#definition>
- European Union Publications Office. (2004). "Directive 2004/42/CE". Retrieved from <https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=CELEX:32004L0042>
- Fang, P., Tang, Z.-J., Chen, X.-B., Tang, Z.-X., Chen, D.-S., Huang, J.-H., Zeng, W.-H., & Cen, C.-P. (2016). "Experimental Study on the Absorption of Toluene from Exhaust Gas by Paraffin/Surfactant/Water Emulsion", Journal of Chemistry, Vol. 2016, pp. 1-9. doi: 10.1155/2016/9385027
- Fang, X., Wu, L., Zhang, Q., Zhang, J., Wang, A., Zhang, Y., Zhao, J., & Mao, H. (2019). "Characteristics, emissions and source identifications of particle polycyclic aromatic hydrocarbons from traffic emissions using tunnel measurement", Transportation Research Part D: Transport and Environment, Vol. 67, pp. 674-684. doi: 10.1016/j.trd.2018.02.021

- Franz, M., Arafat, H. A., & Pinto, N. G. (2000). "Effect of chemical surface heterogeneity on the adsorption mechanism of dissolved aromatics on activated carbon", Carbon, Vol. 38, No. 13, pp. 1807–1819. doi: 10.1016/s0008-6223(00)00012-9
- Gammage, R. B., & Matthews, T. G. (1988). "Volatile organic compounds in indoor air: Types, sources, and characteristics", Environmental Progress, Vol.7, No.4, pp. 279–283. doi: 10.1002/ep.3300070417
- Gaspar, F. W., Castorina, R., Maddalena, R. L., Nishioka, M. G., McKone, T. E., & Bradman, A. (2014). "Phthalate Exposure and Risk Assessment in California Child Care Facilities", Environmental Science & Technology, Vol. 48, No. 13, pp. 7593–7601. <https://doi.org/10.1021/es501189t>
- Gennaro, G. D., Amenduni, A., Brattoli, M., Massari, F., Palmisani, J., & Tutino, M. (2016). "Chemical Characterization of odor active volatile organic compounds emitted from perfumes by GC/MS-O", Environmental Engineering and Management Journal, Vol. 15, No.9, pp. 1963–1969. doi: 10.30638/eemj.2016.211
- Golkhorshidi, F., Sorooshian, A., Jafari, A. J., Baghani, A. N., Kermani, M., Kalantary, R. R., Ashournejad, Q., & Delikhoon, M. (2019). "On the nature and health impacts of BTEX in a populated middle eastern city: Tehran, Iran", Atmospheric Pollution Research, Vol. 10, No. 3, pp. 921-930. doi:10.1016/j.apr.2018.12.020
- Gong, M., Zhang, Y., & Weschler, C. J. (2013). "Predicting dermal absorption of gas-phase chemicals: transient model development, evaluation, and application", Indoor Air, Vol. 24, No. 3, pp. 292–306. doi: 10.1111/ina.12079
- Goss, K.-U., & Eisenreich, S. J. (1996). "Adsorption of VOCs from the Gas Phase to Different Minerals and a Mineral Mixture", Environmental Science & Technology, Vol. 30, No. 7, pp. 2135–2142. doi: 10.1021/es950508f
- Hester, R. E., Harrison, R. M., & Derwent, R. G. (1995). "Volatile Organic Compounds in the Atmosphere" (pp.1-16). Letchworth, Herts: RSC.
- Ho, K., & Lee, S. (2002). "Identification of atmospheric volatile organic compounds (VOCs), polycyclic aromatic hydrocarbons (PAHs) and carbonyl compounds in Hong Kong", Science of The Total Environment, Vol. 289, No. 1-3, pp. 145–158. doi: 10.1016/s0048-9697(01)01031-2
- Hoang, T., Castorina, R., Gaspar, F., Maddalena, R., Jenkins, P. L., Zhang, Q., McKone, T. E., Benefenati, E., Shi, A. Y., & Bradman, A. (2016). "VOC exposures in California early childhood education environments", Indoor Air, Vol. 27, No. 3, pp. 609-621. doi:10.1111/ina.12340
- Hodgson, M., Levin, H., & Wolkoff, P. (1994). "Volatile organic compounds and indoor air", The Journal of Allergy and Clinical Immunology, Vol. 94, No. 2, pp. 296-303. doi: 10.1053/ai.1994.v94.a56008
- Huang, H., Haghighat, F., & Blondeau, P. (2006). "Volatile organic compound (VOC) adsorption on material: influence of gas phase concentration, relative humidity and VOC type", Indoor Air, Vol. 16, No. 3, pp. 236–247. doi: 10.1111/j.1600-0668.2005.00421.x
- Indoor air quality: Organic pollutants : report on a WHO meeting, Berlin, West, 23-27 August 1987. (1989). Copenhagen: World Health Organization, Regional Office for Europe.
- Jia, S., Sankaran, G., Wang, B., Shang, H., Tan, S. T., Yap, H. M., Shen, J., Gutiérrez, R. A., Fang, W., Liu, M., Chang, V. W.-C., Ng, L. C., & Fang, M. (2019). "Exposure and risk assessment of volatile organic compounds and airborne phthalates in Singapore's Child Care Centers", Chemosphere, Vol. 224, pp. 85-92. doi:10.1016/j.chemosphere.2019.02.120
- Jones, A.P. (1999). "Indoor air quality and health", Atmospheric Environment, Vol. 33, No. 28, pp. 4535-4564. Retrieved from [https://doi.org/10.1016/S1352-2310\(99\)00272-1](https://doi.org/10.1016/S1352-2310(99)00272-1)
- Kaushik, C. P., Sangwan, P., & Haritash, A. K. (2012). "Association of Polycyclic Aromatic Hydrocarbons (PAHs) with Different Sizes of Atmospheric Particulate in Hisar City and its Health Aspects", Polycyclic Aromatic Compounds, Vol. 32, No. 5, pp. 626–642. doi: 10.1080/10406638.2012.695318

- Kerns, W. D., Pavkov, K. L., Donofrio, D. J., Gralla, E. J., & Swenberg, J.A. (1983). "Carcinogenicity of Formaldehyde in Rats and Mice after Long-Term Inhalation Exposure", Cancer Res, Vol. 43, No. 9, pp. 4382-4392.
- Keyte, I. J., Harrison, R. M., & Lammel, G. (2013). "Chemical reactivity and long-range transport potential of polycyclic aromatic hydrocarbons – a review", Chemical Society Reviews, Vol. 42, No. 24, pp. 9333-9391. doi: 10.1039/c3cs60147a
- Kim, K., Jahan, S. A., Kabir, E., & Brown, R. J. (2013). "A review of airborne polycyclic aromatic hydrocarbons (PAHs) and their human health effects", Environment International, Vol. 60, pp. 71-80. doi:10.1016/j.envint.2013.07.019
- Kim, Y. M., Harrad, S., & Harrison, R. M. (2001). "Concentrations and Sources of VOCs in Urban Domestic and Public Microenvironments", Environmental Science & Technology, Vol. 35, No. 6, pp. 997–1004. doi: 10.1021/es000192y
- Kira Oz, Bareket Merav, Sabach Sara, and Dubowski Yael. (2019). "Volatile Organic Compound Emissions from Polyurethane Mattresses under Variable Environmental Conditions", Environmental Science & Technology, Vol. 53, No. 15, pp. 9171-9180. doi: 10.1021/acs.est.9b01557
- Kishida, M., Imamura, K., Takenaka, N., Maeda, Y., Viet, P. H., & Bandow, H. (2008). "Concentrations of Atmospheric Polycyclic Aromatic Hydrocarbons in Particulate Matter and the Gaseous Phase at Roadside Sites in Hanoi, Vietnam", Bulletin of Environmental Contamination and Toxicology, Vol. 81, No. 2, pp. 174-179. doi:10.1007/s00128-008-9450-5
- Kliucininkas, L., Martuzevicius, D., Prasauskas, T., Kauneliene, V., Molnar, P., & Strandberg, B. (2011). "Indoor and outdoor concentrations of fine particles, particle-bound PAHs and volatile organic compounds in Kaunas, Lithuania", J. Environ. Monit., Vol. 13, No. 1, pp. 182-191. doi:10.1039/c0em00260g
- Kuusimäki, L., Peltonen, K., Mutanen, P., & Savela, K. (2003). "Analysis of Particle and Vapour Phase PAHs from the Personal Air Samples of Bus Garage Workers Exposed to Diesel Exhaust", The Annals of Occupational Hygiene, Vol. 47, No. 5, pp. 389-398. doi:10.1093/annhyg/meg037
- Lai, I.-C., Lee, C.-L., Zeng, K.-Y., & Huang, H.-C. (2011). "Seasonal variation of atmospheric polycyclic aromatic hydrocarbons along the Kaohsiung coast", Journal of Environmental Management, Vol. 92, No. 8, pp. 2029–2037. doi: 10.1016/j.jenvman.2011.03.026
- Land Transport Authority. (2015). "Singapore Land Transport Statistics in brief 2015". Retrieved from https://www.lta.gov.sg/content/dam/ltagov/who_we_are/statistics_and_publications/statistics/pdf/Statistics_in_Brief_2015_FINAL.pdf
- Land Transport Authority. (2019). "Public Transport Ridership". Retrieved from https://public.tableau.com/profile/lta.strategic.planning#!/vizhome/2019Ridership_Public/PTRidershipDashboard
- Land Transport Authority. (2020). "Monthly Vehicle Statistics". Retrieved from https://www.lta.gov.sg/content/dam/ltagov/who_we_are/statistics_and_publications/statistics/pdf/M09-Vehs_by_Fuel_Type.pdf
- Lau, W.-L., & Chan, L.-Y. (2003). "Commuter exposure to aromatic VOCs in public transportation modes in Hong Kong", Science of The Total Environment, Vol. 308, No. 1-3, pp. 143–155. doi: 10.1016/s0048-9697(02)00647-2
- Lawrence H. Keith. (2015). "The Source of U.S. EPA's Sixteen PAH Priority Pollutants", Polycyclic Aromatic Compounds, Vol. 35, No. 2-4, pp. 147-160, DOI:10.1080/10406638.2014.892886
- Lee, J. Y., Kim, K., Ryu, S. H., Kim, C. H., & Bae, G.-N. (2018). "The Relative Importance of Indoor and Outdoor Sources for Determining Indoor Pollution Concentrations in Homes in Seoul, South Korea", Asian Journal of Atmospheric Environment, Vol. 12, No. 2, pp. 127–138. doi: 10.5572/ajae.2018.12.2.127
- Li, Z. (2009). "Characterization of Polycyclic Aromatic Hydrocarbons (PAH) in airborne particles and assessment of human exposure to PAHs", Doctoral dissertation, Georgia Institute of Technology, Atlanta, USA. Retrieved from <http://hdl.handle.net/1853/28122>

- Lin, C., Yen, G., Cheng, Y., & Shie, R. (2020). "Exposure levels of volatile organic compounds and potential health risks for passengers and workers at an intercity bus terminal", Atmospheric Pollution Research, Vol. 11, No. 10, pp. 1820-1828. doi:10.1016/j.apr.2020.07.024
- Little, J. C., Weschler, C. J., Nazaroff, W. W., Liu, Z., & Hubal, E. A. C. (2012). "Rapid Methods to Estimate Potential Exposure to Semivolatile Organic Compounds in the Indoor Environment", Environmental Science & Technology, Vol. 46, No. 20, pp. 11171–11178. doi: 10.1021/es301088a
- Liu, X., Guo, Z., Sparks, L. E., & Roache, N. F. (2011). "VOC Sink Behaviour on Building Materials – Model Evaluation", Indoor and Built Environment, Vol. 20, No. 6, pp. 661–676. doi: 10.1177/1420326x11409461
- Lu, H., Zhu, L., & Chen, S. (2008). "Pollution level, phase distribution and health risk of polycyclic aromatic hydrocarbons in indoor air at public places of Hangzhou, China", Environmental pollution, Vol. 152, No. 3, pp. 569–575. doi: 10.1016/j.envpol.2007.07.005
- Nisbet, I. C., & LaGoy, P. K. (1992). "Toxic equivalency factors (TEFs) for polycyclic aromatic hydrocarbons (PAHs)", Regulatory toxicology and pharmacology : RTP, Vol. 16, No. 3, pp. 290–300. [https://doi.org/10.1016/0273-2300\(92\)90009-x](https://doi.org/10.1016/0273-2300(92)90009-x)
- Maroni, M., Seifert, B., & Lindvall, T., (1995). "Indoor Air Quality. a Comprehensive Reference Book". Amsterdam, Netherlands: Elsevier Science.
- McMurtry, J. (2012). "Organic Chemistry" (8th ed.). Belmont, CA : Brooks/Cole Cengage Learning
- Menon, G., & Kuang, L. C. (2006). "Lessons from Bus Operations". Retrieved from <https://www.ptc.gov.sg/docs/default-source/publications-and-papers/lessonsfrombusoperationsrev5.pdf>
- Menzie, C. A., Hoeppe, S. S., Cura, J. J., Freshman, J. S., & Lafrey, E. N. (2002). "Urban and suburban storm water runoff as a source of polycyclic aromatic hydrocarbons (PAHs) to Massachusetts estuarine and coastal environments", Estuaries, Vol. 25, No. 2, pp. 165–176. doi: 10.1007/bf02691305
- Miwei, S., Rongzhi, Z., Yunxia W., Yan, Z., Yuan, Z., & Yanjie Z. (2020). "Health risk assessments of polycyclic aromatic hydrocarbons and chlorinated/brominated polycyclic aromatic hydrocarbons in urban air particles in a haze frequent area in China", Emerging Contaminants, Vol. 6, pp. 172-178. doi: 10.1016/j.emcon.2020.04.002
- Motelay-Massei, A., Garban, B., Tiphagne-Larcher, K., Chevreuril, M., & Ollivon, D. (2006). "Mass balance for polycyclic aromatic hydrocarbons in the urban watershed of Le Havre (France): Transport and fate of PAHs from the atmosphere to the outlet", Water Research, Vol. 40, No. 10, pp. 1995–2006. doi: 10.1016/j.watres.2006.03.015
- Murillo, J. H., Villalobos, M. C., Marín, J. F. R., Guerrero, V. H. B., & Arias, D. S. (2017). "Polycyclic aromatic hydrocarbons in PM_{2.5} and PM₁₀ atmospheric particles in the Metropolitan Area of Costa Rica: Sources, temporal and spatial variations", Atmospheric Pollution Research, Vol. 8, No. 2, pp. 320–327. doi: 10.1016/j.apr.2016.10.002
- Oliveira, M., Slezakova, K., Delerue-Matos, C., Pereira, M. C., & Morais, S. (2014). "Polycyclic aromatic hydrocarbons: Levels and phase distributions in preschool microenvironment", Indoor Air, Vol. 25, No. 5, pp. 557-568. doi:10.1111/ina.12164
- Oliveira, R. L., Varandas, L., & Arbilla, G. (2014). "Characterization of polycyclic aromatic hydrocarbon levels in the vicinity of a petrochemical complex located in a densely populated area of the Rio de Janeiro, Brazil", Atmospheric Pollution Research, Vol. 5, No.1, pp. 87-95. doi:10.5094/apr.2014.011
- Omar, M. (2006). "Certificate of Entitlement". Retrieved from https://eresources.nlb.gov.sg/infopedia/articles/SIP_1005_2006-04-07.html.
- Park, J.-S., Wade, T. L., & Sweet, S. (2001). "Atmospheric distribution of polycyclic aromatic hydrocarbons and deposition to Galveston Bay, Texas, USA", Atmospheric Environment, Vol. 35, No. 19, pp. 3241–3249. doi: 10.1016/s1352-2310(01)00080-2

- Parra, M., Elustondo, D., Bermejo, R., & Santamaría, J. (2008). "Exposure to volatile organic compounds (VOC) in public buses of Pamplona, Northern Spain", Science of The Total Environment, Vol. 404, No. 1, pp. 18–25. doi: 10.1016/j.scitotenv.2008.05.028
- Peñuelas, J., Asensio, D., Tholl, D., Wenke, K., Rosenkranz, M., Piechulla, B., & Schnitzler, J. (2014). "Biogenic volatile emissions from the soil", Plant, Cell & Environment, Vol. 37, No. 8, pp. 1866–1891. doi: 10.1111/pce.12340
- Petersen, L. W., Moldrup, P., El-Farhan, Y. H., Jacobsen, O. H., Yamaguchi, T., & Rolston, D. E. (1995). "The Effect of Moisture and Soil Texture on the Adsorption of Organic Vapors", Journal of Environmental Quality, Vol. 24, No. 4, pp. 752–759. doi: 10.2134/jeq1995.00472425002400040028x
- Piccot, S.D., Watson, J.J., & Jones, J.W. (1992). "A global inventory of volatile organic compounds emissions from anthropogenic sources", Journal of Geophysical Research: Atmospheres, Vol. 97, No. D9, pp. 9897–9912. doi: 10.1029/92jd00682
- Pratt, G., Herbrandson, C., Krause, M., Schmitt, C., Lippert, C., McMahon, C., & Ellickson, K. (2018). "Measurements of gas and particle polycyclic aromatic hydrocarbons (PAHs) in air at urban, rural and near-roadway sites", Atmospheric Environment, Vol. 179, pp. 268–278. doi: 10.1016/j.atmosenv.2018.02.035
- Purcaro, G., Navas, J. A., Guardiola, F., Conte, L. S., & Moret, S. (2006). "Polycyclic Aromatic Hydrocarbons in Frying Oils and Snacks", Journal of Food Protection, Vol. 69, No. 1, pp. 199–204. doi: 10.4315/0362-028x-69.1.199
- Rao, G., & Vejerano, E. P. (2018). "Partitioning of volatile organic compounds to aerosols: A review", Chemosphere, Vol. 212, pp. 282–296. doi: 10.1016/j.chemosphere.2018.08.073
- Rathbun, R. E. (2000). "Transport, Behavior, and Fate of Volatile Organic Compounds in Streams", Critical Reviews in Environmental Science and Technology, Vol. 30, No. 2, pp. 129–295. doi: 10.1080/10643380091184183
- Rauh, V. A., Horton, M. K., Miller, R. L., Whyatt, R. M., & Perera, F. (2010). "Neonatology and the Environment: Impact of Early Exposure to Airborne Environmental Toxicants on Infant and Child Neurodevelopment", NeoReviews, Vol. 11, pp. 363–369. <https://doi.org/10.1542/neo.11-7-e363>
- Reşitoğlu, I. A., Altinişik, K., & Keskin, A. (2014). "The pollutant emissions from diesel-engine vehicles and exhaust aftertreatment systems", Clean Technologies and Environmental Policy, Vol. 17, No. 1, pp. 15–27. doi: 10.1007/s10098-014-0793-9
- Riain, C. N., Mark, D., Davies, M., Harrison, R., & Byrne, M. (2003). "Averaging periods for indoor-outdoor ratios of pollution in naturally ventilated non-domestic buildings near a busy road", Atmospheric Environment, Vol. 37, No. 29, pp. 4121–4132. doi:10.1016/s1352-2310(03)00509-0
- Sabino, F. C., Pinto, J. P., Amador, I. R., Martins, L. D., & Solci, M. C. (2016). "Gas-Phase Polycyclic Aromatic Hydrocarbons in the Parking Lot Impacted by Light-Duty Vehicles Burning Gasoline and Ethanol Blends", Journal of the Brazilian Chemical Society, Vol. 27, No. 9, pp. 1551–1557. doi:10.5935/0103-5053.20160034
- Salthammer, T; Mentese, S; Marutzky, R. (2010). "Formaldehyde in the Indoor Environment", Chemical Reviews, Vol. 110, No. 4, pp. 2536–2572, doi: 10.1021/cr800399g
- Schauer, J. J., Kleeman, M. J., Cass, G. R., & Simoneit, B. R. T. (1999). "Measurement of Emissions from Air Pollution Sources. 2. C1 through C30 Organic Compounds from Medium Duty Diesel Trucks", Environmental Science & Technology, Vol. 33, No. 10, pp. 1578–1587. doi: 10.1021/es980081n
- Schwarzenbach René P., Gschwend, P. M., & Imboden, D. M. (2002). "Environmental Organic Chemistry" (2nd ed., pp.1197-1208). New York: Wiley.
- See, S. W., Balasubramanian, R., Yang, T. S., & Karthikeyan, S. (2006). "Assessing Exposure to Diesel Exhaust Particles: A Case Study", Journal of Toxicology and Environmental Health, Part A, Vol. 69, No. 21, pp. 1909–1925. doi:10.1080/15287390600751280

- Shen, H., Huang, Y., Wang, R., Zhu, D., Li, W., Shen, G., Tao, S. (2013). "Global Atmospheric Emissions of Polycyclic Aromatic Hydrocarbons from 1960 to 2008 and Future Predictions", Environmental Science & Technology, Vol. 47, No. 12, pp. 6415–6424. doi: 10.1021/es400857z
- Shih, Y.-H., & Wu, S.-C. (2005). "Distinctive Sorption Mechanisms Of Soil Organic Matter And Mineral Components As Elucidated By Organic Vapor Uptake Kinetics", Environmental Toxicology and Chemistry, Vol. 24, No. 11, pp. 2827-2832. doi: 10.1897/05-186r.1
- Slezakova, K., Castro, D., Begonha, A., Delerue-Matos, C., Alvim-Ferraz, M. D. C., Morais, S., & Pereira, M. D. C. (2011). "Air pollution from traffic emissions in Oporto, Portugal: Health and environmental implications", Microchemical Journal, Vol. 99, No. 1, pp. 51–59. doi: 10.1016/j.microc.2011.03.010
- Solà, M. (2013). "Forty years of Clar's aromatic π -sextet rule", Frontiers in Chemistry, Vol. 1. pp. 22. doi: 10.3389/fchem.2013.00022
- Standards, Productivity and Innovation Board. (2016). "Singapore Standard Code of practice for indoor air quality for air-conditioned buildings". SS 554:2016 (doi:ICS 13.040.20; 91.040.01)
- Steinemann, A. (2017). "Ten questions concerning air fresheners and indoor built environments", Building and Environment, Vol. 111, pp. 279–284. doi: 10.1016/j.buildenv.2016.11.009
- Subramanyam, V., Valsaraj, K., Thibodeaux, L., & Reible, D. (1994). "Gas-to-particle partitioning of polycyclic aromatic hydrocarbons in an urban atmosphere", Atmospheric Environment, Vol. 28, No. 19, pp. 3083–3091. doi: 10.1016/1352-2310(94)00137-a
- Suess, M. J. (1976). "The environmental load and cycle of polycyclic aromatic hydrocarbons", Science of The Total Environment, Vol. 6, No. 3, pp. 239–250. doi: 10.1016/0048-9697(76)90033-4
- Swenberg, J. A., Kerns. W. D., Mitchell, R. I., Gralla. E. J., Pavkov. K. L. (1980). "Induction of Squamous-Cell Carcinomas of the Rat Nasal Cavity by Inhalation Exposure to Formaldehyde Vapor", Cancer Res., Vol. 40, No. 9, pp. 3398-3402.
- Sydbom, A., Blomberg, A., Parnia, S., Stenfors, N., Sandström, T. and Dahlén, S. (2001). "Health effects of diesel exhaust emissions", European Respiratory Journal, Vol. 17, No. 4, pp. 733-746. doi: 10.1183/09031936.01.17407330
- Szabó, J., Nagy, A. S., & Erdős, J. (2015). "Ambient concentrations of PM₁₀, PM₁₀-bound polycyclic aromatic hydrocarbons and heavy metals in an urban site of Győr, Hungary", Air Quality, Atmosphere & Health, Vol. 8, No. 2, pp. 229-241. doi:10.1007/s11869-015-0318-7
- Tang, J., Chan, C., Wang, X., Chan, L., Sheng, G., & Fu, J. (2005). "Volatile organic compounds in a multi-storey shopping mall in guangzhou, South China", Atmospheric Environment, Vol. 39, No. 38, pp. 7374–7383. doi: 10.1016/j.atmosenv.2005.09.018
- Tavares, M., Pinto, J. P., Souza, A. L., Scarmínio, I. S., & Solci, M. C. (2004). "Emission of polycyclic aromatic hydrocarbons from diesel engine in a bus station, Londrina, Brazil", Atmospheric Environment, Vol. 38, No. 30, pp. 5039–5044. doi: 10.1016/j.atmosenv.2004.06.020
- Tong, Z., Chen, Y., Malkawi, A., Adamkiewicz, G., & Spengler, J. D. (2016). "Quantifying the impact of traffic-related air pollution on the indoor air quality of a naturally ventilated building", Environment International, Vol. 89-90, pp. 138-146. doi:10.1016/j.envint.2016.01.016
- U.S. EPA (U.S. Environmental Protection Agency). (2018). "NATA Frequent Questions". Retrieved from <https://www.epa.gov/national-air-toxics-assessment/nata-frequent-questions#risk2>
- U.S. EPA (U.S. Environmental Protection Agency). (2019). "Guidelines for Human Exposure Assessment". (EPA/100/B-19/001). Washington, D.C.: Risk Assessment Forum, U.S. EPA.
- United Nations. (2019). "World Population Prospects - Population Division". Retrieved from <http://population.un.org/wpp/Download/Standard/Population/>.
- Väänänen V, Hämeilä M, Kalliokoski P, Nykyri E, Heikkilä P. (2005). "Dermal Exposure to Polycyclic Aromatic Hydrocarbons among Road Pavers", The Annals of Occupational Hygiene, Vol. 49, No. 2, pp. 167-178. doi: 10.1093/annhyg/meh094
- Vallero, D. (2014). "Fundamentals of Air Pollution" (5th ed., pp. 215-246). San Diego, CA, USA: Elsevier Science. doi: 10.1016/C2012-0-01172-6

- Verma, P. K., Sah, D., Kumari, K. M., & Lakhani, A. (2017). "Atmospheric concentrations and gas–particle partitioning of polycyclic aromatic hydrocarbons (PAHs) and nitro-PAHs at Indo-Gangetic sites", *Environmental Science: Processes & Impacts*, Vol. 19, No. 8, pp.1051–1060. doi: 10.1039/c7em00168a
- Wallace L.A. (1993). "Chemistry and Analysis of Volatile Organic Compounds in the Environment" (pp.1-24). New Delhi, India: Springer Science+Business Media Dordrecht. doi: 10.1007/978-94-011-2152-1
- Watson, J. G., Chow, J. C., & Fujita, E. M. (2001). "Review of volatile organic compound source apportionment by chemical mass balance", *Atmospheric Environment*, Vol. 35, No. 9, pp. 1567–1584. doi: 10.1016/s1352-2310(00)00461-1
- Weschler, C. J., & Nazaroff, W. W. (2008). "Semivolatile organic compounds in indoor environments", *Atmospheric Environment*, Vol. 42, No. 40, pp. 9018-9040. doi:10.1016/j.atmosenv.2008.09.052
- Weschler, C. J., & Nazaroff, W. W. (2010). "SVOC partitioning between the gas phase and settled dust in-doors", *Atmospheric Environment*, Vol. 44, No. 30, pp. 3609–3620. <https://doi.org/10.1016/j.atmosenv.2010.06.029>
- Weschler, C. J., & Nazaroff, W. W. (2012). "SVOC exposure indoors: fresh look at dermal pathways", *Indoor Air*, Vol. 22, No. 5, pp. 356–377. <https://doi.org/10.1111/j.1600-0668.2012.00772.x>
- Winkle, M. R. V., & Scheff, P. A. (2001). "Volatile Organic Compounds, Polycyclic Aromatic Hydrocarbons and Elements in the Air of Ten Urban Homes", *Indoor Air*, Vol. 11, No. 1, pp. 49–64. doi: 10.1034/j.1600-0668.2001.011001049.x
- World Health Organization. (2010). "WHO guidelines for indoor air quality : selected pollutants". Retrieved from http://www.euro.who.int/__data/assets/pdf_file/0009/128169/e94535.pdf
- Yamaguchi, T., Nakajima, D., Ezoe, Y., Fujimaki, H., Shimada, Y., Kozawa, K., Goto, S. (2006). "Measurement of Volatile Organic Compounds (VOCs) in New Residential Buildings and VOCs Behavior over Time", *Journal of UOEH*, Vol. 28, No. 1, pp. 13–27. doi: 10.7888/juoeh.28.13
- Yang, C., Miao, G., Pi, Y., Xia, Q., Wu, J., Li, Z., & Xiao, J. (2019). "Abatement of various types of VOCs by adsorption/catalytic oxidation: A review", *Chemical Engineering Journal*, Vol. 370, pp. 1128–1153. doi: 10.1016/j.cej.2019.03.232
- Yang, F., Kaul, D., Wong, K. C., Westerdahl, D., Sun, L., Ho, K., Tian, L., Brimblecombe, P., & Ning, Z. (2015). "Heterogeneity of passenger exposure to air pollutants in public transport microenvironments". *Atmospheric Environment*, Vol. 109, pp. 42-51. doi:10.1016/j.atmosenv.2015.03.009
- Yu, C. H., Zhu, X., & Fan, Z.-H. (2014). "Spatial/Temporal Variations and Source Apportionment of VOCs Monitored at Community Scale in an Urban Area", *PLoS ONE*, Vol. 9, No. 4. doi: 10.1371/journal.pone.0095734
- Zeng, Q., Jeppesen, E., Gu, X., Mao, Z., & Chen, H. (2018). "Distribution, fate and risk assessment of PAHs in water and sediments from an aquaculture- and shipping-impacted subtropical lake, China", *Chemosphere*, Vol. 201, pp. 612–620. doi: 10.1016/j.chemosphere.2018.03.031
- Zhang, Y., & Tao, S. (2009). "Global atmospheric emission inventory of polycyclic aromatic hydrocarbons (PAHs) for 2004", *Atmospheric Environment*, Vol. 43, No.4, pp. 812–819. doi: 10.1016/j.atmosenv.2008.10.050
- Zhu, J., Hsu, C.-Y., Chou, W.-C., Chen, M.-J., Chen, J.-L., Yang, T.-T., Wu, Y.-S., & Chen, Y.-C. (2019). "PM_{2.5}- and PM₁₀-bound polycyclic aromatic hydrocarbons (PAHs) in the residential area near coal-fired power and steelmaking plants of Taichung City, Taiwan: In vitro-based health risk and source identification", *Science of The Total Environment*, Vol. 670, pp. 439–447. doi: 10.1016/j.scitotenv.2019.03.198
- Zuraimi, M., Roulet, C.-A., Tham, K., Sekhar, S., Cheong, K. D., Wong, N., & Lee, K. (2006). "A comparative study of VOCs in Singapore and European office buildings", *Building and Environment*, Vol. 41, No. 3, pp. 316–329. doi: 10.1016/j.buildenv.2005.01.028

9. CHAPTER 7 – APPENDIX

A. Instrument Settings

a. TD Settings for VOCs and PAHs

Compounds	Primary desorption	Secondary desorption
VOCs	Time: 30 min	Time: 10 min
	Temperature: 335°C	Cold trap temperature: 370°C
	Tube flow rate: 50 mL/min	Column flow rate: 1.5 mL/min
	Split flow rate: 50 mL/min	Column temperature: 220°C
	Cold trap temperature: 0°C	Split flow rate: 1.5 mL/min
PAHs	Time: 12 min	Time: 3 min
	Temperature: 320°C	Cold trap temperature: 380°C
	Tube flow rate: 50 mL/min	Column flow rate: 1.5mL/min
	Split flow rate: 50 mL/min	Column temperature: 220°C
	Cold trap temperature: 25°C	Split flow rate: 10 mL/min

b. GC-MS Settings for VOCs and PAHs

Compounds	Settings of GC	Settings of MS
VOCs	Column	MS Source
	: DB-5MS™, 30 m × 0.25 mm × 0.25 μm	: 250°C
	Carrier gas	Transfer line
	: Helium, 1.5 mL/min	: 320°C
	Oven temperature settings

PAHs	Column	MS Sources
	: DB-5MS™, 30 m × 0.25 mm × 0.25 μm	: 250°C
	Carrier gas	Transfer line
	: Helium, 3.0 mL/min	: 320°C
	Oven temperature settings

B. Method Detection Limits (MDLs)

VOCs	MDL ($\mu\text{g}/\text{m}^3$)	PAHs	MDL (ng/m^3)
Benzene	0.22	Fluorene	0.0509
Toluene	0.23	Phenanthrene	0.0100
Ethylbenzene	0.18	Anthracene	0.0194
Xylene (m,p)	0.19	Fluoranthene	0.0171
Xylene (o)	0.19	Pyrene	0.0066
Styrene	0.11	Benz[a]anthracene	0.0597
1,2-dichlorobenzene	0.18	Chrysene	0.0587
Naphthalene	0.13	Benzo[b]fluoranthene	0.0109
		Benzo[k]fluoranthene	0.0045
		Benzo[a]pyrene	0.0032

C. The total daily doses (D_{total}) of VOCs and PAHs

Age group	Mean	STD	MIN	5 th %	25 th %	50 th %	75 th %	95 th %	Max
Benzene									
Birth to <1 year	2.3028	9.11	0.0001	0.0227	0.1476	0.5038	1.6792	8.9296	806.0344
1 to <2 years	2.3023	8.30	0.0002	0.0248	0.1604	0.5405	1.7726	9.1958	1116.7748
2 to <3 years	1.8804	7.18	0.0001	0.0204	0.1279	0.4297	1.3957	7.2871	669.7816
3 to <6 years	1.4004	5.17	0.0001	0.0152	0.0954	0.3231	1.0402	5.5399	554.7222
6 to <11 years	1.0554	6.88	0.0001	0.0096	0.0594	0.2026	0.6909	3.9566	1342.2635
11 to <16 years	0.5734	2.29	0.0000	0.0061	0.0372	0.1254	0.4112	2.2346	176.6232
16 to <21 years	0.4736	1.78	0.0000	0.0052	0.0313	0.1038	0.3418	1.8649	153.9795
21 to <41 years	0.4506	1.83	0.0000	0.0050	0.0299	0.1008	0.3337	1.7721	256.2594
41 to <61 years	0.4713	1.80	0.0000	0.0052	0.0317	0.1049	0.3445	1.8294	204.5707
61 to <81 years	0.3856	1.36	0.0000	0.0045	0.0264	0.0888	0.2892	1.5078	86.5471
≥81 years	0.3934	1.63	0.0000	0.0048	0.0279	0.0906	0.2918	1.5392	255.9717
Toluene									
Birth to <1 year	7.6207	12.39	0.0048	0.3504	1.6461	3.9428	8.8469	26.3584	619.0476
1 to <2 years	7.7761	11.39	0.0073	0.3980	1.8195	4.2677	9.2414	26.6458	390.2046
2 to <3 years	6.2644	9.18	0.0046	0.3203	1.4924	3.4578	7.4286	21.1872	293.0913
3 to <6 years	4.6031	6.88	0.0018	0.2355	1.0640	2.4875	5.4458	15.7014	205.1424
6 to <11 years	3.5043	7.86	0.0013	0.1456	0.6534	1.5851	3.6616	12.1365	467.2332
11 to <16 years	1.9148	3.23	0.0004	0.0905	0.4030	0.9592	2.1711	6.7443	162.5423
16 to <21 years	1.5664	2.44	0.0013	0.0782	0.3422	0.8192	1.8296	5.4190	64.8349
21 to <41 years	1.5288	2.44	0.0012	0.0735	0.3310	0.7885	1.7536	5.2924	74.2757
41 to <61 years	1.5806	2.48	0.0014	0.0790	0.3538	0.8407	1.8414	5.4528	101.3225
61 to <81 years	1.2940	1.98	0.0014	0.0693	0.3031	0.7007	1.5271	4.4008	91.1843
≥81 years	1.3073	1.95	0.0008	0.0709	0.3065	0.7093	1.5494	4.4381	61.0368
Ethylbenzene									
Birth to <1 year	2.1564	4.62	0.0004	0.0636	0.3259	0.8746	2.2132	8.1003	264.0701
1 to <2 years	2.1848	4.38	0.0009	0.0692	0.3493	0.9275	2.3183	8.1268	209.1618

2 to <3 years	1.7732	3.63	0.0005	0.0558	0.2831	0.7418	1.8508	6.5531	151.8238
3 to <6 years	1.3116	2.67	0.0004	0.0419	0.2096	0.5517	1.3703	4.8436	103.6743
6 to <11 years	0.9816	2.68	0.0003	0.0265	0.1290	0.3520	0.9097	3.6192	160.8611
11 to <16 years	0.5412	1.25	0.0002	0.0168	0.0787	0.2105	0.5391	2.0200	70.4502
16 to <21 years	0.4524	1.03	0.0001	0.0143	0.0673	0.1786	0.4567	1.6881	63.7900
21 to <41 years	0.4312	0.91	0.0001	0.0133	0.0644	0.1718	0.4410	1.6323	39.5968
41 to <61 years	0.4420	0.92	0.0001	0.0148	0.0696	0.1838	0.4588	1.6516	44.0192
61 to <81 years	0.3654	0.72	0.0001	0.0130	0.0594	0.1527	0.3797	1.3832	36.5809
≥81 years	0.3652	0.73	0.0001	0.0132	0.0607	0.1566	0.3883	1.3566	30.8897

Xylene (m,p)

Birth to <1 year	2.1826	4.80	0.0007	0.0622	0.3125	0.8573	2.2150	8.1665	206.4788
1 to <2 years	2.2328	4.69	0.0007	0.0678	0.3392	0.9090	2.3004	8.3348	187.7514
2 to <3 years	1.8076	4.19	0.0007	0.0549	0.2798	0.7461	1.8776	6.6763	561.5042
3 to <6 years	1.3244	2.89	0.0005	0.0410	0.2016	0.5342	1.3555	4.9481	208.5887
6 to <11 years	1.0177	3.54	0.0002	0.0262	0.1274	0.3447	0.9054	3.7305	529.6466
11 to <16 years	0.5536	1.30	0.0003	0.0161	0.0774	0.2082	0.5415	2.0795	70.1305
16 to <21 years	0.4478	0.95	0.0002	0.0139	0.0658	0.1755	0.4508	1.6982	37.8309
21 to <41 years	0.4349	0.97	0.0003	0.0136	0.0641	0.1717	0.4392	1.6428	66.5071
41 to <61 years	0.4505	1.01	0.0001	0.0142	0.0676	0.1788	0.4575	1.6883	54.0064
61 to <81 years	0.3688	0.76	0.0002	0.0127	0.0578	0.1504	0.3804	1.3910	43.2401
≥81 years	0.3728	0.83	0.0001	0.0127	0.0587	0.1526	0.3821	1.3774	44.2314

Xylene (o)

Birth to <1 year	1.4852	2.91	0.0005	0.0518	0.2479	0.6451	1.5953	5.4736	149.1967
1 to <2 years	1.5330	2.81	0.0006	0.0562	0.2674	0.6861	1.6666	5.6541	114.0322
2 to <3 years	1.2188	2.22	0.0005	0.0453	0.2158	0.5564	1.3401	4.4762	95.9725
3 to <6 years	0.9042	1.70	0.0005	0.0347	0.1598	0.4025	0.9773	3.3078	83.0382
6 to <11 years	0.6887	1.86	0.0003	0.0210	0.0986	0.2581	0.6566	2.5257	139.7683
11 to <16 years	0.3772	0.78	0.0002	0.0138	0.0613	0.1572	0.3897	1.4034	41.9165
16 to <21 years	0.3076	0.60	0.0001	0.0117	0.0524	0.1345	0.3277	1.1233	32.8384
21 to <41 years	0.3008	0.60	0.0001	0.0112	0.0501	0.1285	0.3176	1.1143	41.2472
41 to <61 years	0.3111	0.62	0.0000	0.0119	0.0530	0.1352	0.3314	1.1378	40.3585
61 to <81 years	0.2566	0.51	0.0001	0.0105	0.0456	0.1147	0.2762	0.9385	55.8811
≥81 years	0.2555	0.50	0.0001	0.0108	0.0466	0.1166	0.2761	0.9262	46.7633

Styrene

Birth to <1 year	1.4179	3.98	0.0003	0.0260	0.1387	0.4168	1.2282	5.6222	359.2003
1 to <2 years	1.4810	4.31	0.0003	0.0278	0.1482	0.4467	1.2972	5.7887	264.0997
2 to <3 years	1.1548	3.65	0.0003	0.0228	0.1191	0.3581	1.0383	4.5204	561.1908
3 to <6 years	0.8713	2.86	0.0001	0.0173	0.0893	0.2617	0.7484	3.2842	250.1913
6 to <11 years	0.6546	2.57	0.0001	0.0111	0.0568	0.1694	0.5018	2.4922	325.2691
11 to <16 years	0.3614	1.59	0.0001	0.0074	0.0355	0.1042	0.3029	1.3878	376.1399
16 to <21 years	0.2974	0.84	0.0001	0.0063	0.0306	0.0880	0.2544	1.1480	39.0368
21 to <41 years	0.2951	0.95	0.0000	0.0059	0.0289	0.0843	0.2451	1.1365	63.1141
41 to <61 years	0.2981	0.89	0.0000	0.0063	0.0308	0.0880	0.2559	1.1915	112.8919
61 to <81 years	0.2476	0.72	0.0000	0.0056	0.0266	0.0758	0.2156	0.9549	53.3021
≥81 years	0.2447	0.70	0.0001	0.0057	0.0269	0.0763	0.2149	0.9398	54.1693

1,2-dichlorobenzene

Birth to <1 year	1.2674	3.90	0.0001	0.0237	0.1231	0.3691	1.0780	4.9443	388.9530
1 to <2 years	1.2912	3.66	0.0002	0.0249	0.1322	0.3908	1.1383	5.1209	310.3329
2 to <3 years	1.0402	3.14	0.0001	0.0206	0.1061	0.3152	0.9106	4.0293	320.2722
3 to <6 years	0.7739	2.72	0.0002	0.0157	0.0795	0.2319	0.6762	3.0033	522.7057
6 to <11 years	0.5872	2.62	0.0001	0.0102	0.0511	0.1509	0.4465	2.1945	344.2426
11 to <16 years	0.3261	1.18	0.0000	0.0066	0.0319	0.0931	0.2695	1.2444	118.7712
16 to <21 years	0.2632	0.84	0.0001	0.0057	0.0272	0.0775	0.2223	1.0152	69.7922
21 to <41 years	0.2569	0.80	0.0000	0.0055	0.0261	0.0755	0.2179	0.9836	61.5869
41 to <61 years	0.2659	0.84	0.0001	0.0057	0.0276	0.0787	0.2278	1.0213	58.7700
61 to <81 years	0.2193	0.67	0.0000	0.0051	0.0238	0.0674	0.1910	0.8492	69.6133
≥81 years	0.2222	0.67	0.0000	0.0054	0.0244	0.0684	0.1933	0.8462	44.6998
Naphthalene									
Birth to <1 year	1.7929	3.87	0.0012	0.0788	0.3089	0.7635	1.8475	6.5512	403.6123
1 to <2 years	1.8147	3.71	0.0008	0.0827	0.3262	0.7941	1.9088	6.5972	277.1926
2 to <3 years	1.4752	2.96	0.0012	0.0705	0.2707	0.6533	1.5645	5.3907	236.7592
3 to <6 years	1.1107	2.42	0.0008	0.0537	0.2032	0.4864	1.1503	4.0370	296.4535
6 to <11 years	0.8560	2.46	0.0006	0.0359	0.1355	0.3278	0.8034	3.0347	201.5156
11 to <16 years	0.4709	1.05	0.0004	0.0243	0.0883	0.2074	0.4854	1.6879	91.9906
16 to <21 years	0.3902	0.80	0.0002	0.0212	0.0763	0.1777	0.4102	1.3862	56.2657
21 to <41 years	0.3797	0.79	0.0001	0.0203	0.0725	0.1695	0.3928	1.3593	63.4546
41 to <61 years	0.3854	0.78	0.0003	0.0209	0.0758	0.1751	0.4059	1.3602	51.9263
61 to <81 years	0.3292	0.66	0.0002	0.0187	0.0666	0.1538	0.3485	1.1597	33.2746
≥81 years	0.3252	0.59	0.0002	0.0194	0.0689	0.1555	0.3488	1.1479	20.7261
Fluorene									
Birth to <1 year	2.4375	5.76	0.0003	0.1060	0.4162	1.0033	2.4347	8.8891	385.8175
1 to <2 years	2.4486	5.82	0.0004	0.1105	0.4216	1.0131	2.4296	8.8845	516.2000
2 to <3 years	2.0200	4.53	0.0005	0.0955	0.3587	0.8511	2.0252	7.3301	293.4502
3 to <6 years	1.5928	3.65	0.0007	0.0749	0.2812	0.6734	1.6076	5.6372	274.8941
6 to <11 years	1.3227	4.74	0.0004	0.0523	0.2048	0.5024	1.2321	4.7042	823.8825
11 to <16 years	0.8109	2.00	0.0004	0.0366	0.1399	0.3342	0.8070	2.9064	205.8896
16 to <21 years	0.6844	1.58	0.0004	0.0326	0.1221	0.2924	0.6969	2.4546	162.2629
21 to <41 years	0.6586	1.43	0.0003	0.0309	0.1163	0.2816	0.6693	2.3473	63.8587
41 to <61 years	0.6601	1.47	0.0002	0.0322	0.1193	0.2846	0.6764	2.3627	111.6409
61 to <81 years	0.5923	1.27	0.0001	0.0291	0.1089	0.2592	0.6115	2.1160	77.2590
≥81 years	0.6024	1.28	0.0004	0.0302	0.1118	0.2669	0.6278	2.1339	77.0248
Phenanthrene									
Birth to <1 year	6.9553	152.36	0.0000	0.0386	0.2415	0.8555	3.1162	21.4158	45282.1408
1 to <2 years	6.0153	45.96	0.0003	0.0389	0.2413	0.8564	3.0814	20.7049	6749.3283
2 to <3 years	5.1177	34.32	0.0003	0.0338	0.2109	0.7456	2.6736	17.9005	4335.9375
3 to <6 years	4.5788	37.57	0.0001	0.0269	0.1715	0.6133	2.2050	15.2822	6892.2170
6 to <11 years	3.8494	30.98	0.0001	0.0197	0.1271	0.4651	1.7370	12.4036	4198.8788
11 to <16 years	2.7044	40.24	0.0001	0.0140	0.0884	0.3203	1.1899	8.4402	11161.2361
16 to <21 years	2.3171	24.54	0.0000	0.0121	0.0778	0.2819	1.0415	7.3325	2906.7104
21 to <41 years	2.1186	17.99	0.0001	0.0117	0.0733	0.2636	0.9814	6.9235	2345.5536
41 to <61 years	1.9878	14.62	0.0000	0.0117	0.0750	0.2706	0.9931	6.9143	2622.3109
61 to <81 years	1.9734	20.03	0.0001	0.0109	0.0700	0.2519	0.9205	6.3374	4774.7240

≥81 years	2.0323	18.03	0.0000	0.0113	0.0721	0.2586	0.9535	6.6949	3412.2251
Anthracene									
Birth to <1 year	2.6909	19.58	0.0001	0.0173	0.1088	0.3806	1.3728	8.9466	2103.2687
1 to <2 years	2.6133	18.86	0.0000	0.0176	0.1073	0.3750	1.3304	8.9370	2428.1965
2 to <3 years	2.1913	14.72	0.0001	0.0151	0.0928	0.3235	1.1606	7.6117	1864.6141
3 to <6 years	1.7997	12.16	0.0001	0.0121	0.0762	0.2675	0.9625	6.3737	1395.6028
6 to <11 years	1.5500	14.99	0.0001	0.0087	0.0552	0.2004	0.7338	4.9765	3165.6452
11 to <16 years	1.0406	10.64	0.0000	0.0062	0.0389	0.1377	0.5007	3.3950	2185.0586
16 to <21 years	0.8924	7.68	0.0000	0.0055	0.0338	0.1209	0.4391	3.0016	989.2372
21 to <41 years	0.8416	5.73	0.0000	0.0052	0.0327	0.1165	0.4194	2.8498	508.4600
41 to <61 years	0.8105	5.43	0.0000	0.0053	0.0331	0.1164	0.4160	2.7884	642.7660
61 to <81 years	0.7470	4.39	0.0000	0.0049	0.0302	0.1063	0.3797	2.5553	303.2304
≥81 years	0.7858	5.79	0.0000	0.0050	0.0313	0.1106	0.3962	2.6797	748.3579
Fluoranthene									
Birth to <1 year	5.6402	15.34	0.0007	0.2041	0.8180	2.0647	5.2429	20.8446	1004.6962
1 to <2 years	5.1535	14.47	0.0019	0.1998	0.7914	1.9554	4.8613	18.6341	1809.1593
2 to <3 years	4.7033	12.55	0.0024	0.1785	0.7027	1.7527	4.3868	17.2504	737.8049
3 to <6 years	4.3091	14.54	0.0008	0.1439	0.5915	1.5026	3.8680	15.7224	2059.7994
6 to <11 years	4.2096	16.49	0.0010	0.1058	0.4475	1.2104	3.3260	15.2104	1956.1403
11 to <16 years	3.0041	11.36	0.0008	0.0739	0.3267	0.8929	2.4388	11.0730	1304.8252
16 to <21 years	2.5894	8.12	0.0009	0.0663	0.2900	0.7957	2.1739	9.6949	534.5604
21 to <41 years	2.4557	7.91	0.0008	0.0640	0.2785	0.7562	2.0844	9.3742	879.4854
41 to <61 years	2.3836	7.22	0.0006	0.0647	0.2767	0.7454	2.0280	8.9735	468.9698
61 to <81 years	2.2959	7.11	0.0005	0.0610	0.2649	0.7171	1.9849	8.8523	752.7299
≥81 years	2.4412	8.25	0.0001	0.0638	0.2762	0.7480	2.0582	9.3266	1079.8113
Pyrene									
Birth to <1 year	14.5200	200.17	0.0002	0.0468	0.3371	1.3353	5.4751	44.3505	48504.3649
1 to <2 years	13.1538	131.73	0.0002	0.0466	0.3300	1.2990	5.2347	41.2397	23656.4777
2 to <3 years	13.2996	207.76	0.0001	0.0394	0.2891	1.1333	4.6035	37.8020	37988.4714
3 to <6 years	11.7997	287.92	0.0002	0.0319	0.2380	0.9409	3.8925	31.7393	79227.1410
6 to <11 years	10.6060	111.92	0.0000	0.0251	0.1841	0.7550	3.2204	28.5090	12832.8264
11 to <16 years	8.0046	187.58	0.0000	0.0170	0.1297	0.5309	2.2864	21.2098	51398.9242
16 to <21 years	6.0785	58.34	0.0000	0.0157	0.1150	0.4726	1.9915	17.4126	5830.6738
21 to <41 years	5.9285	90.01	0.0001	0.0147	0.1104	0.4508	1.9120	17.2003	22605.9039
41 to <61 years	5.9054	79.93	0.0001	0.0147	0.1104	0.4532	1.9188	16.6493	10557.7445
61 to <81 years	5.3362	49.08	0.0000	0.0139	0.1022	0.4227	1.8030	16.3887	6376.9837
≥81 years	6.9234	133.65	0.0000	0.0145	0.1084	0.4437	1.8913	17.1867	26632.6945
Benz[a]anthracene									
Birth to <1 year	23.5559	200.92	0.0001	0.0736	0.5371	2.1435	8.7805	72.1213	25402.4160
1 to <2 years	21.1357	319.37	0.0004	0.0746	0.5259	2.0550	8.1805	65.2882	86967.8346
2 to <3 years	20.0743	232.43	0.0002	0.0640	0.4637	1.8329	7.4809	62.1379	40241.4489
3 to <6 years	18.4826	274.23	0.0001	0.0541	0.3885	1.5627	6.3844	53.4452	55009.7271
6 to <11 years	17.8492	229.69	0.0001	0.0392	0.2984	1.2434	5.3941	48.5543	41752.4159
11 to <16 years	12.7876	227.84	0.0001	0.0285	0.2158	0.8840	3.8230	35.1306	61466.7808
16 to <21 years	11.1642	149.08	0.0000	0.0250	0.1897	0.7931	3.4789	31.9777	32981.1086
21 to <41 years	9.8437	90.18	0.0001	0.0240	0.1814	0.7594	3.2502	29.7732	8602.2760

41 to <61 years	9.7722	86.66	0.0000	0.0244	0.1848	0.7517	3.2132	29.9468	9098.1779
61 to <81 years	10.7360	187.13	0.0001	0.0233	0.1695	0.7093	3.0450	28.9878	40770.1113
≥81 years	11.4183	261.98	0.0000	0.0237	0.1777	0.7445	3.2535	30.0887	74296.5235
Chrysene									
Birth to <1 year	25.5686	103.05	0.0034	0.4524	2.1951	6.3596	18.8465	95.7754	8235.2053
1 to <2 years	23.1009	118.52	0.0016	0.4609	2.1325	6.0512	17.4570	85.1227	25657.6705
2 to <3 years	21.0919	87.04	0.0035	0.4030	1.8934	5.4246	15.7101	78.7906	8027.3765
3 to <6 years	19.8869	84.20	0.0011	0.3316	1.6042	4.7033	14.0922	73.8354	7962.2814
6 to <11 years	19.3333	130.05	0.0011	0.2463	1.2231	3.7957	12.0367	69.9156	23943.5673
11 to <16 years	13.9838	72.42	0.0010	0.1730	0.9068	2.8206	9.0023	51.9589	9352.2610
16 to <21 years	12.2954	56.28	0.0013	0.1537	0.7965	2.4978	7.9721	45.4125	3843.4879
21 to <41 years	11.9168	169.22	0.0013	0.1486	0.7778	2.3993	7.5362	42.1502	50689.2484
41 to <61 years	11.4186	58.90	0.0005	0.1526	0.7772	2.3563	7.4056	42.1353	5787.5132
61 to <81 years	10.8187	51.28	0.0012	0.1404	0.7310	2.2602	7.1967	40.0483	5769.0266
≥81 years	11.6609	67.83	0.0010	0.1499	0.7671	2.3784	7.4880	42.4618	8725.6014
Benzo[b]fluoranthene									
Birth to <1 year	9.1433	25.92	0.0023	0.3232	1.2893	3.2650	8.3257	34.0760	2452.7848
1 to <2 years	8.0447	20.10	0.0034	0.3178	1.2406	3.0716	7.6825	29.5237	1487.2989
2 to <3 years	7.5596	20.02	0.0029	0.2777	1.1171	2.8012	7.0481	27.9674	1558.0188
3 to <6 years	6.9739	17.61	0.0016	0.2293	0.9495	2.4442	6.3921	26.6883	1132.9044
6 to <11 years	6.8370	24.62	0.0012	0.1655	0.7207	1.9582	5.4523	25.7233	1946.1806
11 to <16 years	5.0460	17.31	0.0012	0.1192	0.5395	1.4983	4.1864	19.4581	2201.7724
16 to <21 years	4.4084	13.74	0.0015	0.1069	0.4789	1.3434	3.7656	16.9787	1113.0040
21 to <41 years	4.1476	11.88	0.0008	0.1017	0.4576	1.2794	3.5726	16.0406	656.9982
41 to <61 years	4.0176	11.73	0.0013	0.1024	0.4589	1.2450	3.4701	15.4360	820.1703
61 to <81 years	3.9738	11.34	0.0022	0.0973	0.4444	1.2202	3.3979	15.3473	694.3546
≥81 years	4.1034	11.66	0.0008	0.0994	0.4548	1.2743	3.6032	16.0776	829.0606
Benzo[a]pyrene									
Birth to <1 year	6.7269	18.57	0.0021	0.2205	0.9052	2.3364	6.0581	25.2171	1332.5486
1 to <2 years	6.0620	15.95	0.0018	0.2175	0.8699	2.2165	5.6323	22.5936	1128.5483
2 to <3 years	5.6576	16.24	0.0012	0.1915	0.7797	1.9981	5.1256	20.9304	1821.6582
3 to <6 years	5.2169	14.06	0.0006	0.1575	0.6639	1.7453	4.6439	19.9217	783.7521
6 to <11 years	5.3602	26.74	0.0019	0.1140	0.5147	1.4321	4.0541	19.4243	4716.9759
11 to <16 years	3.7605	12.48	0.0012	0.0843	0.3845	1.0782	3.0874	14.3948	1315.2117
16 to <21 years	3.3333	10.39	0.0003	0.0727	0.3405	0.9648	2.7484	12.7246	555.8341
21 to <41 years	3.0900	10.74	0.0006	0.0716	0.3254	0.9194	2.6010	11.8950	1018.6782
41 to <61 years	2.9797	9.41	0.0004	0.0711	0.3238	0.9037	2.5499	11.4061	1165.7791
61 to <81 years	2.9170	8.62	0.0006	0.0670	0.3130	0.8735	2.4907	11.3179	623.2005
≥81 years	3.1551	11.04	0.0008	0.0705	0.3250	0.9262	2.6169	12.0532	1582.4551

D. Concentrations of VOCs and PAHs in Various Microenvironments

VOC ($\mu\text{g}/\text{m}^3$)	Singapore		United Kingdom ^c							Iran ^d	Spain ^e	Brazil ^f	Taiwan ^g
	Bus interchange (This study)		Residence							Bus interchanges (24)	Bus (106)	Shopping mall Basement car park (36)	Bus interchanges (36)
	Indoor (49) ^a	Bus park (26)	Coach station (12)	Train station (12)	Roadside (12)	Department store (8)	Restaurant (6)	Smoking (32)	Non- Smoking (32)				
Benzene	2.94 ^b (2.56)	2.62 (2.75)	20.0 (16.1)	46.5 (37.7)	49.6 (22.4)	10.5 (5.6)	22.7 (4.0)	16.3 (15.3)	11.5 (11.8)	260.13 (56.76)	2.18 (1.73)	54.14 (19.72)	6.13 (2.75)
Toluene	15.11 (15.95)	12.65 (14.06)	47.3 (33.8)	135.3 (117.1)	108.1 (50.3)	56.7 (29.2)	57.0 (20.2)	29.2 (17.4)	47.6 (22.0)	683.65 (125.79)	7.13 (4.84)	209.24 (49.78)	74.15 (70.05)
Ethylbenzene	3.56 (3.05)	3.57 (3.37)	3.8 (1.3)	7.4 (7.6)	12.4 (8.6)	3.4 (2.4)	6.2 (3.0)	1.9 (1.2)	2.7 (1.2)	554.09 (107.41)	1.15 (0.77)	45.87 (18.59)	5.77 (2.97)
Xylenes										1144.95 (158.64)			
Xylene (m,p)	3.62 (3.34)	4.17 (5.36)									1.84 (1.34)	89.46 (33.72)	10.72 (4.83)
Xylene (p)			3.5 (1.0)	6.9 (7.5)	11.6 (8.4)	3.1 (2.2)	5.6 (2.8)	1.4 (0.9)	2.4 (1.0)				
Xylene (m)			10.1 (3.7)	18.8 (18.7)	32.3 (21.2)	8.9 (6.4)	16.3 (8.2)	4.3 (2.9)	6.3 (3.1)				
Xylene (o)	2.55 (2.09)	2.73 (2.56)	3.5 (1.3)	7.5 (8.1)	13.2 (9.7)	3.5 (2.5)	6.0 (2.9)	1.4 (1.1)	2.5 (1.3)		1.29 (0.92)	29.47 (10.73)	9.28 (4.30)
Styrene	2.03 (1.77)	2.01 (1.84)	0.6 (0.4)	1.4 (0.9)	1.7 (1.5)	1.1 (0.5)	1.4 (1.1)	0.8 (0.8)	0.7 (0.7)				1.23 (0.70)
1,2-dichlorobenzene	1.90 (1.83)	1.44 (1.85)									0.74 (0.80)		
Naphthalene	2.69 (1.85)	1.75 (1.70)	1.1 (1.9)	1.6 (0.9)	12.1 (38.0)	0.7 (0.4)	0.9 (0.4)	0.5 (0.3)	1.0 (1.3)				2.08 (1.61)

a. The number of samples

b. Arithmetic mean value and standard deviation

c. Kim, Y. M. et al., (2001); d. Golkhorshidi, F. et al., (2019); e. Parra, M. et al., (2008); f. Castro, B. P. et al., (2015); g. Lin, C. et al., (2020)

gPAHs (ng/m ³)	Singapore		Brazil ^c	Brazil ^d	Finland ^e	Vietnam ^f		Portugal ^g
	Bus interchange (This study)		Bus interchange (14)	Indoor car park (8)	Bus garage (106)	Bus interchange (1) Roadside (9)		Childcare center (105)
	Indoor (32) ^a	Bus park (17)						
Fluorene	2.81 (3.27) ^b	4.39 (5.97)	140.2 (17.3)	44.5 (9.90)	64			5.60
Phenanthrene	2.51 (3.34)	1.18 (1.64)	348 (32.7)	25.0 (4.10)	405	300	150 (54)	9.60
Anthracene	1.23 (1.43)	0.79 (1.03)	31.1 (6.1)	21.5 (0.60)		96	15 (6.1)	0.15
Fluoranthene	1.68 (1.92)	1.43 (1.33)	37.9 (7.9)	38.2 (0.80)	1.3	180	36 (14)	0.52
Pyrene	2.60 (4.27)	4.98 (9.33)	39.8 (8.9)	2.40 (0.12)	0.3	270	65 (30)	0.72
Benz[a]anthracene	4.09 (5.21)	4.06 (4.76)	1.4 (0.3)			13	1.8 (0.78)	1.70
Chrysene	5.39 (4.44)	5.01 (4.32)	3.8 (0.8)			2.6	1.2 (0.40)	0.10
pPAHs - PM ₁₀ (ng/m ³)	Singapore		Costa Rica ^h	Taiwan ⁱ	Brazil ^j	Hungary ^k		
	Bus Interchange (This study)		City areas (523)	School (46)	Roadsides (22)	Ambient Air (280)		
	Indoor (9)	Bus park (9)						
Fluoranthene	0.16 (0.18)	0.17 (0.18)	0.15 (0.19)	0.23 (0.20)	0.20 (0.14)	1.78 (3.32)		
Pyrene	0.25 (0.26)	0.27 (0.26)	0.23 (0.27)	0.14 (0.08)	0.23 (0.17)	1.51 (2.56)		
Benz[a]anthracene	0.35 (0.33)	0.84 (0.33)	7.83 (11.84)	0.15 (0.06)	0.45 (0.62)	1.17 (2.05)		
Chrysene	0.55 (0.49)	0.42 (0.49)	0.21 (0.17)	0.21 (0.12)	0.58 (0.74)	1.20 (1.70)		
Benzo[b]fluoranthene	0.20 (0.20)	0.36 (0.20)	0.28 (0.35)	0.06 (0.04)	0.87 (1.05)			
Benzo[a]pyrene	0.52 (1.01)	0.43 (1.01)	0.20 (0.13)	0.26 (0.17)	1.23 (1.76)	1.01 (1.51)		
pPAHs - PM _{2.5} (ng/m ³)	Singapore		Singapore ^l	Lithuania ^m		Costa Rica ^h	Taiwan ⁱ	Portugal ^g
	Bus Interchange (This study)		Bus Interchange	Roadside	Roadside	City areas (86)	Elementary school (48)	Childcare Center (105)
	Indoor (8)	Bus park (8)						
Fluoranthene	0.12 (0.11)	0.16 (0.19)	0.76 (0.22)	7.6 (3.4)	3.5 (1.2)	0.08 (0.04)	0.24 (0.22)	0.37
Pyrene	0.17 (0.19)	0.17 (0.27)	1.95 (0.68)	9.5 (4.1)	3.9 (1.3)	0.16 (0.08)	0.14 (0.08)	0.29
Benz[a]anthracene	0.40 (0.41)	0.46 (0.31)	0.74 (0.13)	7.0 (3.0)	2.8 (1.1)	16.81 (12.79)	0.14 (0.06)	0.12
Chrysene	0.29 (0.31)	0.47 (0.43)	1.50 (0.47)	9.2 (4.1)	4.5 (1.7)	0.14 (0.07)	0.20 (0.12)	0.31
Benzo[b]fluoranthene	0.13 (0.17)	0.22 (0.20)	0.82 (0.13)	5.8 (2.0)	3.0 (0.7)	0.17 (0.11)	0.05 (0.02)	
Benzo[a]pyrene	0.10 (0.10)	0.41 (0.68)	1.46 (0.83)	6.2 (3.9)	3.2 (1.0)	0.20 (0.09)	0.24 (0.17)	0.36

a. The number of samples

b. Arithmetic mean value and standard deviation

c. Tavares, M. et al., (2004); d. Sabino, F. C. et al., (2016); e. Kuusimäki, L. et al., (2003); f. Kishida, M. et al., (2008); g. Oliveira, M. et al., (2014); h. Murillo, J. H. et al., (2017); i. Zhu, J. et al., (2019); j. Oliveira, R. L. et al., (2014); k. Szabó, J. et al., (2015); l. See, S. W. et al., (2006); m. Kliucininkas, L. et al., (2011)