INDOOR EXPOSURE TO ENDOCRINE DISRUPTING CHEMICALS AND ASTHMA IN SCHOOL-AGED CHILDREN: A CASE-CONTROL STUDY

İÇ MEKANDA ENDOKRİN BOZUCU KİMYASALLARA MARUZ KALMA VE OKUL ÇAĞINDAKİ ÇOCUKLARDA ASTIM: BİR VAKA KONTROL ÇALIŞMASI

PARISA BABAEI

PROF. DR. GÜLEN GÜLLÜ

Supervisor

Submitted to

Graduate School of Science and Engineering of Hacettepe University

as a Partial Fulfillment to the Requirements

for the Award of the Degree of Doctor of Philosophy

in Environmental Engineering

2024

ABSTRACT

INDOOR EXPOSURE TO ENDOCRINE DISRUPTING CHEMICALS AND ASTHMA IN SCHOOL-AGED CHILDREN: A CASE- CONTROL STUDY

Parisa BABAEI

Doctor of Philosophy, Department of Environmental Engineering

Supervisor: Prof. Dr. Gülen GÜLLÜ

June 2024, 197 pages

The use of synthetic chemicals in many industrial and agricultural areas causes environmental pollution on a large scale. Known as one of the most widely used synthetic chemicals, endocrine-disrupting chemicals (EDCs) are being released into the environment at an exponential rate due to increasing population, expanding industrial activity, and increasing agricultural practices. Persistent EDCs, which consist of pharmaceuticals, personal care products, pesticides, surfactants, and several industrial chemicals, have hydrophobic and lipophilic properties that enable them to accumulate in the human body. The effects of EDCs on the body are multifaceted and can affect a wide variety of systems and functions. The extent to which these effects occur depends on several variables, including the type of EDCs, the dose and duration of exposure, and the sensitivity of the individual. Since children spend most of their time indoors, they are largely exposed to these chemicals. For this reason, various diseases such as asthma are frequently encountered in young children. The factors that determine the emergence of asthma can be observed in a wide range, from genetics to lifestyle and environmental factors. Environmental factors stand out among the reasons for regional differences in asthma prevalence and recently increasing rates. However, the importance of these

factors is still not well known. This study examines the association between exposure to EDCs (Persistent organic pollutants (POPs), alkylphenols and their ethoxylated) in indoor dust and the development of school-aged children's asthma in Türkiye, Ankara. This research is the first study in the region, focusing on dust sample analyses from the homes of 110 case (asthmatic) and 130 control (non-asthmatic) children. By using several statistical analyses such as the Spearman correlation test, Kruskal-Wallis analysis of variance and the Positive matrix factorization (PMF) model, we gained a comprehensive understanding of the data, identifying significant differences and factors contributing to those differences. According to the results, 4-5-6 ring PAHs were linked with traffic emissions, likely due to proximity to the main street. 2 and 3-ring PAHs showed associations with different factors, including the heating system, and smoking at home. Household activities and proximity to main streets have been found to have an impact on PCB concentrations. Newly purchased electronics and children's beds produced of wood or plastic had the highest levels of PBDEs. The study also showed that the amount of alkylphenol ethoxylates in dust is highly influenced by household practices and living circumstances. In this study used logistic regression models to examine the impact of family lifestyle and environmental conditions and selected EDCs on the risk of asthma in children. When odd ratio values are examined, living on the first or lower floors, using wallpaper as wall covering, not having a separate kitchen and owning a pet, living in older houses, having more than 4 people at home, frying food two or more times a week, smoking at home, using new furniture, frequency of detergent use, and using wool in the child's bed may affect the development of asthma. The results of this study reveal that environmental factors affect the development and severity of asthma.

Keywords: House dust, Endocrine-disrupting chemicals, School-aged children, Asthma

ÖZET

İÇ MEKANDA ENDOKRİN BOZUCU KİMYASALLARA MARUZ KALMA VE OKUL ÇAĞINDAKİ ÇOCUKLARDA ASTIM: BİR VAKA KONTROL ÇALIŞMASI

Parisa BABAEI

Doktora, Çevre Mühendisliği Bölümü

Tez Danışmanı: Prof. Dr. Gülen GÜLLÜ

Haziran 2024, 197 sayfa

Sentetik kimyasalların birçok endüstriyel ve tarımsal alanda kullanılması büyük çapta çevre kirliliğine neden olmaktadır. En yaygın kullanılan sentetik kimyasallardan biri olan endokrin bozucu kimyasallar (EDC'ler), artan nüfus, genişleyen endüstriyel faaliyet ve artan tarımsal uygulamalar nedeniyle çevreye katlanarak artan bir oranda salınmaktadır. Farmasötikler, kişisel bakım ürünleri, pestisitler, yüzey aktif maddeler ve çeşitli endüstriyel kimyasallardan oluşan kalıcı EDC'ler, insan vücudunda biriken, hidrofobik ve lipofilik özelliklere sahiptir. EDC'lerin vücut üzerindeki etkileri çok yönlüdür ve çok çeşitli sistem ve işlevleri etkileyebilmektedir. Bu etkilerin ortaya çıkma derecesi, EDC'lerin türü, maruz kalma dozu ve süresi ve bireyin duyarlılığı dahil olmak üzere çeşitli değişkenlere bağlıdır. Çocuklar zamanlarının çoğunu kapalı mekanlarda geçirdikleri için bu kimyasallara büyük oranda maruz kalmaktadırlar. Bu nedenle küçük çocuklarda astım gibi çeşitli hastalıklara sıklıkla rastlanmaktadır. Astımın ortaya çıkışını belirleyen faktörler genetikten yaşam tarzına ve çevresel faktörlere kadar geniş bir yelpazede görülebilmektedir. Astım prevalansında bölgesel farklılıklar ve son zamanlarda artan oranların nedenleri arasında çevresel faktörler öne çıkmaktadır. Ancak bu faktörlerin önemi hala tam olarak bilinmemektedir. Bu çalışma, ev içi tozdaki

EDC'lere (Kalıcı organik kirleticiler (KOK'lar), alkilfenoller ve bunların etoksilatlıları) maruz kalma ile Türkiye, Ankara'da okul çağındaki çocuklarda astım gelişimi arasındaki ilişkiyi incelemektedir. Bu araştırma, 110 vaka (astımlı) ve 130 kontrol (astımlı olmayan) çocuğun evinden alınan toz örneklerinin analizlerine odaklanan bölgedeki ilk çalışmadır. Spearman korelasyon testi, Kruskal-Wallis varyans analizi ve Pozitif matris çarpanlara ayırma (PMF) modeli gibi çeşitli istatistiksel analizleri kullanarak, veriler hakkında kapsamlı bir anlayış elde edilerek, önemli farklılıkları belirlenmiştir ve bu farklılıklara katkıda bulunan faktörleri ortaya çıkarılmıştır. Sonuçlara göre, 4-5-6 halkalı PAH'lar muhtemelen ana caddeye yakınlık nedeniyle trafik emisyonlarıyla bağlantılı olduğu ortaya çıkmıştır. 2 ve 3 halkalı PAH'lar, ısıtma sistemi ve evde sigara içilmesi gibi farklı faktörlerle ilişkilendirilmiştir. Ev aktiviteleri ve ana caddeye yakınlığın PCB konsantrasyonları üzerinde etkisi olduğu bulunmuştur. Yeni satın alınan elektronik eşyalar ve ahşap veya plastikten üretilmiş çocuk yatakları en yüksek düzeyde PBDE'ye sahip olduğu gösterilmiştir. Çalışma ayrıca, tozdaki alkilfenol etoksilat miktarının ev içi uygulamalardan ve yaşam koşullarından oldukça etkilendiğini ortaya çıkarmıştır. Bu çalışmada aile yaşam tarzının, çevre koşullarının ve seçilen EDC'lerin çocuklarda astım riski üzerindeki etkisini incelemek için lojistik regresyon modelleri kullanılmıştır. Olasılık değerleri incelendiğinde, birinci veya alt katlarda yaşama, duvar kaplaması olarak duvar kağıdı kullanma, ayrı mutfağının olmaması ve evcil hayvan sahibi olmak, eski evlerde yaşama, evde 4'ten fazla kişinin bulunması, haftada iki veya daha fazla kez kızartma yapmak, evde sigara içilmesi, yeni mobilya kullanılması, deterjan kullanım sıklığı, çocuğun yatağında yün kullanılması astım gelişimini etkileyebilmektedir. Bu çalışmanın sonuçları, çevresel faktörlerin astımın gelişimini ve şiddetini etkilediğini ortaya koymaktadır.

Anahtar Kelimeler: Ev tozu, Endokrin bozucu kimyasallar, Okul çağındaki çocuklar, Astım.

ACKNOWLEGMENT

I would like to express my sincere gratitude to my advisor, Prof. Dr. Gülen GÜLLÜ, for enlightening my path with his valuable insights and comments at every stage of this study, and for always supporting and encouraging me. I am honoured and happy to have had the opportunity to work with her. Thank you very much for accompanying me on this long and difficult journey, and for always teaching me to persevere patiently.

I am also thankful to my thesis monitoring committee, Prof. Dr. Ümit Murat ŞAHİNER and Prof. Dr. F. Nur BARAN AKSAKAL, and thesis examining committee, Assoc. Prof. Dr. Mihriban CİVAN and Asst. Prof. Dr. Derya Deniz GENÇ TOKGÖZ for their valuable time and comments. Additionally, I would like to thank to Prof. Dr. Özge UYSAL SOYER and Prof. Dr. Bülent Enis ŞEKEREL for their guidance and valuable suggestions.

I would like to thank my dear friend Efsun Nikravan Madan for being with me through all the difficulties and always supporting me during all my Phd journey.

It is also my duty to express my gratitude to Tuğçe ÖZER, Ceren SÖNMEZ, Ozan KAYA, Kerem Mert DEMİRTAŞ and Merve AYDIN for helping me in the process of collecting house dust samples necessary for this study.

I would like to thank my valuable professors and friends at Hacettepe University, Department of Environmental Engineering for their support.

I am very thankful to The Scientific and Technological Research Council of Turkey (TUBITAK) for the financial support (117Y088 project).

Finally, I would like to express my sincere gratitude to my family, who have always supported me and are my greatest luck in this life, for the love, patience, and encouragement they have shown me at every moment. My dear father, mother, and brother, I am grateful to you.

TABLE OF CONTENTS

ABSTRACT i
ÖZET iii
ACKNOWLEGMENT iv
TABLE OF CONTENTS vi
LIST OF FIGURES ix
LISTS OF TABLES xi
LIST OF ABBREVIATIONS xiii
1 INTRODUCTION 1
2 THEORETICAL BACKGROUND 4
2.1 Indoor Air Quality 4
2.2 Household air pollution 7
2.3 Sources of Indoor Chemical Pollutants 8
2.4 Endocrine Disrupting Chemicals (EDCs) 8
2.5 Persistent Organic Pollutants (POPs) 11
2.5.1 Polycyclic Aromatic Hydrocarbons (PAHs) 13
2.5.2 Polychlorinated bisphenyls (PCBs) 15
2.5.3 Polybrominated Diphenyl Ethers (PBDE) 17
2.6 Alkylphenols (APs) and Alkylphenol Polyethoxylated (APEs) 20
2.7 Definition and Prevalence of Asthma 21
2.8 Risk Factors in the Development of Asthma 24
2.9 Asthma in children 26
2.10 Association between Endocrine Disruptors and Asthma 27
2.11 Association between Persistent organic pollutants and Asthma 28
2.12 Association between Alkylphenols (APs) and Alkylphenol Ethoxylates (APEs) and Asthma 30
2.13 Literature Review 31
2.14 Researches Gaps 35
2.15 Aim and Scop of this study 37
2.16 Significance and Limitations of this Study 37
3 MATERIALS AND METHODS 40

3.1 Study Population 40
3.2 Questionnaire survey 41
3.3 Dust sample collection 42
3.4 Chemicals, materials and sample analyses 42
3.4.1 Analysis of Persistent Organic Pollutants 42
3.4.1.1 Polycyclic Aromatic Hydrocarbons (PAHs) Analyzes: 42
3.4.1.2 Polychlorinated Biphenyls (PCBs) Analyzes: 44
3.4.1.3 Polibromlu difenil eter (PBDEs) Analyzes: 46
3.4.1.4 Sample extraction and enrichment 47
3.4.1.5 Clean-up Procedures 48
3.4.2 Alkylphenol and alkylphenol ethoxylates analysis 50
3.4.2.1 Chemicals and reagents 50
3.4.2.2 Dust samples preparation 50
3.4.2.3 Instrumental analysis 51
3.4.3 Quality assurance and control 51
3.4.4 Statistical analysis 54
3.4.4.1 Positive matrix factorization (PMF) model 55
3.4.4.2 Binary Logistic Regression Model 57
4 RESULTS AND DISCUSSION 60
4.1 Evaluation of Survey Questions 60
4.2 Detection frequencies and levels of selected EDCs in indoor dust: 62
4.3 The levels and sources of PAHs in indoor dust 65
4.3.1 Distribute the Sources 69
4.3.1.1 Correlation analysis results between PAHs isomers 69
4.3.1.2 Diagnostic ratio 70
4.3.1.3 Evaluation of the relationships between home conditions and PAH analysis results (Kruskal-Wallis tests) 72
4.3.1.4 Positive matrix factorization (PMF) model 75
4.4 PCBs levels in indoor dust 76
4.4.1 Distribute the Sources 80
4.4.1.1 Correlation analysis results between PCB isomers 80

4.4.1.2 Evaluation of the relationships between home conditions and PCB analysis results (Kruskal-Wallis tests) 81
4.4.1.3 Positive Matrix Factorization (PMF) model: 84
4.5 The levels of PBDEs in indoor dust 85
4.5.1 Source apportionment of PBDEs 88
4.5.1.1 Correlation analysis results between PBDEs isomers 88
4.5.1.2 Evaluation of the relationships between home conditions and PCB analysis results (Kruskal-Wallis tests) 89
4.5.1.3 Positive Matrix Factorization (PMF) model 91
4.6 Alkylphenols, and alkylphenol ethoxylate 92
4.6.1 Influencing factors of Alkylphenols and alkylphenol ethoxylates in household dust 93
4.6.2 Source apportionment of dust 4-n-NP, di-NPE, and 4-t-OP via a logistic regression model 96
4.7 Effect of target pollutants known as endocrine disrupting chemicals on childhood asthma development 99
4.7.1 Characteristics of The Asthma Group 99
4.7.2 Results of a questionnaire survey 102
4.7.3 Comparison of pollutant concentrations between case and control groups: 104
4.7.4 Associations between exposure to typical EDCs and childhood asthma. 110
4.7.5 Association of pediatric asthma severity with exposure to common household dust chemicals 113
5 CONCLUSION 117
5.1 Recommendations for future studies 122
6 REFERENCES 124

LIST OF FIGURES

Figure 2-1 . Annual number of deaths by risk factors (2017 and 2019) 5
Figure 2-2. Death rate from indoor air pollution 6
Figure 2-3. Mechanism of EDCs 9
Figure 2-4. General chemical structure of PCBs. 15
Figure 2-5. General structural formula of PBDEs. 18
Figure 2-6. GAN Phase I: Severity of asthma symptoms by age group and country income level [86] 23
Figure 2-7. Asthma control among children, adolescents, and adults worldwide 24
Figure 2-8. Gaps in asthma care in low-and middle-income countries [90]. 25
Figure 2-9. Types of Persistent Organic Pollutants (POPs) included in the indoor studies and their percentages 33
Figure 3-1. Different regions of Ankara where house dust samples were collected 41
Figure 3-2. Flow chart of operation within EPA PMF- Base Model [180]. 57
Figure 4-1. Levels of 16 PAHs of indoor dust. 65
Figure 4-2. The profile of the analyzed PAHs 66
Figure 4-3. Comparison of PAH concentrations obtained in this study with data in the literature (ng/g) 67
Figure 4-4. Spearman rank correlation for PAH isomers 70
Figure 4-5. Source-determining PAH rates 72
Figure 4-6. Graphical distribution of PAH isomers resulting from home characteristics 74
Figure 4-7. Determination of PAH source in house dust using PMF model (Profile of factors) 76
Figure 4-8. Levels of PCBs in house dust 77
Figure 4-9. Contributions of homologous groups to total PCB concentrations in indoor dust 77
Figure 4-10. Comparison of total ∑ PCBs levels with other countries 78
Figure 4-11. Spearman rank correlation of PCBs isomers 81
Figure 4-12. Effect of proximity to the main street on PCB values 82
Figure 4-13. Effect of house cleaning frequency on PCB levels measured in house dust. 82

Figure 4-14. Effect of new furniture on PCB levels measured in house dust. 83
Figure 4-15. Effect of repairs or painting in the last year on PCB levels 84
Figure 4-16. Profile of factors for deternination source of PCBs isomers.....85
Figure 4-17. Leveles of PBDEs in house dust 85
Figure 4-18. Comparison of total PBDE levels with other countries.....86
Figure 4-19. Spearman rank correlation of PBDEs isomers 88
Figure 4-20. Effect of house age on PBDE levels.....89
Figure 4-21. Effect of new furniture purchased in the last year on PBDE values 90
Figure 4-22. Effect of child's bed material on PBDE values 91
Figure 4-23. PMF factor profiles of household dust of PBDEs. 92
Figure 4-24. Effect of home conditions and the lifestyle of the people living in the homes on the levels of APs and APE 96
Figure 4-25. Relationships between 4- NP and di- NPE and building conditions, family-related information, and lifestyle behaviors. 99
Figure 4-26. Distributions of target pollutants with a statistical difference between their median values in the case and control groups 108
Figure 4-27. Images taken around the houses in the control group 109
Figure 4-28. The results of examining the differences between the case and control groups of homologous and total isomers.....110
Figure 4-29. Odds ratio for risk of asthma in association with increase in household dust chemicals concentrations.....112
Figure 4-30. Significant odds ratios for childhood asthma by EDCs in indoor dust 115

LISTS OF TABLES

Table 2-1. Chemical structures of 16 PAH isomers (molecular weights, formulas, CAS numbers, and structural shapes) 14
Table 2-2. Chemical structures of 8 PCBs isomers (molecular weights, formulas, CAS numbers, and structural shapes) 16
Table 2-3. Chemical structures of 15 PBDEs isomers (molecular weights, formulas,CAS numbers, and structural shapes) 18
Table 3-1. Surrogate and standard solutions used for PAH analysis 43
Table 3-2. GC-MS operation parameters for PAH analysis 43
Table 3-3. GC-MS calibration parameters for PAH analysis 44
Table 3-4. Surrogate and standard solutions used for PCB analysis 44
Table 3-5. GC-MS operation parameters for PCB analysis 45
Table 3-6. GC-MS calibration parameters for PCB analysis 45
Table 3-7. Surrogate and standard solutions used for PBDE analysis 46
Table 3-8. GC-MS calibration parameters for PBDE analysis 47
Table 3-9. GC-MS calibration parameters for PBDE analysis 47
Table 3-10. MRM conditions and retention times for target chemicals 51
Table 3-11. Limits of Detection (ng/μl) and Method Detection Limits (ng/g d.w.) of target analytes 52
Table 3-12. Recoveries of the elements in the reference material (SRM2585) 53
Table 4-1. Characteristics of study population 60
Table 4-2 . The detection frequencies (DFs) and the levels of all targeted analytes 63
Table 4-3. Source-identifying PAH ratios 71
Table 4-4. Correlation of PAH isomers with home conditions 73
Table 4-5. APs and APE concentrations (in ng/g) in indoor dust were compared to those found in other investigations in the current study. 93
Table 4-6. Results of the binary logistic regression model to determine the relationship between APs and APE and building conditions and lifestyle behaviors. 98
Table 4-7. Characteristics of the asthma group 101
Table 4-8. Questionnaire survey (n_{Case} = 110, $n_{Control}$ =130) to identify childhood asthma risk factors 102

Table 4-9. Selected EDCs detection frequencies (DFs), concentrations (DFs), and intergroup variations in indoor dust 106
Table 4-10. Odds ratio (OR) of EDCs exposure on school-age children asthma 111
Table 4-11. Odds ratios for risk of severe asthma in association with EDCs concentrations in children with asthma 114

LIST OF ABBREVIATIONS

4-n-NP: 4-n-nonylphenol

4-t-OP: 4-tert-octylphenol

ABS: Acrylonitrile Butadiene Stien

ACT: Asthma Control Test

APEs: Alkylphenol Polyethoxylated

APs: Alkylphenols

BFRs: Brominated flame retardants

BPA: bisphenol A

CAS: Chemical Abstracts Service

CB: chlorinated biphenyl

CCCEH: Columbia Center for Children's Environmental Health

di-NPE: Nonylphenol diethoxylate

di-NPE: nonylphenol diethoxylate

EDCs: Endocrine-Disrupting Chemicals

EEDs: environmental endocrine disruptors

EPA: Environmental Protection Agency

ETS: environmental tobacco smoke

FEV1: Forced expiratory volume in 1 second

FVC: Forced vital capacity

GAN: Global Asthma Network

GINA: Global Initiative for Asthma

GSAMP: Asthma Management and Prevention

GSAMP: Global Strategy for Asthma Management and Prevention

HCB: hexachlorobenzene

IAQ: Indoor air quality

IPCS: International Programme on Chemical Safety

ISAAC: International Study of Asthma and Allergies in Children

IUPAC: International Union of Pure and Applied Chemistry

LPG: liquefied petroleum gas

NBFRs: novel brominated flame retardants

NCD: non-communicable disease

NP: Nonylphenol

NPEO: Nonylphenol Ethoxylates

OC: Octylphenol

OPFRs: organophosphate flame retardants

PACs: polyaromatic compounds

PAHs: Polycyclic aromatic hydrocarbons

PBDEs: polybrominated diphenyl ethers

PCBs: Polychlorinated biphenyls

POPs: Persistent organic pollutants

sVOCs: Semi-Volatile Organic Compounds

UNEP: United Nations Environment Program

WHO: World Health Organization

1 INTRODUCTION

Many healthcare providers are concerned about how environmental exposures contribute to respiratory diseases. While people generally recognize the importance of outdoor air quality to their health, they may overlook the potential harmful effects of indoor air pollution. The Environmental Protection Agency (EPA) oversees regulations for both outdoor and indoor air quality. According to the EPA, indoor pollutant levels can be up to 100 times higher than outdoor levels, ranking indoor air quality among the top 5 environmental risks to public health [1].

The past two decades have witnessed rapid modernization and urbanization worldwide, leading to significant advancements in people's daily lives. However, this progress has also brought about unprecedented environmental consequences, both indoors and outdoors, prompting increased public awareness and calls for enhanced living standards.

Building envelopes are intentionally designed to create a barrier between occupants and the outdoor environment. This design inadvertently facilitates the accumulation of indoor pollutants. These pollutants include particulate matter, inorganic gases, volatile organic compounds (VOCs), semi-volatile organic compounds (sVOCs), mold, radioactive compounds, and endocrine-disrupting chemicals (EDCs).

In this new age, children spend most of their time in closed environments such as schools and homes. This longer time spent indoors emphasizes how critical it is to comprehend and lessen the negative health effects of indoor air pollution. Exposure to complex chemicals significantly threatens human health, especially in vulnerable populations such as children. Children's immune systems are still developing; therefore, the quality of the air they breathe indoors is vital to their health. Asthma is a disease that occurs due to "chronic" exposure to various microbial agents or toxins and comes in attacks, in which genetic and environmental factors play an active role. Environmental allergens and irritants, such as indoor and outdoor air pollution, home dust mites, moulds, and contact with chemicals, fumes, or dust, are known to raise the risk of asthma. Asthma is a common non-communicable disease (NCD) in children and adults, affecting an estimated 262 million people and causing 461,000 deaths in 2019. There is a significant increase in asthma cases in young children worldwide, especially in developed and developing

countries [2–4]. In Türkiye, also asthma is the most frequent chronic health problem among children and adults. According to Türkiye 2020 Asthma Guidelines, there are approximately 4 million asthma patients in this country, and it is known that the prevalence of asthma ranges between 0.7-21.2% in children and 1.2-9.4% in adults. In this report, one in every 8 children and one in every 12 adults have asthma, and about 30% of doctor-diagnosed asthmatics reported that they were absent from school or hospitalized for asthma [5,6].

Many consumer product chemicals have been studied for exposure potential in the indoor environment. Some compounds found in building materials, furniture, and consumer products have been identified as EDCs, making them possible indoor pollutants, and suggesting that indoor exposure may contribute more to overall EDC exposure. This finding realized that changes in modifiable environmental exposures are likely to play a key part in the rising prevalence of non-communicable diseases like asthma and obesity among children in affluent nations, which cannot be attributed solely to genetic alterations [7]. Many high-volume compounds, including some that have already been identified as EDCs, have consumer usage (e.g., in plastics, detergents, and other household and consumer products), making them potential indoor pollutants.

Persistent organic pollutants (POPs) are among the most significant EDCs. Under the international treaty known as the Stockholm Convention, several POPs have been banned or restricted in numerous countries. Nevertheless, these pollutants continue to be of concern due to their persistence, ability to bioaccumulate, long-range transport, and adverse health effects [8,9]. Among them, polycyclic aromatic hydrocarbons (PAHs), polychlorinated biphenyls (PCBs), and polybrominated diphenyl ethers (PBDEs) are the most frequently found environmental pollutants. These pollutants' ubiquity, high to moderate volatility, and environmental persistence accumulate in organisms and various diseases.

PAHs, PCBs, PBDEs, alkylphenols, and alkylphenol ethoxylates, are some of the substances that will be examined in this thesis as potential EDCs. The purpose of the study is to shed light on the possible causes of morbidity associated with indoor air quality and how it affects children's asthma. Here, we present a representative case-control study within the realm of retrospective research, exploring the associations between typical EDCs such as POPs, alkylphenols, and alkylphenol ethoxylates exposure and childhood asthma.

The complex heterogeneous mixture known as house dust is made up of both biologically derived materials (such as hair and skin) and particulate matter from both indoor and outdoor sources (such as soil particles that have migrated or been tracked in from the outdoors and airborne particles that have infiltrated indoors). Particles in interior building spaces that have settled on furnishings, floors, carpets, and other surfaces are referred to as indoor settled dust. In fact, the number of organic contaminants found in indoor dust can be used as a reasonable proxy for assessing the extent of indoor pollution exposure.

In this thesis (case-control study), children aged 6-11 years were selected as the target population. Concentrations of three types of POPs, alkylphenols, and alkylphenol ethoxylates in house dust samples collected from the homes of both case (asthmatic) and control (non-asthmatic) children were measured. These measurements were then matched with data from a systematic questionnaire survey. Subsequently, the relationship between the amounts of these contaminants and the severity of asthma was investigated using statistical models.

2 THEORETICAL BACKGROUND

2.1 Indoor Air Quality

Human health is highly dependent on-air quality, and as people spend more time indoors, the quality of the air becomes increasingly important. Because studies have shown that indoor air quality (IAQ) is more contaminated than outdoor air, there has been a growing focus on the issues of poor IAQ among scientists and the public [10].

Indoor air quality is a global concern. Both short- and long-term exposure to indoor air pollution can lead to various health problems, including respiratory diseases, heart disease, cognitive impairments, and cancer. Pollution in the indoor environment can arise from two sources: pollution generated by substances and activities within the space, and pollution resulting from pollutants entering the space from the outside.

Indoor pollution is primarily attributed to factors such as inadequate ventilation, the absence of air conditioning systems, human activities, and various materials, chemicals, and gases. Recognizing IAQ as a multidisciplinary concern, organizations like the United States Environmental Protection Agency (US EPA) and the World Health Organization (WHO) have categorized pollutants into several classifications [11].

The Global Burden of Disease study is a comprehensive global research endeavor that examines the causes and risk factors contributing to mortality and morbidity worldwide. Study estimates of the annual number of deaths attributed to a wide range of risk factors are shown in Figure 2-1 . Indoor air pollution is a risk factor for some of the world's leading causes of death, including heart disease, pneumonia, stroke, diabetes, and lung cancer. The increase in the number of deaths caused by indoor air pollution between 2017 and 2019 underscores the urgent need for measures to address this issue and improve indoor air quality globally [12]. This number was estimated at 3.2 million annually in 2020 [13].

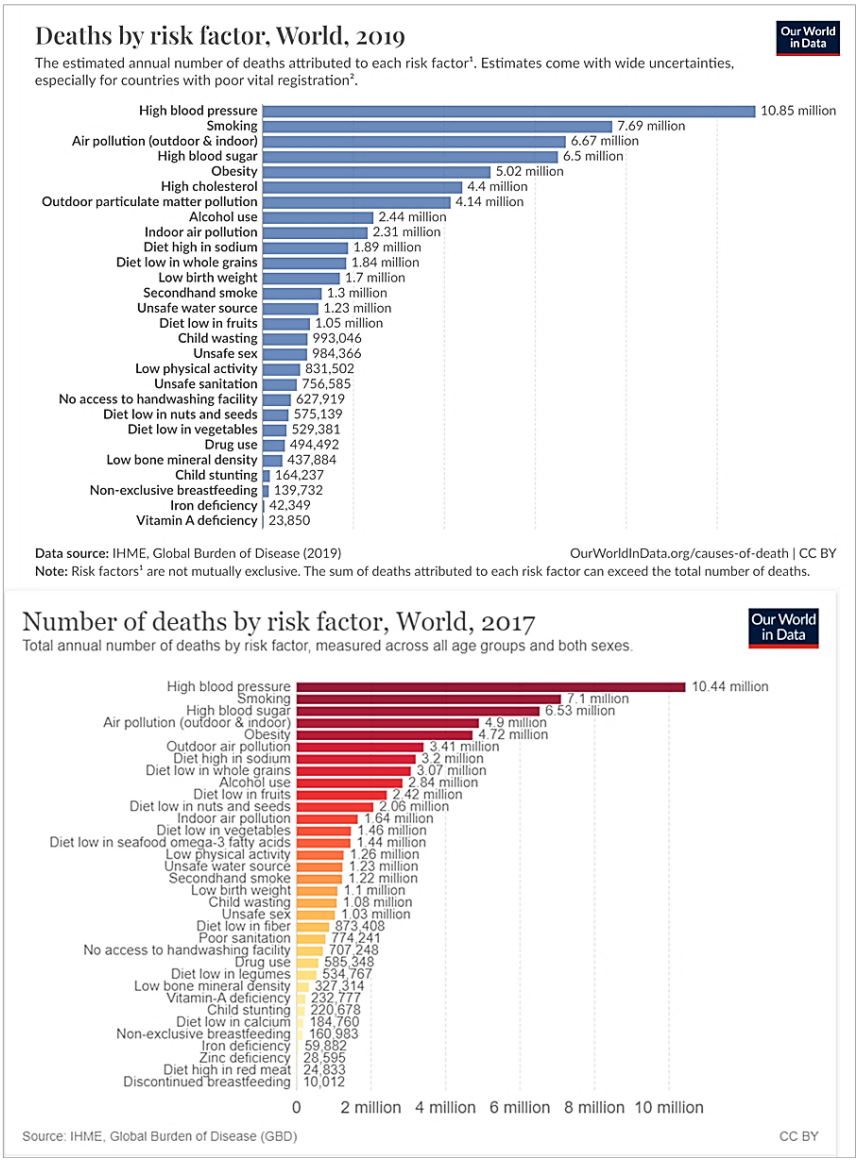


Figure 2-1 . Annual number of deaths by risk factors (2017 and 2019)

Visualizing death rates from indoor air pollution on a map provides a clear representation of differences in death rates attributed to this problem across different countries and regions. The map in Figure 2-2., with death rates measured as deaths per 100,000 people, gives an idea of the severity of the problem in different parts of the world. This visual aid helps policymakers and public health experts identify areas where intervention is most urgently needed to reduce the health impacts of indoor air pollution.

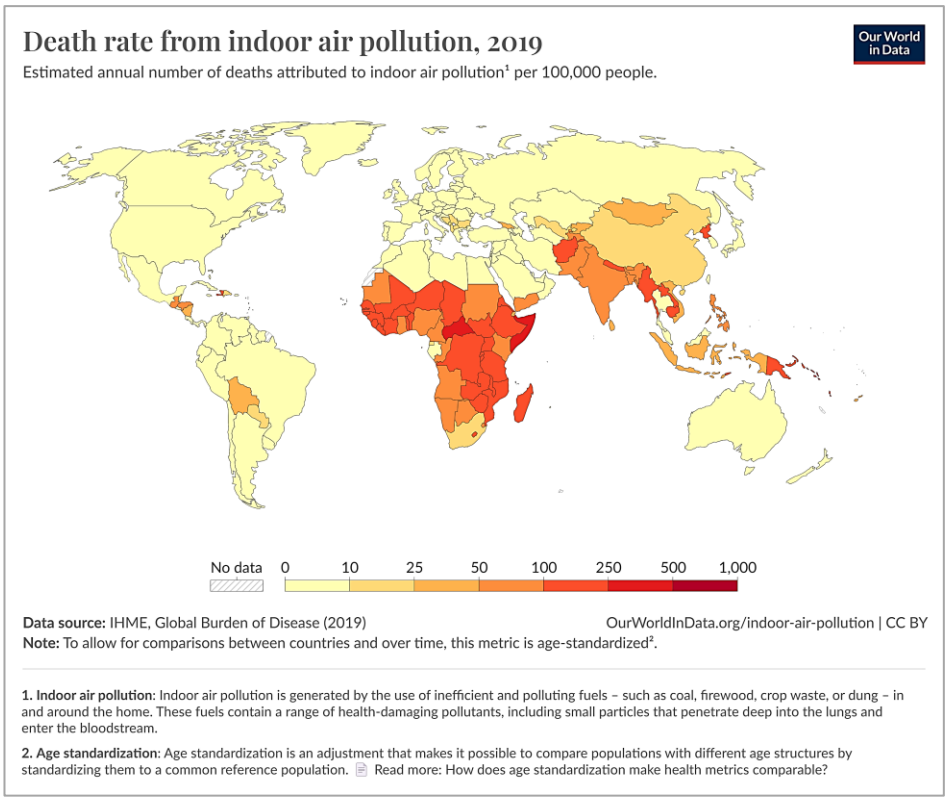


Figure 2-2. Death rate from indoor air pollution

What is clear is the large variation in death rates between countries: Rates are high in low-income countries, especially in sub-Saharan Africa and Asia. These rates are comparable to those in high-income nations; in North America, the rate is less than 0.1 deaths per 100,000 people. That is a discrepancy of more than a thousand.

Burning solid fuels for cooking and heating in homes, such as coal, charcoal, dung, and crop waste, is the main cause of indoor air pollution. Particulate matter, which is created when these fuels burn, poses a serious risk to one's health, especially for respiratory conditions. One of the main risk factors for these diseases getting worse is burning these fuels in small, enclosed areas like homes. Solid fuels are typically used for cooking in low-income households because cleaner fuels are either too expensive or unavailable. Thus, we observe a substantial correlation between the number of deaths caused by indoor air pollution and the availability of clean cooking fuels. This is illustrated in Figure 2-2, which shows that nations with relatively limited access to clean fuels—i.e., a significant reliance on solid fuels—have the highest death rates from indoor air pollution. To lower indoor air pollution and safeguard public health, it is imperative to increase the use of clean fuels and technology. These include renewable energy sources like solar and

electricity, as well as fuels like biogas, liquefied petroleum gas (LPG), natural gas, alcohol fuels, and biomass stoves that adhere to WHO emission guidelines [13].

2.2 Household air pollution

WHO estimates that household air pollution was responsible for an estimated 3.2 million deaths per year in 2020, including over 237 000 deaths of children under the age of 5 [13]. Fireplaces, kitchens, furniture, wall insulation, and personal care products are among the significant sources of household air pollutants. Household cleaning products have become particularly relevant as a source of indoor pollution, especially now when many people are cleaning more frequently and using stronger disinfectants to reduce the spread of viral infections. Efforts to enhance energy efficiency by making homes airtight have inadvertently led to reduced outdoor ventilation rates. Consequently, indoor pollutants accumulate to levels that would be considered harmful [14]. Crucially, indoor air toxicity goes beyond the impact of specific contaminants on health. Harmful indoor molecules such as hydroxyl radicals, oxygenated volatile organic compounds, and secondary ozonide can also originate from the reactive chemistry of pollution [15]. Ultimately, the indoor environment's pollutants are starting to be recognized as a significant factor in indoor air quality [16].

House dust is indeed a multifaceted amalgamation of various components originating from both indoor and outdoor sources. It encompasses a diverse array of materials such as biological debris like hair and skin particles, as well as particulate matter derived from indoor activities like heating, cooking, and smoking, alongside outdoor sources like soil particles and airborne pollutants that infiltrate indoor environments. Settled indoor dust comprises particles that have accumulated on surfaces, floors, carpets, and other objects within building interiors, reflecting the complex interplay between indoor and outdoor environments [17]. Indeed, indoor dust serves as a repository for numerous organic contaminants, making its contamination levels a useful proxy for evaluating indoor exposure to various pollutants [18]. Research indicates that the consumption of indoor dust tainted with organic pollutants contributes more to total exposure than diet, particularly for young children who play on the floor and put their hands in their mouths [19–22]. The use of inefficient and polluting fuels and technologies within and around the home produces household air pollution, which is made up of a variety of harmful pollutants to human health, including tiny particles that can enter the bloodstream and deeply infiltrate the lungs. In poorly ventilated dwellings, indoor air can have levels of

fine particles 100 times higher than acceptable. According to one study, up to 82% of the PBDE body burden in humans is thought to be caused by consuming household dust [23].

A multimodal strategy involving legislative support, community involvement, technological innovation, and international collaboration is needed to address household air pollution.

2.3 Sources of Indoor Chemical Pollutants

Chemicals in indoor air come from a variety of sources, including outdoor air. The building's location and leakiness determine the chemicals that are introduced to the outside air. For instance, homes closer to busy highways and/or with higher rates of ventilation and infiltration will have higher indoor concentrations of traffic-generated pollutants. There are also many indoor sources of reactive chemicals:

- Cleaning agents and air fresheners,
- Electronic equipment,
- Smoking,
- Combustion appliances, cooking, and heating,
- Home improvement measures such as painting,
- Building materials including wood, PVC pipes, and cable insulation,
- Furnishings including carpets, other floor coverings, and wall coverings,
- Pesticides,
- Personal care products,
- Pets,
- Bacteria and fungi.

2.4 Endocrine Disrupting Chemicals (EDCs)

Substances that can interact with the endocrine (or hormonal) systems are known as Endocrine Disrupting Chemicals. Scientific consensus currently favors the definition of EDCs put out by the International Programme on Chemical Safety (IPCS) and the World Health Organization (WHO) in 2002: “An endocrine disrupter is an exogenous substance or mixture that alters function(s) of the endocrine system and consequently causes adverse health effects in an intact organism, or its progeny or populations” [24].

The body's endocrine systems are widely involved in the short- and long-term control of metabolic activities. Endocrine systems play a complex role in controlling growth (including remodeling and growth of bones), gastrointestinal, cardiovascular, and kidney functions, as well as responses to stress in all its manifestations. Disease is an inevitable consequence of disorders affecting any of the endocrine systems, which involve both excessive and insufficient hormone secretion. The repercussions of these disorders can affect various organs and functions and are frequently fatal or severely debilitating. When considered in this broad context, environmental substances that have endocrine-disruptive or agonistic properties constitute a potentially serious concern. However, much depends on the level, length, and timing of exposure, so just because humans and wildlife are exposed to certain chemicals does not guarantee that the relevant endocrine system will be disturbed in a way that is clinically noticeable [25].

Substances known as EDCs interact with the endocrine system of the body, which controls hormone production. These substances have the ability to mimic hormones, block hormone receptors, or obstruct the synthesis, metabolism, or excretion of hormones (Figure 2-3. Mechanism of EDCs).

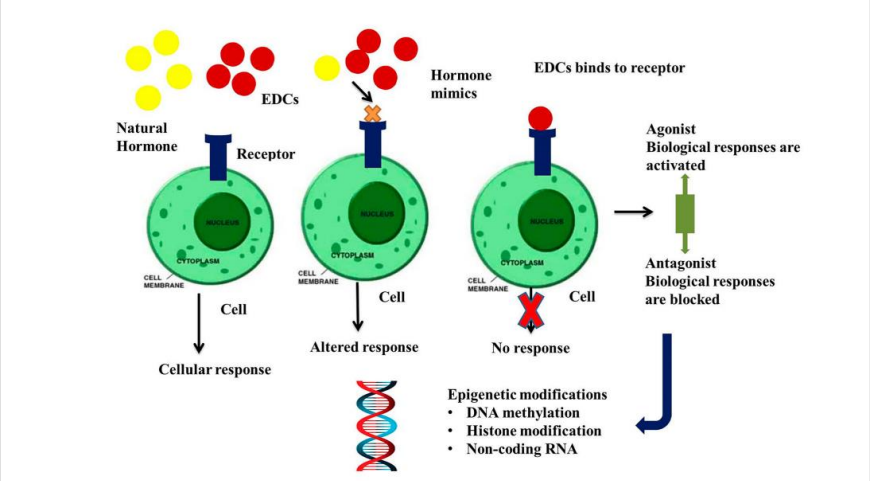


Figure 2-3. Mechanism of EDCs

Endocrine disruptors "interfere with the synthesis, secretion, transport, binding, action, or elimination of natural hormones in the body that are responsible for development, behavior, fertility, and maintenance of homeostasis (normal cell metabolism)" and are present in a wide range of household and industrial products [26,27].

EDCs are certain chemicals, both natural and man-made, that have been shown to interfere with hormone action and affect how the endocrine system functions.

In a 2012 report, WHO stated that humans may experience negative consequences from low-level exposures [28].

In cooperation with professionals worldwide, WHO is working on EDCs to [29]:

- Raise awareness and refresh your knowledge
- Promote studies on the impacts on children's health
- Create resources to lessen and prevent exposures in children and pregnant women.
- Create and maintain essential knowledge and training resources for healthcare professionals.

Natural and manufactured hormones, plant components, pesticides, chemicals used in consumer goods and the plastics industry, and other industrial by-products and pollutants are just a few of the chemical classes that are included in EDCs. They are frequently extensively distributed and ubiquitous in the surroundings. Of the approximately 85,000 compounds produced, about 6,000 are regarded as EDCs [30]. Pesticides, fungicides, metals, plasticizers, POPs, industrial chemicals, nonylphenols, pharmaceuticals, and phytoestrogens are all included in EDCs. EDCs are a broad category of substances that include manufactured goods, medications, detergent, plant-derived substances, and manufactured homes and consumables. Certain EDCs fall into more than one category:

- Phytoestrogens originating from plants - enhanced by legumes (genistein, daidzein), flaxseed, almonds, soy products, cereals, and breads;
- Industrial chemicals include pesticides, lubricants (PCBs), combustion products (PAHs, dioxins), and flame retardants (PBDEs);
- Personal care, household goods, and consumables: Benzophenone-3 and oxybenzone, bisphenol A (BPA), phthalates, perchlorate, dioxins, cosmetics, sunscreens, toys, food and beverage packaging materials, contaminated food, contaminated groundwater, tobacco goods, tea tree and lavender oils;
- Medical supplies: bags (made of BPA and phthalates), gloves, and intravenous tubing;
- Pharmacies: Synthetic and natural steroids (estradiol, diethylstilbestrol [DES])

The majority of EDCs have a relatively long half-life in the body because they are lipophilic and bioaccumulate in adipose tissue. The harmful effects of EDCs exposure

may manifest later in life or may not appear at all in some individuals, posing challenges in determining the complete impact of such exposure.

It is undeniable that endocrine disrupting chemicals cause serious health effects, emphasizing the critical need for additional research in this area.

This thesis investigates the dust levels of POPs and alkylphenols, known as EDCs, and examines their impact on indoor air quality and asthma severity.

2.5 Persistent Organic Pollutants (POPs)

Due to the lack of moisture, sunlight, and microbial activity, biotic and abiotic degradation occurs very slowly in indoor environments. As a result, dust-borne POPs are likely to persist much longer and present a long-term health risk. It is anticipated that surface cleaning or air exchange operations will be the main methods of removing semi-volatile organic compounds (sVOCs) from indoor space [31]. The two main subcategories of organic contaminants that are frequently found in household dust are (a) chemicals released from consumer products, machinery, and construction materials; these include flame retardants, such as polybrominated diphenyl ethers (PBDEs), novel brominated flame retardants (NBFRs), polychlorinated biphenyls (PCBs), and organophosphate flame retardants (OPFRs); (b) compounds released from occupant activities, such as polycyclic aromatic hydrocarbons (PAHs). Due to the fact that the majority of these pollutants are sVOCs, they are dispersed throughout the gas phase, airborne particles, settled dust, and surfaces that are accessible [32].

POPs can function as environmental endocrine disruptors (EEDs), potentially leading to various physiological toxicities, including immunotoxicity. This aspect has garnered significant attention because exposure to these chemicals is suspected to disrupt the immune system, possibly contributing to the development of certain difficult-to-treat diseases in humans [33,34].

POPs are primarily generated by industrial processes, either intentionally or as byproducts. Many of these chemicals are utilized in large quantities and added to a wide range of products, serving as flame retardants, lubricants, insulators, pesticides, and in various other applications [35].

PBDEs, PCBs, and PAHs are three groups of typical halogenated POPs to which people are commonly exposed in their daily lives.

PAHs are defined as those that can bioaccumulate, withstand environmental degradation, and travel over long distances by air. Due to their lipophilic nature, PAHs are easily absorbed through all routes of exposure and can enter the body quickly. The majority of their metabolites are hydroxylated and glucuronide-based PBDEs are primarily found in brominated flame retardants (BFRs), which are widely used in electronics, textiles, and furniture manufacturing [36]. Among the 209 congeners of PBDEs, deca-brominated diphenyl ether (BDE-209) is the most extensively used in China. Consequently, it has led to higher levels of exposure in both the environment and the human body [37]. PCBs are primarily utilized in electronics and building materials for their excellent lubrication and insulation properties [38]. In the indoor environment, these semi-volatile compounds have the potential to escape from household appliances' materials and subsequently be absorbed by humans.

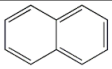
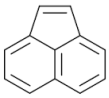
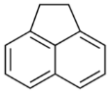
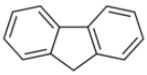
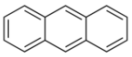
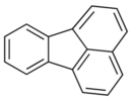
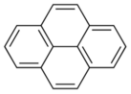
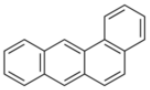
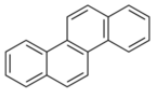
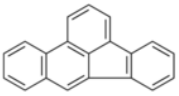
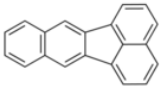
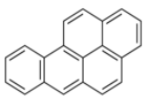
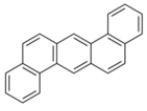
In May 2001, 12 POP species (aldrin, chlordane, DDT, dieldrin, endrin, heptachlor, hexachlorobenzene (HCB), Mirex, toxaphene, PCBs, PCDDs, and PCDFs) were initially prohibited for use through the Stockholm Convention. Thereafter, nine additional POPs were prohibited in May 2009 [39]. An international environmental pact called the Stockholm Convention on POPs aims to either completely eradicate or severely limit the use and production of persistent organic pollutants. On May 22, 2001, the treaty was adopted, and on May 17, 2004, it came into effect. The Stockholm Convention's main goal is to safeguard the environment and public health from POPs. These substances are well-known for their capacity for long-range environmental movement, their tenacity in the environment, and their capacity to bioaccumulate via the food chain [40]. The Stockholm Convention imposes obligations to eliminate or severely restrict the production and use of various POP pesticides and industrial chemicals. It also mandates strong measures to prevent or control the release of certain POPs formed as by-products of various combustions. However, despite these restrictions, POPs persist in the environment and the human body for decades. They are persistent and toxic, bioaccumulating in fatty tissue and reaching higher concentrations as they move up the food chain. Consequently, POPs continue to be found in various environmental samples such as water, soil, and air.

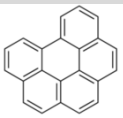
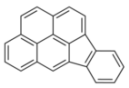
2.5.1 Polycyclic Aromatic Hydrocarbons (PAHs)

PAHs are a group of organic compounds characterized by the presence of two or more fused aromatic rings. These compounds can exhibit various shapes, including angular, linear, circular, or clustered arrangements. PAHs are known pollutants with high solubility in oils but low solubility in water. They have a propensity to adhere to particles with small diameters, such as dust, and are semi-volatile, meaning they can evaporate into the air and be transported through the environment via processes like evaporation and precipitation.

The scientific and public awareness regarding the environmental pollution and associated health risks linked to polyaromatic compounds (PACs), including PAHs and their derivatives, has grown significantly in both industrialized and developing nations. PAHs are sVOCs which are mostly formed during incomplete combustion of fossil fuels, biomass, home burning, power generation, and the pyrosynthesis of organic materials [41–43]. They are divided into two groups: anthropogenic and natural origin. PAHs have two main types of anthropogenic sources: pyrogenic (i.e., incomplete combustions of organic material, such as fuels, coal, wood, tobacco, and oil) and petrogenic (i.e., direct evaporation of petroleum products, such as gasoline and diesel fuel). When considering natural sources of PAHs, we can include events such as volcanic eruptions and forest fires, which release PAHs into the environment through the combustion of organic matter [44]. While lower-molecular-weight semivolatile PAHs, like pyrene, can originate from both petrogenic and pyrogenic sources, the higher molecular-weight nonvolatile PAHs containing 5 to 6 benzene rings are primarily produced by pyrogenic emissions. Semivolatile PAHs also have significant indoor sources, including activities such as space heating, cooking, smoking, or burning incense or candles. On the other hand, nonvolatile PAHs primarily originate from traffic emissions and heating oil combustion. The molecular weights, CAS numbers, closed formulas, and structural shapes of the 16 PAH isomers examined in this study are shown in Table 2-1.

Table 2-1. Chemical structures of 16 PAH isomers (molecular weights, formulas, CAS numbers, and structural shapes)

PAH isomers	Molecular weights (g/mol)	Formulas	CAS numbers	Structural shapes
Naftalin	128.17	C10H8	91-20-3	
Asenaftelen	152.19	C12H8	208-96-8	
Asenaften	154.21	C12H10	83-32-9	
Floren	166.22	C13H10	86-73-7	
Fenantren	178.23	C14H10	85-01-8	
Antrasen	178.23	C14H10	120-12-7	
Floranten	202.25	C16H10	206-44-0	
Piren	202.25	C16H10	129-00-0	
Benzo[a]antrasen	228.29	C18H12	56-55-3	
Krisen	228.29	C18H12	218-01-9	
Benzo[b]floranten	252.31	C20H12	205-99-2	
Benzo[k]floranten	252.31	C20H12	207-08-9	
Benzo[a]piren	252.31	C20H12	50-32-8	
Dibenzo[a,h]antrasen	278.35	C22H14	53-70-3	

PAH isomers	Molecular weights (g/mol)	Formulas	CAS numbers	Structural shapes
Benzo[g,h,i]perilen	276.33	C22H12	191-24-2	
indeno[1,2,3-cd]-piren	276.33	C22H12	193-39-5	

CAS, Chemical Abstracts Service

EPA has included 7 PAH isomers in the carcinogenic class: benzo[a]pyrene, benzo[a]anthracene, indenol[1,2,3-cd]pyrene, chrysene, benzo[b]fluoranthene, benzo[k]fluoranthene and dibenzo[a,h]anthracene [45].

2.5.2 Polychlorinated bisphenyls (PCBs)

A class of man-made organic compounds known as PCBs are biphenyl molecules with different numbers of chlorine atoms attached. PCBs are composed of two linked benzene rings in a biphenyl structure, with some or all of the hydrogen atoms replaced by chlorine atoms to create 209 distinct congeners. $C_{12}H(10\text{-}m\text{-}n)Cl(m+n)$ is the generic chemical formula, where (m + n) denotes the number of chlorine atoms on the two rings. The general chemical structure of PCBs is shown in Figure 2-4.

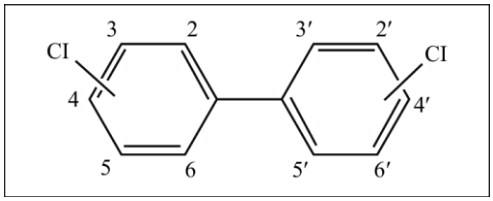
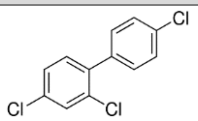
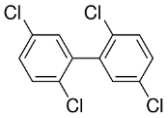
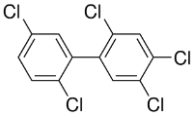
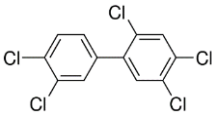
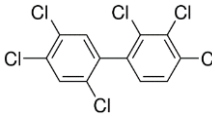
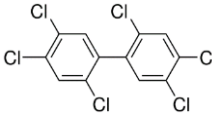
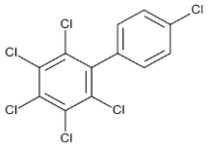
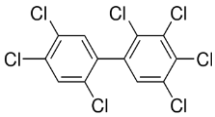


Figure 2-4. General chemical structure of PCBs.

The chemical properties and Chemical Abstracts Service (CAS) registry number of 10 PCBs isomers whose levels were examined in indoor dust are given in Table 2-2.

Table 2-2. Chemical structures of 8 PCBs isomers (molecular weights, formulas, CAS numbers, and structural shapes)

PCB isomers	IUPAC name	Formula	CAS No.	Molecular Weight	Structural shape
PCB 28	2,4,4′-TriCB	C12H7Cl3	7012-37-5	257.54	
PCB 52	2,2′,5,5′-TetraCB	C12H6Cl4	35693-99-3	291.99	
PCB 101	2,2′,4,5,5′-PentaCB	C12H5Cl5	37680-73-2	326.43	
PCB 118	2,3′,4,4′,5-PentaCB	C12H5Cl5	31508-00-6	326.43	
PCB 138	2,2′,3,4,4′,5′-HexaCB	C12H4Cl6	35065-28-2	360.88	
PCB 153	2,2′,4,4′,5,5′-HexaCB	C12H4Cl6	35065-27-1	360.88	
PCB 166	2,3,4,4′,5,6-HexaCB	C12H4Cl6	41411-63-6	360.88	
PCB 180	2,2′,3,4,4′,5,5′-HeptaCB	C12H3Cl7	35065-29-3	395.32	

CAS, Chemical Abstracts Service, CB, chlorinated biphenyl; IUPAC, International Union of Pure and Applied Chemistry

PCBs, are organic contaminants that are persistent and possess hazardous and carcinogenic properties. PCBs were manufactured commercially in the United States, mostly under the brand name Aroclor. These were congener combinations that were frequently discovered in oil emulsions [46].

Their stability, insulating qualities, and resilience to heat and fire made them useful in hydraulic systems, electrical equipment, and other industrial uses. However, PCBs are

also classified as POPs, which means that they can build up in ecosystems and species, including humans, and that they are not easily broken down in the environment.

PCBs are one of the 12 pollutants included in the POPs class by the United Nations Environment Program (UNEP). In May 2001, their use was initially prohibited through the Stockholm Convention [47]. Since the 2001, PCBs have been outlawed or severely limited in numerous nations due to their detrimental consequences. Even decades after their use has been stopped, they nevertheless remain a threat to the environment and public health because to their tenacity. Around the world, there is a continuous effort to clean up PCB-contaminated areas and lessen the ongoing effects of these pollutants on the environment and human health.

PCBs exhibit similar properties; they are highly resistant to acids, bases, and heat, which renders them excellent as insulators and lubricants. They were widely utilized as dielectric and heat-transfer fluids, plasticizers in paints and surface coatings, and flame retardants in transformers, capacitors, and consumer goods due to their non-flammability and electrical insulating qualities [48].

PCBs are present at measurable levels in all environmental media (soil and sediments, water, air) worldwide and also possibly in the body of every human being.

Persistent and lipophilic substances like PCBs naturally sink in dust where they are absorbed by the soil's organic carbon and become rather persistent after that. PCBs can infiltrate the soil and dust through a variety of processes, including air deposition, industrial emissions from manufacture, use, and disposal, and erosion and leachate from adjacent contaminated regions [49].

2.5.3 Polybrominated Diphenyl Ethers (PBDE)

A class of bioaccumulate and recalcitrant halogenated chemicals known as PBDEs has become a significant environmental contaminant. These are artificial substances that are added to a range of consumer and business goods to help them resist fire and flames.

To meet flame retardancy regulations set by various jurisdictions worldwide, a group of synthetic chemicals known as brominated flame retardants (BFRs) are added to a wide range of polymers, foam, plastic, textile, and building materials. BFRs include 50–85% bromine by weight [50].

PBDEs are a class of compounds that share the chemical formula C12H(0–9)Br(1–10)O and a common structure as brominated flame retardants (BFRs). There are 209 potential congeners when any one of the diphenyl ether moiety's 10 hydrogen atoms is swapped out for bromine [51]. In Figure 2-5, structure formula is shown.

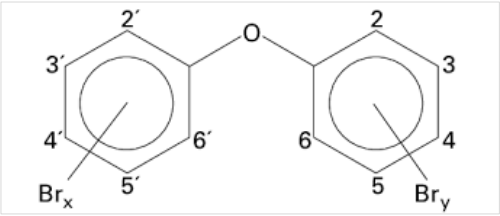
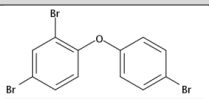
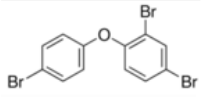
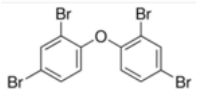
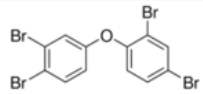
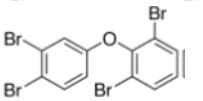
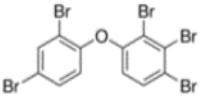
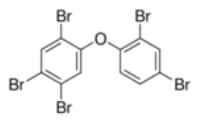
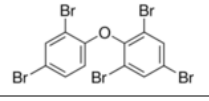
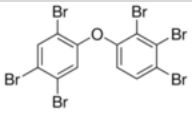
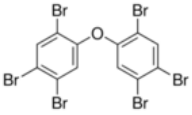
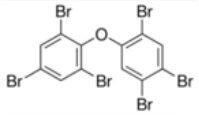
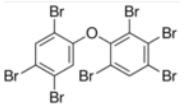
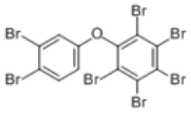
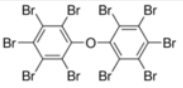


Figure 2-5. General structural formula of PBDEs.

The structural forms, CAS numbers, structural formulas, closed formulas and molecular weights of the 14 PBDEs isomers whose levels were examined in indoor dust are given in Table 2-3.

Table 2-3. Chemical structures of 15 PBDEs isomers (molecular weights, formulas,CAS numbers, and structural shapes)

PBDE isomers	IUPAC name	Molecular Weight (gr/mol)	CAS Number	Formula	Structural shape
PBDE 17	2,2′,4-tribromodiphenyl ether	436.92	147217-75-2	C12H7Br3O	
PBDE 28	2,4,4′-tribromodiphenyl ether	406.90	41318-75-6	C12H7Br3O	
PBDE 47	2,2′,4,4′-tetrabromodiphenyl ether	485.79	5436-43-1	C12H6Br4O	
PBDE 66	2,3′,4,4′-tetrabromodiphenyl ether	485.79	189084-61-5	C12H6Br4O	
PBDE 71	2,3',4',6-tetrabromodiphenyl ether	485.79	189084-62-6	C12H6Br4O	
PBDE 85	2,2′,3,4,4′-pentabromodiphenyl ether	564.69	182346-21-0	C12H5Br5O	
PBDE 99	2,2′,4,4′,5-pentabromodiphenyl ether	564.69	60348-60-9	C12H5Br5O	
PBDE 100	2,2′,4,4′,6-pentabromodiphenyl ether	564.69	189084-64-8	C12H5Br5O	

PBDE isomers	IUPAC name	Molecular Weight (gr/mol)	CAS Number	Formula	Structural shape
PBDE 138	2,2′,3,4,4′,5′-hexabromodiphenyl ether	643.58	182677-30-1	C12H4Br6O	
PBDE153	2,2′,4,4′,5,5′-hexabromodiphenyl ether	643.58	68631-49-2	C12H4Br6O	
PBDE 154	2,2′,4,4′,5,6′-hexabromodiphenyl ether	643.58	207122-15-4	C12H4Br6O	
PBDE 183	2,2′,3,4,4′,5′,6-heptabromodiphenyl ether	722.48	207122-16-5	C12H3Br7O	
PBDE 190	2,3,3',4,4',5,6-heptabromodiphenyl ether	436.92	189084-68-2	C12H3Br7O	
PBDE 209	decabromodiphenyl ether	406.90	1163-19-5	C12Br10O	

CAS, Chemical Abstracts Service, CB, chlorinated biphenyl; IUPAC, International Union of Pure and Applied Chemistry

Three technical mixes of penta-, octa-, and deca-BDEs are sold commercially. Each mixture has varying amounts of the brominated congeners rather than being a pure combination of penta-, octa-, or deca-congeners [52]. Polymer matrices are seeded with PBDEs, but they are not covalently attached to them. They eventually diffuse out of the polymer matrix, take to the air, and spread worldwide [53].

PBDEs have been discovered in sewage treatment plant biosolids, fish and other marine life, humans, animals, soil, sediments, and the air [54]. Both leaching from typical household goods and industrial manufacturing facilities releases them into the environment. Household waste, such as electronics, furniture, bedding, and foam cushions, is the primary non-point source of PBDEs [55].

PBDEs are widely used as flame retardants in various products. The rationale for using brominated compounds as flame retardants is based on the ability of halogen atoms, generated from the thermal decomposition of the Bromo organic compound, to chemically reduce and retard the development of fire. When these compounds are exposed to high temperatures, they release halogen atoms, which then interfere with the combustion process by capturing free radicals, thereby slowing down the chemical reactions that sustain the fire. Factors favoring the use of PBDEs include their high

bromine content, which provides excellent flame-retardant properties, thermal stability, and relatively low cost. These attributes make PBDEs effective in reducing the flammability of a wide range of materials used in consumer products, including electronics, textiles, and furnishings. Penta-BDEs were used in bed linens, curtains, upholstery fabrics, sofa cushions, carpets, polyurethane foams, paints used in the marine sector and industry, insulation panels, etc. The usage area of Octa-BDEs can be said as Acrylonitrile Butadiene Stien (ABS) plastics such as telephones, photocopy and fax machines, computers, televisions, kitchen tools and equipment, hair dryers. Deca-PBDEs were used as Polystyrenes (HIPS) such as electronic devices, upholstery textiles, polyethylenes, paints used in marine vehicles, paints used in industry, and poly carbonates [56].

Studies have shown that exposure to PBDEs is associated with various health problems such as: Cancer, Neurological and behavioral disorders, hyperthyroidism, reproductive anomalies, respiratory diseases, and Infertility [57].

2.6 Alkylphenols (APs) and Alkylphenol Polyethoxylated (APEs)

Environmental phenolic chemicals, due to their widespread occurrence and potent estrogenic properties, pose a risk to human exposure. The phenolic organic contaminants APs and APEs are used in various household applications, and they may enter the environment during production and use, potentially appearing in indoor dust. However, little is known about the levels of environmental phenolics in indoor environments. Alkylphenols, which are by-products of the microbial breakdown of APEs, are a prominent class of EDCs [58]. Octylphenol (OP) is a member of a wider family of alkylphenols, and it is used as an intermediate to produce phenol/formaldehyde resins, which are used in rubber, in pesticides and paints, as well as in manufacture of octylphenol ethoxylates. As a member of APE family, 4-Octylphenol Monoethoxylate (4-OPME) may be produced by degradation of non-ionic surfactants such as 4-tert-octylphenol polyethoxylate . It is also a non-steroidal estrogen [59]. 4-tert-octylphenol (4-t-OP) is an AP and is used to make surfactants used in detergents, industrial cleaners, and emulsifiers whereas it is classified as a category 1 endocrine disruptor [60]. Nonylphenols (NPs) and nonylphenol ethoxylates (NPEs) such as 4-n-nonylphenol (4-n-NP) and nonylphenol diethoxylate (di-NPE) are produced in large volumes and are used

for industrial processes and in consumer laundry detergents, personal hygiene, automotive, latex paints, and lawn care products [61]. The two alkylphenols that are most frequently found in the environment are 4-n-NP and 4-t-OP. These alkylphenols (4-n-NP and 4-tOP) are by-products of the biodegradation of nonylphenol polyethoxylates, and octylphenol polyethoxylates which account for approximately 80 % and 20 % of the total APEs, respectively [62]. Non-ionic surfactants, such as alkylphenols and alkylphenol polyethoxylates, have already been classified as EDCs and are used in plastics, detergents, paints, industrial applications, including paper production, textile processing, as well as other household and consumer goods. This makes them potentially significant indoor pollutants. Octylphenol, nonylphenol, and nonylphenol ethoxylates can enter the body through ingestion, inhalation, or direct contact with items applied to the skin [63–65]. Studies have shown that exposure to NPs at high levels can have negative effects. These side effects include hepatotoxic effects and antiandrogenic activity in addition to irritation of the lungs, digestive tract, eyes, and skin [66–68]. The 4-nonylphenol and 4-tert-octylphenol interfere with several biological systems, including hormone receptor proteins, hormone transport proteins, and hormone secretory enzymes, to change how hormones operate [69]. Nonylphenol and its ethoxylate homologs are strongly hydrophobic. Reported log Kow values for NP (Nonylphenol) and NPEO (Nonylphenol Ethoxylates) in the literature vary but are generally low, indicating their ready sorption to organic materials like dust in environmental matrices. Furthermore, the high Koa (octanol-air partition coefficient) values of these compounds suggest their potential for long-distance air transport and accumulation [70]. Since the degradation processes of these chemicals are typically slow, dust serves as a record of the substances historically used in a home. Consequently, house dust becomes an important medium and route of exposure to alkylphenols and alkylphenol polyethoxylates as sVOCs. Several studies have confirmed the presence of these compounds in settled dust [65,71–74]. Comparing the data from this study with those from other studies can provide insights into the geographic, demographic, and temporal patterns of exposure to these substances.

2.7 Definition and Prevalence of Asthma

Asthma, a chronic and diverse disease of the lower airways, is characterized by persistent inflammation and airway hyperreactivity, leading to symptoms such as coughing, wheezing, chest tightness, and difficulty breathing. The etiology of asthma is multifaceted [75]. Over the last thirty years, increased knowledge of the unique

characteristics of asthma (phenotypes) and mechanisms (endotypes) has led to the development of improved diagnostic and therapeutic instruments. These advancements enable tailored and stratified interventions based on individual differences in response to different treatments [76]. Furthermore, asthma development and heterogeneity in phenotyping, as well as steroid responsiveness, are influenced by genetic polymorphisms, environmental variables, and epigenetic factors. These factors play significant roles in shaping the individual variations observed in asthma manifestation and response to treatment [77–79]. Enhancing asthma control and minimizing exacerbations can be achieved through environmental interventions and effective exposure management [80]. To prevent side effects from oral corticosteroids (OCS) and to improve control of symptoms and exacerbations in patients with severe asthma, new treatments and therapeutic targets are needed.

In Türkiye as well as throughout the world, asthma is a major cause of morbidity. Although the prevalence of asthma varies between countries and even within regions, it ranges between 1-20% in the world's general population and affects an estimated 300 million people worldwide [81]. These regional differences are explained by the heterogeneity of genetic and environmental factors. Data obtained from studies conducted in Türkiye indicate that the prevalence of asthma in adults ranges from 1.2% to 9.4%, while the prevalence of asthma-like symptoms varies between 9.8% and 27.3% [82]. In some prevalence studies of allergic diseases in Türkiye, the prevalence of asthma, allergic rhinitis, and eczema in children between the ages of 6-18 was 1.8%-17.8%, 7.9%-43.2% and 2.1%, respectively. It was found to be 10.7% [83–85].

Standardized research on the prevalence of asthma worldwide is essential to advancing our knowledge of the illness. These kinds of investigations might help lower the severity, mortality, and prevalence of asthma. The Global Asthma Network (GAN) has established a unique network of collaborators, enabling a range of studies to advance knowledge about asthma. GAN has furnished standardized global information on asthma, encompassing the frequency and seriousness of asthma symptoms in children, adolescents, and adults worldwide. During Phase I, the Global Asthma Network (GAN) evaluated the prevalence of asthma symptoms from 2015 to 2020 through questionnaires, employing a standardized research protocol consistent with the International Study of Asthma and Allergies in Children (ISAAC). As seen in Figure 2-6 in GAN Phase I, the collective prevalence of present asthma symptoms stood at 9.1% among children, 11.0%

among adolescents, and 6.6% among adults. Disparities were observed based on the income level of the countries, with lower prevalence rates across all three age groups noted in low- to lower-middle-income nations, while the highest prevalence rates were recorded in high-income countries [86].

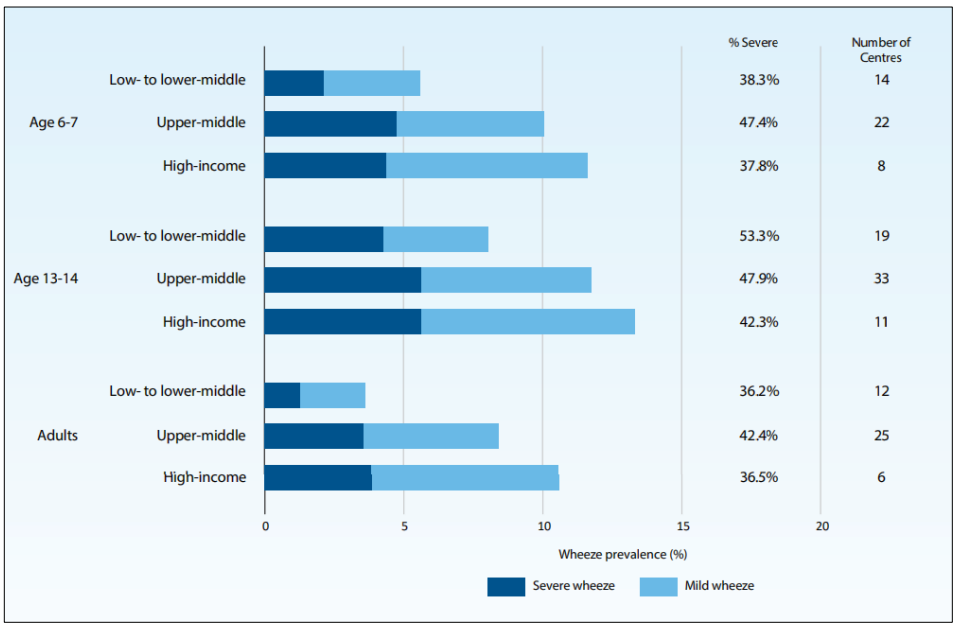


Figure 2-6. GAN Phase I: Severity of asthma symptoms by age group and country income level [86]

While most asthma symptoms across all age groups should be able to be controlled, only about half of individuals with asthma achieve well-controlled symptoms despite treatment guidelines. Alarmingly, one in five children and adolescents, as well as one in eight adults, experience uncontrolled symptoms, resulting in unnecessary suffering for patients and their families, and imposing avoidable burdens on health systems.

Asthma diagnosis by a physician was confirmed in 6.3% of children across 44 centers in 16 countries, 7.9% of adolescents across 63 centers in 25 countries, and 3.5% of adults across 43 centers in 17 countries. Overall, 44.1% of children, 55.4% of adolescents, and 61.1% of adults with asthma achieved well-controlled symptoms. However, a notable proportion experienced uncontrolled asthma (25.3% for children, 22.3% for adolescents, and 16.0% for adults) (Figure 2-7) [86].

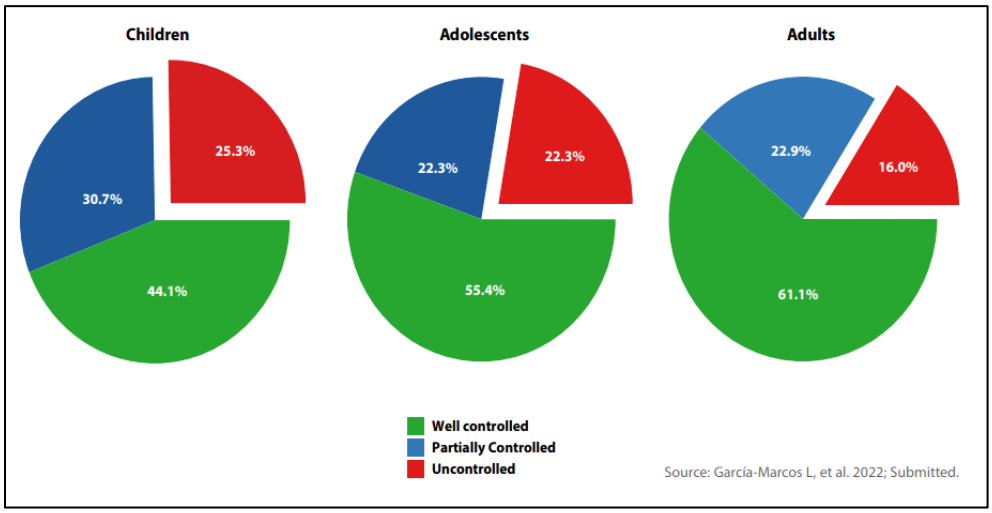


Figure 2-7. Asthma control among children, adolescents, and adults worldwide

2.8 Risk Factors in the Development of Asthma

Despite advancements in therapy, the ongoing increase in asthma prevalence indicates a limited understanding of its fundamental causes. Like prevalence data, studying risk factors and protective relationships in asthma has been challenging due to the multitude of associated factors. Notably, there is a significant overlap in risk factors for childhood and adult-onset asthma.

According to research, asthma risk factors include genetic predisposition, atopic structure, gender, obesity, and environmental factors. Symptoms can be triggered by allergens, stress, exercise, contact with irritating substances, environmental air pollution, cigarette smoke, and viral infections [87].

The likelihood of having asthma is significantly influenced by genetic factors [88]. While extensive research has been conducted on the genetic components of asthma, with numerous candidate genes identified, it appears that epigenetic and environmental factors play a significant role in the expression of asthma phenotypes. Further investigation is necessary to delve deeper into the epigenetics of this disease [89]. There are likely multiple gaps in the diagnosis and treatment of asthma, potentially resulting in poorly controlled asthma, severe asthma, reduced quality of life, and avoidable asthma-related mortality. Proper asthma management could mitigate these issues, alleviating both the human suffering and the economic burden associated with the condition. The gaps in asthma management are diverse and involve various factors, including those related to

patients, healthcare providers, healthcare systems, and policies [90]. Figure 2-8 provides an overview of the various gaps in asthma management from a high-level perspective.

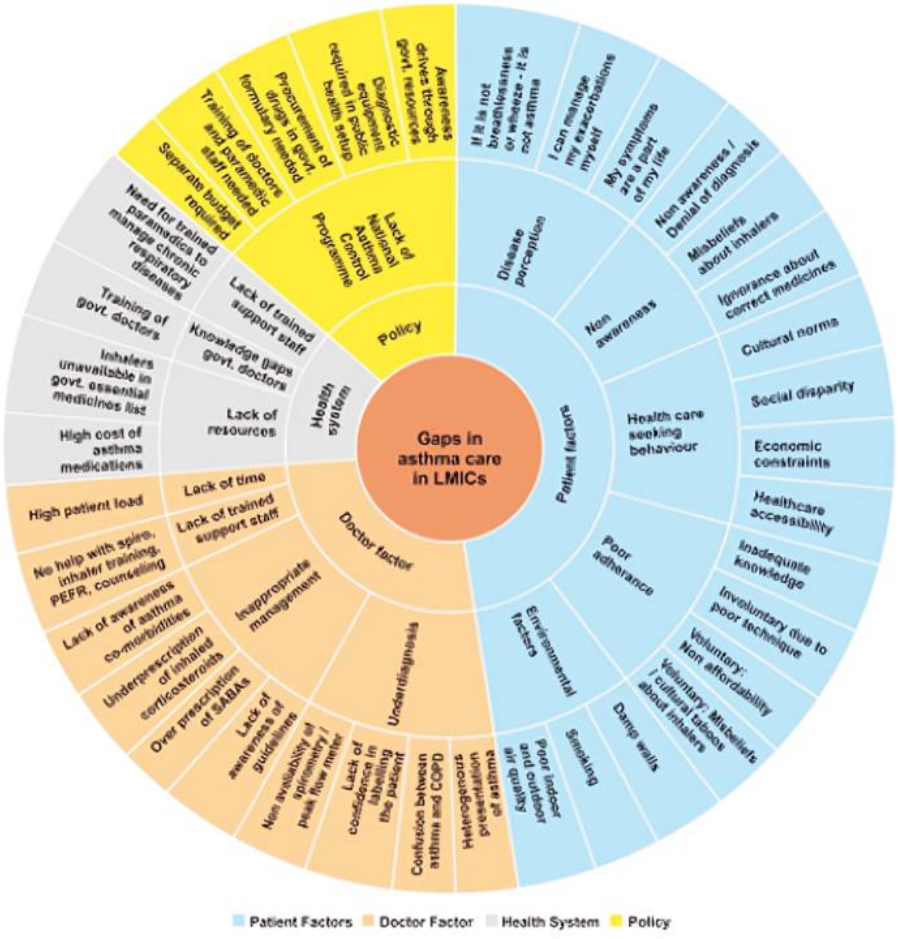


Figure 2-8. Gaps in asthma care in low-and middle-income countries [90].

One of these gaps, where little research has been conducted, is environmental factors. Environmental factors are considerably more probable than genetic factors in causing the rises in asthma prevalence in certain global regions. However, our understanding of all these factors, their interplay with each other, and their interaction with genetic elements remains incomplete. If there is no family history of atopy, a lack of awareness regarding environmental factors that elevate asthma risk may result in denial of the diagnosis [90].

Several environmental factors can impact the probability of developing asthma. These include indoor and outdoor air quality, exposure to tobacco smoke, pet ownership, type of fuel used for heating and cooking, and various other environmental factors that can exacerbate the severity of asthma symptoms [75,91,92].

The quality of air plays a significant role in influencing asthma symptoms and can act as a trigger for asthma attacks.

Some studies found a link between the presence of air pollutants, particularly particulate matter, in patients' homes, asthma status, and the frequency of innate immune cells (ILCs) in induced sputum. A noteworthy affirmative association was seen between asthma control and the quantity of PM with a diameter of less than 10 μm, coupled by a higher incidence of ILC2s in sputum that was produced [93,94]. The chance of having asthma can be influenced by numerous environmental factors. Both the symptoms of asthma and the likelihood of an asthma attack are influenced by air quality. Many studies have demonstrated the relationship between outdoor air pollutants such as particles (PM10), ozone (O3), nitrogen dioxide (NO2) and sulphur dioxide (SO2) and asthma [95,96].

However, the fact that indoor pollutants are one of the causes of asthma has been addressed in fewer studies. Some research has been done to investigate possible determinants of symptoms related to allergies and asthma. Kitchen ventilation, mold, dust bins, indoor emissions of volatile organic compounds (VOCs), and indoor dust particles have been suggested to be associated with an increased risk of allergies and asthma [97,98]. Current research indicates the existence of gaps, with a limited number of studies comprehensively evaluating potential environmental risk factors—whether they are objectively quantified or self-reported—in both indoor and outdoor environments. Additionally, it remains unclear which combination of these risk factors poses the highest health risk for children. Therefore, it is very important to conduct studies such as these thesis studies to better reveal the relationship between asthma and environmental factors. This thesis study investigates indoor air pollutants, which are among the most important environmental factors.

2.9 Asthma in children

Asthma, characterized by reversible airway obstruction and inflammation, stands as the most prevalent chronic disease among children. Those with asthma face an elevated risk of exacerbations when exposed to heightened concentrations of ambient air pollutants [99,100]. Recent studies also indicate that ambient air pollutants, particularly those emanating from traffic sources, may escalate the risk of new-onset asthma [101]. Childhood asthma poses a significant threat to respiratory health and can greatly disrupt normal functioning in children [102]. In recent decades, there has been a remarkable

increase in both the incidence and mortality rates of this immune disease [103]. Many children with asthma exhibit atopic characteristics, meaning they are sensitized to aeroallergens, as evidenced by symptoms and the results of allergen challenges or detection of allergen specific IgE [104].

For the diagnosis of asthma in preschool-aged children, the Global Initiative for Asthma (GINA) 2014 Global Strategy for Asthma Management and Prevention (GSAMP) offers an alternate method [105]. The following are the essential attributes that are advised:

- Recurrent Cough: Coughing fits that tend to be worse at night or in the early morning.
- Wheezing: It is a high-pitched whistling sound that is frequently heard when exhaling.
- Dyspnea: Shortness of breath or difficulty breathing.
- Decreased Activity Levels: A discernible drop in the child's typical level of activity, frequently brought on by respiratory problems.
- Family History: A child's risk of having asthma is increased if there is a history of asthma in the family or other atopic disorders, such as eczema or allergic rhinitis.
- Empirical Reaction to Treatments for Asthma: When asthma drugs such bronchodilators or inhaled corticosteroids are used, symptoms improve, indicating that asthma is the underlying cause.

2.10 Association between Endocrine Disruptors and Asthma

In recent decades, accumulating evidence has indicated that exposure to EDCs is implicated in the development of various human diseases, with multiple mechanisms potentially involved [106,107]. According to the WHO, about 800 chemicals have been identified or suspected to have the capability of disrupting various mechanisms of the endocrine system, including receptor binding and hormone synthesis. This interference can potentially lead to adverse health effects in individuals or populations exposed to these chemicals and may play a significant role in the development of various endocrine disorders [106,108]. Exposure to EDCs has been linked to elevated levels of oxidative stress and alterations in the production of cytokines. For example, EDCs may influence the modulation of TH2 cells, leading to changes in IgE production and eosinophilic

responses. Additionally, EDC exposure has been associated with hypomethylation of the promoter region of TNF-α [109,110].

Previous research revealed a link between children's individual and coexposure to low amounts of EDCs in classrooms and an elevated risk of obesity, asthma, and nasal obstruction in the three months prior. Additionally, they discovered that EDCs exposure was linked to modifications in the autonomic nervous system, particularly parasympathetic dysautonomia. This finding raises the possibility that EDCs may raise parasympathetic activity, which in turn raises the risk of obesity, respiratory disorders, and asthma [111].

Consumer products expose people to a variety of chemicals that might cause asthma attacks and disturb hormones [112]. A U.S. study identified a wide range of endocrine-disrupting and asthma-related compounds in hair products used by Black women, including those marketed towards children. This highlights the potential for significant exposure to harmful chemicals through everyday personal care products in this demographic [113].

Prior research has predominantly concentrated on the impact of exposure to specific semi-volatile organic chemicals as EDCs, like bisphenol and phthalates, on respiratory health [114–116]. Wang et al suggested that EDCs play a role in the development of asthma. According to the results of this study, higher exposure to EDCs is associated with lower methylation of TNF-α 5′ CGI, which is associated with airway inflammation, hypersensitivity, dysregulation of immune cells, and higher risk of asthma in children [117].

Concerns about endocrine disruption and asthma caused by exposure to chemicals in consumer products have been raised by research. But there is insufficient testing or labeling data to consider products as sources of exposure.

2.11 Association between Persistent organic pollutants and Asthma

The development of asthma is considered to result from geoenvironmental interactions, where environmental factors, including POPs, may play significant roles [118].

Even though the production of many POPs, such as PBDEs and PCBs, has been banned in numerous countries since the 1970s, these compounds can still be detected in the

human population due to their ability to bioaccumulate. Exposure to POPs during early life may have adverse effects on the development of children's respiratory and immune systems [119].

The evidence of adverse effects of POPs on the developmental respiratory and immune systems in children remains limited, and the biological mechanisms underlying such effects are not fully understood. Several studies have evaluated the effects of early-life exposure to POPs on immune cell counts, such as T cells or B cells. The goal of these studies is to examine potential biological mechanisms underlying the association between POP exposure and immune and respiratory health in children [119,120].

Traditionally, traffic and second-hand smoke have been recognized as primary causes of childhood asthma. However, emerging research suggests that POPs may also play a significant role as potential triggers. Several investigations have reported associations between prenatal or postnatal exposure to typical POPs and childhood asthma or asthma-related syndromes [121].

PAHs and other anthropogenic oxidants are hypothesized to influence the risk of asthma by inducing bronchial inflammation and bronchoconstriction [122].

Experimental studies indicate that pyrene has the potential to induce either allergic or nonallergic immune responses [123,124]. A group at the Columbia Center for Children's Environmental Health (CCCEH) reported that increased prenatal exposure to the sum of 8 nonvolatile PAHs combined with environmental tobacco smoke (ETS) was linked to wheezing and asthma in infants and young children [125,126]. Epidemiological evidence suggests strong associations between PAH exposure levels and various adverse health effects on the respiratory and cardiovascular systems. These effects include reduced lung function, asthma, myocardial infarction, as well as increased risks of skin and bladder cancer, and overall mortality. In the past, many studies primarily reported qualitative associations between exposure to PAHs in ambient air and asthma. Only a few studies focused on quantifying the association between PAHs exposure and childhood asthma, and the findings were inconsistent. However, several studies have supported the notion that exposure to PAHs increases the risk of asthma in childhood [127,128]. According to the results of a study by Wang et al., exposure to PAHs is positively correlated with oxidative stress, and oxidative stress may play a role in the development or exacerbation of asthma [129]. Several studies have reported associations between PAHs exposure and

various adverse outcomes in children. These include asthma symptoms, biomarkers of asthma, respiratory health, and cognitive development [130,131]. PAHs have also been associated with deficits in pulmonary function and increased incidence of wheezing in children [132]. Epidemiological data also support the notion that exposure to PAHs in children contributes to asthma morbidity [133].

PBDEs, PCBs and PAHs are common POPs that may be associated with childhood asthma. Certain studies have reported associations between prenatal and postnatal exposure to typical POPs and childhood asthma, as well as asthma-related syndromes. Researchers have extensively demonstrated the adverse effects of PCBs on children's immune systems, which may also be associated with asthma [134]. A Canadian cohort study observed associations between prenatal exposure to PCB153 and an increased incidence of respiratory infections [135]. Moreover, a study on a birth cohort comprising nearly 900 mother-child pairs found that prenatal exposure to hexachlorobenzene (HCB) and -PCB-118)- elevated the likelihood of developing asthma by the age of 20 [136]. Meng et al. demonstrated potential associations between childhood asthma and internal exposure levels of selected typical POPs [137]. In Flemish children, Van Den Heuvel et al. discovered that dioxin was associated with a reduced risk of asthma, while PCB exposure was linked to an increased risk of asthma [138]. There is compelling evidence indicating that non-coplanar, non-dioxin-like PCBs can disrupt the function of the immune system [139]. Given that both dioxin-like and non-dioxin-like PCBs affect the expression of numerous genes, there are multiple potential pathways through which immune system function could be altered [140].

2.12 Association between Alkylphenols (APs) and Alkylphenol Ethoxylates (APEs) and Asthma

Non-ionic surfactants, such as alkylphenols and alkylphenol polyethoxylates, have already been classified as EDCs and are used widely in cleaning products such as detergents, as well as other household and consumer goods. The human can be exposed to octylphenol, nonylphenol, and nonylphenol ethoxylates by eating, breathing, or coming into direct touch with skin-care products [63]. Cleaning products are mixtures of many chemicals, including alkylphenols and alkylphenol ethoxylates, which are known to contain sensitizers, disinfectants, and fragrances. These ingredients can act as strong airway irritants and are associated with lower respiratory tract issues and asthma symptoms. Regular exposure to such chemicals, especially in poorly ventilated indoor

environments, can exacerbate respiratory problems and contribute to the development and severity of asthma [141].

Recent studies have linked the oxidative stress response to asthma in cleaners working in hazardous environments. Oxidative stress, caused by an imbalance between the production of free radicals and the body's ability to detoxify them, can damage cells and tissues. Cleaners are often exposed to high levels of chemicals found in cleaning products, such as alkylphenols, disinfectants, and fragrances, which can induce oxidative stress. This exposure may contribute to the onset or worsening of asthma symptoms in these workers, highlighting the need for improved safety measures and the use of less harmful cleaning products [142]. According to the study conducted by Amani et al., workers in the detergent and cleaning products industry are at a higher risk of developing occupational asthma. Cleaning products contain chemical substances that can contribute to the development of occupational asthma or exacerbate symptoms in individuals with pre-existing asthma. These substances can irritate the respiratory tract or cause sensitization, adversely affecting workers' general health and quality of life [141].

NP and OP can build up in the human body and produce allergy disorders due to their low solubility, high hydrophobicity, and low estrogenic action. By changing cytokine synthesis, NP and OP can both have an impact on T cells, which are crucial for the development and maintenance of asthma in mice [143,144].

2.13 Literature Review

Asthma is a common non-communicable disease (NCD) in children and adults, affecting an estimated 262 million people and causing 461,000 deaths in 2019. There is a significant increase in asthma cases in young children worldwide, especially in developed and developing countries [145–147]. In Türkiye, asthma is also the most frequent chronic health problem among children and adults. According to Türkiye 2020 Asthma Guidelines, there are approximately 4 million asthma patients in this country, and it is known that the prevalence of asthma ranges between 0.7-21.2% in children and 1.2-9.4% in adults. In this report, one in every 8 children and one in every 12 adults have asthma, and about 30% of doctor-diagnosed asthmatics reported that they were absent from school or hospitalized for asthma [148]. A limited number of studies conducted in various regions in Türkiye have shown that air pollution may cause a decrease in lung functions

of school children and increase asthma emergency admissions and hospital admissions in children and adults [149–154].

Some compounds found in building materials, furniture, and consumer products have been identified as EDCs, making them possible indoor contaminants/pollutants, and suggesting that indoor exposure may contribute more to overall EDC exposure [155]. This finding realized that changes in modifiable environmental exposures are likely to play a key part in the rising prevalence of non-communicable diseases like asthma and obesity among children in affluent nations, which cannot be attributed solely to genetic alterations [156]. Childhood is a pivotal period of sensitivity to EDC exposure, as early exposure can have long-term consequences that endure into maturity and harm future generations. This emphasizes the importance of gaining a better knowledge of the impact of EDC exposure on the health of youngsters, who may be particularly vulnerable to low concentration effects because of their extensive time spent indoors. Therefore, we aimed to assess the association between indoor EDCs exposure in house and asthma in school-age children. A study by Paciência et al. surprisingly showed that even low exposure to EDCs can increase the risk of asthma. They analyzed data from a cross-sectional study with 815 participants from 20 schools in Porto, Portugal. Asthma was defined based on lung function, symptoms were evaluated, and body mass index (BMI) and airway reversibility were computed. Over the course of a week, the concentrations of two aldehydes and thirteen VOCs were tested in 71 classrooms. Their results provide more evidence that EDCs play a part in the development of obesity and asthma. Furthermore, even minimal indoor exposure may have an impact on the likelihood of developing obesity, respiratory disorders, and asthma [111].

POPs are one of the most important components of EDCs. Limited research have identified associations between prenatal and postnatal internal exposures of many POPs [157–159], however, it has not been extensively studied whether exposure to indoor dustborne particles of POPs is associated with asthma.

Figure 2-9 shows the types of POPs included in the studies and their study percentages. The figure highlights the distribution and prevalence of different POPs examined across various research studies, providing insight into which pollutants are most commonly investigated. As seen in Figure 2-9, the PAH, PCB, and PBDE species that are most abundant in the indoor environment were investigated. In this thesis, in addition to these common species, alkylphenol ethoxylates and alkylphenols, which have been the subject

of very few studies, are among the compounds to be tested as endocrine-disrupting chemicals. This thesis scope aims to provide a more comprehensive understanding of the variety and impact of EDCs present in indoor environments, potentially uncovering less-studied but significant contributors to indoor pollution and health risks such as asthma.

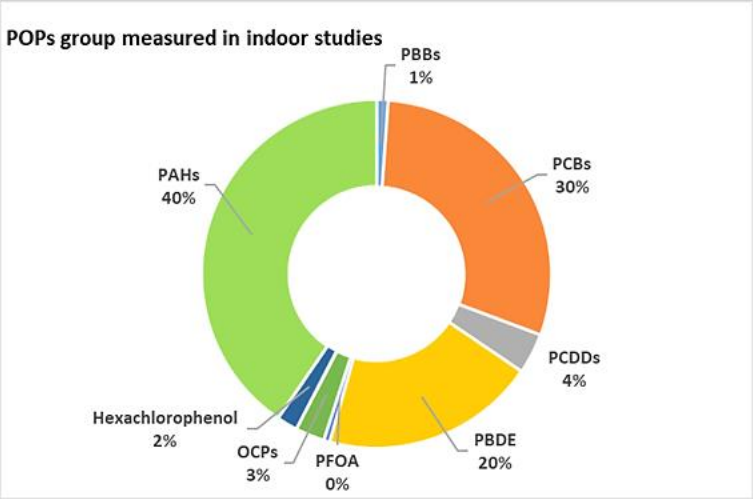


Figure 2-9. Types of Persistent Organic Pollutants (POPs) included in the indoor studies and their percentages

Asthma symptoms have been linked to exposure to PAHs, which make up a significant portion of fine particulate matter from combustion sources, according to recent studies. A study by Karimi et al investigated the sources of childhood PAH exposure and the relationship between airborne PAH exposure and childhood asthma prevalence and exacerbation [160]. One of the main ways that children are exposed to PAHs is through tobacco smoke, which has also been linked to childhood asthma. A positive correlation between exposure to tobacco smoke in the home and both incident and current childhood asthma (RR 1.21, 95% CI 1.08–1.36) was found in a meta-analysis of papers published between 1970 and 2005 [161]. According to research, exposure to PAHs has been linked to adverse outcomes in children, including asthma symptoms, asthma markers, respiratory health, and cognitive development [162–164].

PCBs are a group of more than 200 chemicals with various uses and structural configurations. They are commonly employed in industries such as polymers, electronics, paints, and insulation, as well as in coolant applications. However, PCBs were first identified as POPs under the Stockholm Convention [165] and classified by the US EPA as likely human carcinogens (Group B2) due to their persistence,

bioaccumulation potency, and toxicity. Despite this, many studies have shown that PCBs accumulate in indoor dust [166–168].

PBDEs are the subject of the great bulk of research on indoor dust pollutants conducted globally [169,170]. Few epidemiological studies have evaluated the effects of PBDEs on human health, even though PBDE has been found in indoor dust on a regular basis, raising concerns about health [171–173]. Canbaz et al investigated whether PBDE concentrations in indoor dust were associated with the development of childhood asthma. They selected 110 children who developed asthma at age 4 or 8 years and 110 matched controls from a large perspective birth cohort (BAMSE – Barn, Allergy, Milieu Stockholm Epidemiology). They analyzed the concentration of 21 PBDEs in dust collected from the mother's bed about two months after birth [174].

The study by Ge Meng et al. shows potential associations between internal exposure concentrations of particulate POPs and the development of childhood asthma. Significant differences in PBDE, PCB, and OCP levels emerged between the case and control groups included in the study. According to the results, significantly higher levels were seen in the cases. The internal exposure concentrations of several POPs were found to be positively linked with childhood asthma in various logistic regression models [175].

As it seems, even though most of these pollutants are limited by the Stockholm convention signed in 2001 to protect the environment and human health, these pollutants are still encountered in various environmental samples in studies conducted around the world.

The alkylphenols, by-products of microbial degradation of alkylphenol polyethoxylates (APP), constitute an important class of EDCs. Long-chain alkylphenols (APs), such as OP and NP, are a class of compounds used by industry to improve the production of a wide range of products, including plastics, lubricants, detergents, cosmetics, and pesticides. APEOs, such as nonylphenol and octylphenol ethoxylates, are the most common nonionic surfactants. Surface-active chemicals' primary feature is their ability to concentrate at diverse surfaces and form micelles in solutions [176].

Human exposure to alkylphenols and alkylphenol ethoxylates has been demonstrated to occur by dust ingestion, air inhalation, dietary exposure via food and water intake, and skin contact with goods containing these chemicals [64,65,177].

Several studies have found that excessive alkylphenols and alkylphenol ethoxylates exposure negatively. Other side effects include antiandrogenic activity, hepatotoxic impact, and irritation of the lungs, digestive system, and eyes and skin [66,67].

Recent research has demonstrated the importance of house dust as a medium and route of exposure for EDCs such NPs and NPEs [64,65]. Many organic contaminants have been implicated in indoor dust as a receptacle and concentrator; thus, levels of contaminants in indoor dust can be utilized as a proxy to assess the risk for exposure to pollutants in the interior environment [178].

In France, sixty-six sVOCs, including alkylphenols, were measured in the particulate phase in dwellings. 4-nonylphenol was infrequently detected, whereas 4-tert-butylphenol and 4-tert-octylphenol were found in approximately one out of every two households [32]. Rudel et al., sampled indoor air and dust in 120 homes on Cape Cod, analyzing for alkylphenols and alkylphenol polyethoxylates. Among the most prevalent substances found were alkylphenols, including 4-n-NP and its mono- and diethoxylates in air and dust samples [112].

As far as we know, there is currently no information about the presence of these chemicals in house dust and their relationship with asthma in Türkiye. This thesis aims to fill this gap by investigating the levels of EDCs in indoor dust and examining their potential links to asthma severity. By doing so, it seeks to contribute to the body of knowledge on indoor environmental health and the prevalence of asthma and guide future research and regulatory actions in Türkiye.

2.14 Researches Gaps

Even while evidence linking exposure to EDCs with unfavorable health outcomes is mounting, drawing conclusions from observational data in epidemiological research is still difficult. Therefore, experimental research is essential to verify observational results and offer light on the molecular mechanism behind the identified relationships. To get over these constraints, more comprehensive strategies like connecting experimental and observational human data and putting more emphasis on chemical mixes that closely resemble real-world exposures could be crucial. There is currently insufficient evidence regarding endocrine mechanisms, and overall, there is a lack of study in this area.

While environmental factors are believed to be more influential than genetic factors in driving increases in asthma prevalence in certain regions of the world, our understanding

of these factors and their interactions with genes remains incomplete. Moreover, some environmental factors may exert different effects in affluent and non-affluent countries.

The possible impact of EDCs on asthma and other health disorders has attracted a lot of interest. Nonetheless, in order to have a more comprehensive understanding of the global association between EDCs and asthma, a number of gaps in the current study must be filled. Here are some key gaps:

- Coverage by Region: The majority of research on EDCs and asthma is done in industrialized nations, mostly in Europe and North America. Research in poor nations, such those in Africa, Asia, and Latin America, where genetics and varying environmental exposures may have an impact on the association between EDCs and asthma, is deficient.
- Long-Term Research: To fully comprehend the long-term impacts of EDC exposure on the onset and progression of asthma, longer-term research is required. Since many of the current research are cross-sectional, it is challenging to determine long-term effects and causality.
- Evaluation of Exposure: Better techniques are required to evaluate each person's exposure to EDCs. Many of the studies conducted now use measures taken at a single time point, which leaves out variability and cumulative exposure over time. It is required to use more thorough exposure assessment instruments, such as biomarkers and personal monitoring.
- Effects of Mixture: Research frequently concentrates on individual EDCs rather than the synergistic effects of several EDCs. People are exposed to complex mixes of EDCs in real-world situations, and determining the underlying health hazards requires an understanding of the combined impact of these EDCs.
- Intervention and Research on Policies: There aren't many research that assess the efficacy of strategies to lessen exposure to EDCs or lessen their impact on asthma. Translating scientific discoveries into practice requires research on public health interventions, policy initiatives, and their results.
- Interactions between the environment and socioeconomics: There hasn't been much research done on how socioeconomic status, environmental exposures, and EDCs interact with asthma. A comprehensive understanding of the interplay between these factors can offer insights into the etiology and inequalities of asthma.

To close these gaps and create global strategies for researching and reducing the effects of EDCs on asthma, a multidisciplinary team of toxicologists, epidemiologists, doctors, and policymakers is needed.

2.15 Aim and Scop of this study

As mentioned in the previous sections, indoor air quality seriously affects the health of children, especially those who spend most of their time indoors. Poor indoor air quality can exacerbate respiratory conditions such as asthma, leading to increased morbidity and impacting overall well-being and development. Therefore, understanding and mitigating the sources of indoor pollutants is crucial for protecting children's health.

The primary purpose of this thesis is to examine the effects of indoor environmental quality, which is determined by the analysis of EDCs (persistent organic pollutants and surface-active substances) of house dust, on the development of asthma in school-age children. The concentrations of EDCs will be analyzed in house dust samples from case (asthmatic) and control (non-asthmatic) children.

Furthermore, at the end of the study below items are expected to be determined:

• Determination of endocrine-disrupting chemicals concentration and pollutant sources of house dust taken from the homes of school-age children

• Determination of the relationship between endocrine-disrupting chemicals levels observed in house dust and the development of asthma in schoolchildren.

• Evaluation of the relationship between the levels of pollutants observed in house dust and some socio-demographic characteristics of children with asthma.

• Contribute to improving air quality in indoor environments where children live.

• Create a database for indoor air quality and possible health effects.

This thesis has the potential to furnish sufficient evidence to advocate for more restrictive use of EDCs in order to prevent human exposure during sensitive stages such as asthma.

2.16 Significance and Limitations of this Study

There has been a significant increase in the incidence of asthma in children in the last decade. The rise of illness in children is a significant problem, both for the long-term treatment costs for families and the economy of the country, as well as for growing unhealthy generations. Standardized research on the frequency, severity, and mortality

of asthma worldwide is essential for improving our understanding of the condition. It is not possible to define a single natural course of asthma, and although previous studies have shown that the effects of environmental factors on the severity of asthma are more important than genetic factors, information on the relationship between these factors and their effects on the genetic structure is lacking.

In this thesis, the levels of indoor air pollutants, one of the most important environmental factors affecting the development and severity of asthma, are investigated. The main objective of this study is to determine EDCs levels in house dust and find out how these chemical pollutants affect asthma disease.

We can summarize the significance of this study as follows:

- Comprehensive Analysis: This study expands the understanding of indoor air quality and its impact on asthma severity by providing a comprehensive review of the levels, sources, and relationships of POPs and alkylphenols in indoor dust to home conditions and living habits. As a case-control study, it included both asthmatic (case) and non-asthmatic (control) children. In addition, a sampling period lasting one year can reveal the cumulative effect of exposure to pollutants.
- Focus on Less Studied Compounds: This research investigating less studied alkylphenol ethoxylates and alkylphenols as EDCs fills an important gap in the current scientific literature.
- Health Effects: Stricter controls are required to safeguard vulnerable groups, particularly children, as the findings point to possible health hazards related to indoor pollution exposure.
- Impact on Policy and Public Health: The findings may help public health professionals and policymakers impose stronger regulations for household items and higher air quality standards to reduce exposure to dangerous chemicals.
- Global Relevance: By comparing pollutant levels with data from different nations, this study offers a worldwide perspective and adds to a larger conversation on indoor pollution and its health effects. Although a few researchers around the world have worked on this subject, no study has been conducted in Türkiye to determine the amount of pollutants in the house dust of children with asthma and correlate it with the development of asthma. Therefore, the data obtained from this study are of great importance and will contribute to world literature.

- Practical Recommendations: The study provides actionable information on improving cleaning practices, minimizing the use of products containing harmful chemicals and increasing indoor ventilation, and reducing indoor pollution that will affect the severity of asthma in children and families such as: smoking, using detergents, frying, etc.

In this study, like other experimental studies, various factors, including differences in population characteristics, confounding variables, sampling times and analytical methods, and exposure to chemical mixtures rather than single chemicals, pose challenges in drawing definitive conclusions. The limitations of this study may be as follows:

- Sampling Variability: Dust samples taken from particular homes may not be representative of all indoor environments, which affects the study's conclusions.
- Temporal Restrictions: The sampling was done in a constrained amount of time. The generalizability of the results may be impacted by seasonal variations in pollutant levels brought on by adjustments to heating, ventilation, and human activity.
- Chemical Analysis Limitations: The study finds connections between pollutant levels and characteristics of households, but it is unable to prove causation. To confirm the origins of these contaminants and their impact on asthma, more experimental research is required. Furthermore, this study only looked into a small number of endocrine disrupting chemicals due to budgetary constraints.
- Population Specificity: Because the study was limited to a single area, its conclusions might not apply to communities in other socioeconomic, environmental, or cultural contexts.
- Potential Confounding Factors: Despite efforts to control various factors (Temperature, humidity, etc.), unaccounted for confounding variables may still exist that could influence the results, such as the presence of other pollutants or individual differences in asthma susceptibility.

Despite its limitations, this study significantly advances the understanding of indoor pollutants and their effects on health, particularly asthma severity. The knowledge gained can guide future research, policymaking, and public health interventions to create safer indoor environments.

3 MATERIALS AND METHODS

In this study, our general goals are twofold: (1) to determine the levels and sources of endocrine-disrupting chemicals in house dust, and (2) to evaluate the relationship between these chemicals in settled house dust and factors influencing variability in asthma severity. We used indoor dust samples collected from ten different regions of Ankara to ensure sufficient variability in the levels of chemical pollutants. Additionally, we prepared surveys consisting of approximately 70 questions to determine environmental factors. For most epidemiological data on allergic diseases, we relied on patient follow-up forms prepared by physicians. These forms provide information on the frequency and prevalence of symptoms, as well as the presence or absence of a medical diagnosis. Last, we employed various statistical analyses to evaluate both the analytical results and the survey data.

3.1 Study Population

In this study, case and control groups were composed of 6-11 year old children with and without asthma in different regions of Ankara, the capital of Türkiye (Figure 3-1). The case group (children with asthma) consisted of patients aged 6-11 years who were diagnosed with asthma by professional doctors at Hacettepe University Pediatrics Clinic. The sample size to be reached in the study was calculated as 100 for each of the case and control groups, taking into account the number of 850 children who applied to the outpatient clinic and were followed up or newly diagnosed with asthma, with a 5% error rate, a 95% confidence interval, and a 10% exposure frequency. Taking into account the non-response rate of the patients during follow-up (20%), possible losses that may occur due to loss of follow-up or return of house dust, it was planned to reach 150 children and their families in both groups. Since 30 case groups stated that they gave up taking part in the study in the later stages, the number of case groups remained at 110.

To form the control group, the relevant permission was received by the Ankara Provincial Directorate of National Education to reach school-aged children between the ages of 6-11 who were not diagnosed with asthma within the Ankara provincial borders. Based on the permission received by the Ankara Provincial Directorate of National Education, a total of 5 primary schools in different regions of Ankara were visited in order to reach the control group we wanted, and after interviews were held with the school principal and

teachers, the information letters we prepared for the families were delivered to the families of children aged 6–11. In the information letters, the purpose and scope of the project are generally explained. Families who wanted to participate in the project voluntarily were asked to fill out a volunteer consent form. After reaching 150 families to form the control group, 20 families later gave up participating in the study, and thus the number of children in the control group became 130.

The socioeconomic position of the families taking part in the research in the asthma and control groups is comparable. Consent forms were prepared to provide information about the study to the invited families, and these forms were filled out by the families who agreed to participate in the study. Moreover, this study received the approval of the institutional review board (IRB) from The local Ethical Committees in Türkiye (26/02/2016-E.7061). Appendix A has the approval from the ethics committee, and Appendix B contains the form requesting parental consent.

Figure 3-1. Different regions of Ankara where house dust samples were collected

3.2 Questionnaire survey

In parallel with the sample process, a questionnaire survey was carried out to assess the risk variables associated with pediatric asthma. The specifically designed survey form was applied to the families of children who wanted to participate in the study voluntarily, using a face-to-face interview technique. In the survey form, the characteristics of the

house you live in (which floor it is located on, its size, whether there is humidity, heating system, wall covering type, carpet, furniture, etc.), whether there are animals in the house, human activities in the house (smoking, cooking methods, frequency of ventilation) are included. , etc.) questions (detailed in Table 4-1. Characteristics of study populationIBM SPSS 23 was used to process the survey results and perform additional statistical analysis.

3.3 Dust sample collection

At least three dust samples representing long-term exposure to persistent pollutants were collected at 4-month intervals spanning one year from all homes to be included in the study. Dust samples were taken from vacuum cleaners in the homes of the children participating in the study. To store the dust samples, previously cleaned aluminum foiled and labeled zip bags were used. At the end of the sampling period, all dust samples from the same house were combined and homogenized, and then a composite dust sample was prepared and then passed through a 100 mesh sieve to separate the coarse particles they contained. Dust samples were stored in the freezer at -20°C until analysis. Temperature and humidity levels in the sampling houses were recorded.

3.4 Chemicals, materials and sample analyses

3.4.1 Analysis of Persistent Organic Pollutants

Analysis methods have been created for PAH, PCB, PBDE analysis. The chemicals used during the extraction and pre-enrichment of the samples for PAH, PCB and PBDE analysis (Dichloromethane, Acetone, Ethyl acetate, Hexane and Sodium sulfate) were of 99% purity and Petroleum ether was of 90% purity, florisil (200 meshes) and neutral aluminum oxide (0.063-0.2 mm) and was obtained from Merc. Nitrogen gas is 99% high purity.

3.4.1.1 Polycyclic Aromatic Hydrocarbons (PAHs) Analyzes:

Certified PAH surrogate and standard solutions for analysis were purchased from AccuStandarts (USA) and Ultra Scientific (USA). Surragate compounds were used to determine the losses that may occur during the analysis, extraction, cleaning and enrichment processes in the samples (AccuStandard M-525 IS).

The list of standards used for calibrations and quality controls is shown in Table 3 1.

Table 3-1. Surrogate and standard solutions used for PAH analysis

	Chemical Compounds	Concentration	Supply Company
PAH (mix)	Nap, Acy, Ace, Flu, Ant, Phe, Flt, Pyr, BaA, Chr, BbF, BkF, BaP, DahA, Ind, BghiP	100 µg /mL	Ultra Scientific U-PM-610-1
PAH (sur)	Acenaphthene-d10, Crysened-12, Perylene-d12, Phenanthrene-d10	2000	AccuStandard M-525-IS-PAK 2.0 mg/ml in CH2Cl2

PAH analyze was carried out on the gas chromatography coupled with tandem mass spectrometry consisted of an Agilent 6890 GC system (Agilent Technologies, Santa Clara, CA, USA) connected to a triple quadrupole mass spectrometer 5975C (Agilent Technologies, Santa Clara, CA, USA) device at Hacettepe University Environmental Engineering Department. Data analysis is performed by Agilent Mass Hunter Quantitative Analysis software (Version：B.07.01/Build 7.1.524.0). GC MS calibration was performed with a calibration solution prepared with 16 target PAH compounds (Ultra Scientific PM 610) and 4 surragate compounds (AccuStandard M-525 IS). Then, 7 SIM windows were determined for analysis in Selected Ion Monitoring (SIM) mode. Detailed information about GC-MS operation parameters and Calibration is given in Table 3-2 and Table 3-3.

Table 3-2. GC-MS operation parameters for PAH analysis

Operation Parameters	
GC Column	30m*250µm*0.25µm Nominal Film
Liner	Thickness. 5% Phenyl Methyl Siloxane. HP 5MS, capillary column Deactivated Glass Cotton Splitless Glass Liner (Agilent Technologies)
Carrier Gas	Ultra Pure Helium, 99.999%, 1 ml/min
Injection Type	Splitless
Temperature of Injection Port	280 0C
Oven Temperature	70 ℃ (2 min), 7 ℃/min to 250 ℃ (5 min), 5 ℃/min to 300 ℃ (8 min)
Injection Volume	1 µL
Detector	MS
Mass Spectometer	Electron impact, 70eV
Mass Spectometerquadropole Temperature	150 ℃
Mass Spectrometry Source Temperature	230 ℃

Table 3-3. GC-MS calibration parameters for PAH analysis

PAH isomers	Ions (m/z)	Retention time (min)	Linear range (ppm)	Regression coefficient (r^2)
Nap	128, 127, 129	11.015	0.05-5	0.997
Acy	152, 151, 153	16.394	0.05-5	0.998
Ace	153, 154, 152	17.043	0.05-5	0.998
Flu	166, 165, 167	18.758	0.05-5	0.997
Phe	178, 176, 179	21.937	0.05-5	0.999
Ant	178, 176, 179	22.082	0.05-5	0.995
Flt	202, 200, 101	25.953	0.05-5	0.997
Pyr	202, 200, 101	26.662	0.05-5	0.998
BaA	228, 226, 229	30.859	0.05-5	0.994
Chr	228, 226, 229	31.014	0.05-5	0.999
BbF	252, 253, 126	36.334	0.05-5	0.998
BkF	252, 253, 126	36.465	0.05-5	0.997
BaP	252, 253, 126	37.942	0.05-5	0.997
Ind	276, 277, 138	42.98	0.05-5	0.995
DahA	278, 276, 139	43.193	0.05-5	0.998
BgP	176, 138, 277	43.911	0.05-5	0.998

3.4.1.2 Polychlorinated Biphenyls (PCBs) Analyzes:

For PCB analyses, a surrogate standard was determined to take into account losses that may occur during the ultrasonic extraction, pre-enrichment, clean-up, final enrichment and transportation of the samples to vials. 209 PCB compounds (congeners); PCB 14 (3,5-dichlorobiphenyl) is used as the surrogate standard to detect losses of components between PCB 1 and PCB 39, and PCB 65 (2,3,5,) is used as a surrogate standard to detect losses of components between PCB 40 and PCB 169. 6-tetrachlorobiphenyl) and finally PCB 166 (2,3,4,4',5,6-hexachlorobiphenyl) was preferred as a surrogate standard to detect the losses of components between PCB 170 and PCB 209. The identified surrogate compounds and target PCB compounds are given in Table 3-4.

Table 3-4. Surrogate and standard solutions used for PCB analysis

	Chemical Compounds	Concentration	Supply Company
PCB (mix)	PCB-14, 28, 52, 65, 101, 118, 138, 153, 166, 180	100 µg /mL	Dr. Ehrenstorfer Reference Materials
PCB (Sur)	3,5-Dichlorobiphenyl (PCB 14) 2,3,5,6-Tetrachlorobiphenyl (PCB 65) 2,3,4,4',5,6-Hexachlorobiphenyl (PCB 166)	100 100 100	Chem Service

Polychlorinated biphenyl compounds consisting of PCB 28, 52, 101, 118, 138, 153 and 180. The indicator PCB compound was chosen for the purpose of easy comparison of data coming from many laboratories around the world. The fact that these compounds are considered stable in biological and environmental systems, and practical and economic reasons are other factors in choosing these 7 PCB compounds as indicator compounds. In addition, since these 7 indicator PCB compounds are dominant in the biotic and abiotic matrix, they are assumed to represent all PCBs. In order to read the samples correctly, the calibration solution prepared with 7 PCB compounds (PCB 28, 52, 101, 118, 138, 153, 180) and 3 surrogate PCB compounds was calibrated by reading them on the GC-MS (ECD) device. Detailed information about GC-MS operation parameters and Calibration is given in Table 3-5 and Table 3-6.

Table 3-5. GC-MS operation parameters for PCB analysis

Operation Parameters	
GC Column	30m*250µm*0.25µm Nominal Film Thickness. 5% Phenyl
Liner	Methyl Siloxane. HP 5MS, capillary column
Carrier Gas	Deactivated Glass Cotton Splitless Glass Liner (Agilent Technologies) Ultra Pure Helium, 99.999%, 1 ml/min
Injection Type	Splitless
Temperature of Injection Port	280 0C
Oven Temperature	70 ℃ (2 min), 25 ℃/min to 150 ℃ (1 min), 3 ℃/min to 200 ℃ (1 min), 8 ℃/min to 280 ℃(5 min)
Injection Volume	1 µL
Detector	ECD
Mass Spectometer	Electron impact, 70eV
Temperature	150 ℃
Mass Spectrometry Source Temperature	230 ℃

Table 3-6. GC-MS calibration parameters for PCB analysis

PCB isomers	**Ions (m/z)**	**Retention time (min)**	**Linear range (ppm)**	**Regression coefficient (r^2)**
PCB 14	222, 152, 224	8.792	0,001-4	0,996
PCB 28	256, 258, 186	9.846	0,001-4	0,998
PCB 52	292, 290, 220	10.362	0,001-4	0,998
PCB 65	292, 290, 294	10.566	0,001-4	0,996
PCB 101	326, 328, 324	11.830	0,001-4	0,998
PCB 118	326, 328, 324	12.992	0,001-4	0,998
PCB 138	360, 362, 290	13.412	0,001-4	0,999
PCB 153	360, 362, 358	13.962	0,001-4	0,999
PCB 166	360, 362, 358	14.268	0,001-4	0,997
PCB 180	394, 396,398	15.305	0,001-4	0,998

3.4.1.3 Polibromlu difenil eter (PBDEs) Analyzes:

The list of standards used for calibration and quality control in PBDE analyzes is shown in Table 3 7.

Table 3-7. Surrogate and standard solutions used for PBDE analysis

	Chemical Compounds	Concentration	Supply Company
PBDE (mix)	PBDE17, BDE28, BDE47, BDE66, BDE85, BDE99, BDE100, BDE153, BDE154, BDE183, BDE190, BDE207, BDE209	5 µg /mL	AccuStandard (BDE-COC)
PBDE (Sur)	BDE 15 ve BDE 209	50 µg/mL	Cambridge Isotope Laboratories, Inc.

Calibration of the device for PBDE isomers included 14 target PBDE mix mixtures (Accusstandard) (BDE 17 (2,2,4'-tribromodiphenyl ether), BDE 28 (2,4,4'-tri bromodiphenyl ether), BDE 47 (2,2' ,4,4'-tetra bromodiphenyl ether), BDE 66 (2,3',4,4'-tetra bromodiphenyl ether), BDE 71 (2,3',4',6-tetra bromodiphenyl ether), BDE 85 (2,2',3,4',4-penta bromodiphenyl ether), BDE 99 (3,3',4,4',5-penta bromodiphenyl ether), BDE 100 (3,3',4,4', 6-penta bromodiphenyl ether), BDE 138 (2,2',3,4,4',5'-hexa bromodiphenyl ether), BDE 153(2,2',4,4',5,5'-hexa bromodiphenyl ether), BDE 154 (2,2',4,4',5,6'-hexa bromodiphenyl ether), BDE 183 (2,2',3,4,4',5',6-hepta bromodiphenyl ether) , BDE 190 (2,3,3',4,4',5,6-hepta bromodiphenyl ether) and BDE 209 (2,2',3,3',4,4',5,5',6, 6'-hepta bromodiphenyl ether) and 2 recovery standards were made with calibration solution mixtures prepared at 8 different concentrations.

Detailed information about GC-MS operation parameters and Calibration is given in Table 3-8 and Table 3-9.

Table 3-8. GC-MS calibration parameters for PBDE analysis

Operation Parameters	
GC Column **Liner**	30m*250µm*0.25µm Nominal Film Thickness. 5% Phenyl Methyl Siloxane. HP 5MS, capillary column Deactivated Glass Cotton Splitless Glass Liner (Agilent Technologies)
Carrier Gas	Ultra Pure Helium, 99.999%, 1 ml/min
Injection Type	Splitless
Temperature of Injection Port	295 0C
Oven Temperature	100℃ (1 min), 8 ℃/min to 320 ℃ (6 min)
Injection Volume	1 µL
Detector	MS
Mass Spectometer	Electron impact, 70eV
Mass Spectometerquadropole Temperature	150 ℃
Mass Spectrometry Source Temperature	230 ℃

Table 3-9. GC-MS calibration parameters for PBDE analysis

PBDE isomers	Ions (m/z)	Retention time (min)	Linear range (ppm)	Regression coefficient (r^2)
BDE-17	246,248,406	11.614	5-100	0.997
BDE-28	246,248,406	14.002	5-100	0.998
BDE-47	486	14.378	5-100	0.998
BDE-66	486	16.655	5-100	0.997
BDE-71	486	16.978	5-100	0.999
BDE-85	404,564,566	17.327	5-100	0.995
BDE-99	404,564,566	18.848	5-100	0.997
BDE-100	404,564,566	19.375	5-100	0.998
BDE-138	644	20.312	5-100	0.994
BDE-153	644	20.847	5-100	0.999
BDE-154	644	21.557	5-100	0.998
BDE-183	562	22.485	5-100	0.997
BDE-190	564	23.598	5-100	0.997
BDE-209	797	28,372	25-500	0.995

3.4.1.4 Sample extraction and enrichment

A crucial stage in sample preparation before chromatography is extraction. Target analytes must be separated from a complicated sample or significantly greater sample volume. Interfering sample components that could clog GC and HPLC columns are eliminated during the process.

In this study, for extraction and enrichment, from the samples in the deep freezer, 1 gr was weighed on a precision scale and placed in glass tubes for PBDE, PCB and PAH analyses. Recovery standards were injected onto the weighed dust samples for quality control and the recovery rates were 88.73 ± 14% (Asenaftin-d10), 80.98 ± 17% (Phe-d10), 81.90 ± 12% (Perylene-d12), 90.50 ± 19%. (Chr-d12), 92 ± 11% (BDE15), 68 ± 6 (BDE128), 85.50 ± 9 (PCB 14) and 75.40 ± 10 (PCB 65). Then, 10 ml of acetone:hexane (1:1) solution was added to the tubes, and the mouths of the tubes were closed with Teflon tape. It was kept at room temperature and in the dark for a total of two nights.

After the first night, the liquid portion of the samples, which were exposed to an ultrasonic bath with the heating feature turned off for 1 hour, was taken into amber bottles using a pasteur pipette and stored in the refrigerator. 10 ml of acetone:hexane (1:1) solution was added to the samples remaining in the glass tube for the second time and kept in the dark at room temperature for another night. After the second night, the liquid part of the samples, which were exposed to an ultrasonic bath with the heating feature turned off for 1 hour, was combined with the liquid part of the first night. At the end of two nights, the combined liquids were transferred to clean centrifuge tubes and centrifuged for 10 minutes at 3500 revolutions/min. Thus, it was cleared of precipitable particles. Then, the sample was taken from the centrifuge tube into a clear amber glass flask using a pasteur pipette and pre-enriched in a rotary evaporator to reduce the volume to 4 ml.

3.4.1.5 Clean-up Procedures

A column cleaning (clean-up) procedure was applied to remove other contaminants that might interfere with the analysis of target contaminants in the samples. For the preparation of the cleaning columns, alumina and silicic acid were used to remove organic pollutants in the samples obtained as a result of pre-enrichment, and sodium sulfate was used to retain water and moisture.

- **Preparation of chemicals to be used in the Cleaning Column**

Some necessary preparations were made for the cleaning of the materials used in the column construction and the activation of the chemicals.

The glassware used was washed with hot water and laboratory detergent that did not contain organic materials and rinsed thoroughly. Then it was rinsed with deionized water. After the rinsing process, all glassware was washed again with acetone and hexane, dried in an oven at 110 °C for 4 hours and made ready for the experiment. To clean the glass

wool, high purity hexane was added to the beaker cleaned in the previous stage, covering the glass wool, and kept in the fume hood until the hexane evaporated. The dried glass wool was placed in an amber bottle with a lid, thus preventing contact with air and making it ready for use. The required amount of silicic acid was taken into a clean container and activated by keeping it in an oven at 130 ℃ for 16 hours and cooled in a desiccator for 1 hour. The cooled silicic acid was placed in a clean volumetric flask and deactivated by 5% with ultrapure water. It was shaken well to avoid clumping and kept in the dark for 1 hour. Silicic acid, which was ready for use, was used in the column process within 12 hours. The amount of alumina needed for activation of alumina was taken into a clean container and activated by keeping it in a muffle furnace at 350℃ for 6 hours and cooled in a desiccator for 1 hour. The cooled alumina was placed in a clean volumetric flask, deactivated by 6% with ultrapure water, shaken well to prevent lumps, and kept in the dark for 1 hour. It was used in the colon procedure within 12 hours. The amount of sodium sulfate needed to dewater was taken into a clean container and activated by keeping it in a muffle furnace at 350℃ for 6 hours and cooled in a desiccator for an one hour. It was used in the colon procedure within 12 hours.

Finally, a glass column with a length of 15 cm and a diameter of 1 cm was used to prepare the clean-up column. A precision balance was used for column preparation, and 0.1 gr of glass wool, 3 gr of silicic acid, 2 gr of alumina and 1 gr of sodium sulfate were weighed from bottom to top, respectively, and poured into the column and made ready.

- **Clean-up and final enrichment**

The cleaning column used for PAH, PCB and PBDE separation was first washed with 20 ml dichloromethane and then with 20 ml petroleum ether. Then, the sample whose volume was reduced to 5 ml in the pre-enrichment stage was added. Then, 60 ml of petroleum ether was passed through the column to remove PCB and PBDE contaminants from the column and the sample was placed in an amber bottle, and immediately afterwards, 40 ml of diclothomethane was passed through the column to remove PAH contaminants.

The volumes of the first and second fractions taken from the column were reduced to approximately 4 ml by a rotary evaporator with the heating feature turned off. Samples whose volume decreased to 4 ml were taken into clean 5 ml amber vials with the help of pasteur pipettes, and the hexane solvent was changed by adding hexane 4 times under

pure nitrogen gas, and enrichment was performed by reducing the volume to 200 µl. Since PAH contaminants were separated from the column together in the first and second fractions, equal volumes (100 µl each) of the first and second fractions were mixed in the vial before PAH analysis and thus ready for reading. PCB and PBDE analysis was performed on the remaining 100 µl of the first fraction.

3.4.2 Alkylphenol and alkylphenol ethoxylates analysis

3.4.2.1 Chemicals and reagents

All solvents were HPLC grade. Acetonitrile and acetone were from Isolab (Eschau, Germany), MeOH was from Honeywell Research Chemicals (North Carolina, USA) and hexane was from Merck (Darmstadt, Germany). Octylphenol (OP) was from Accustandard (New Haven, USA), 4-Octylphenol Monoethoxylate (4-OPME) was from Toronto Research Chemicals (North York, ON, Canada), 4-tert-octylphenol (4-tOP) and 4-n-nonylphenol (4-n-NP) were from Dr. Ehrenstorfer (Augsburg, Germany) while nonylphenol diethoxylate (di-NPE) was from Sigma Aldrich (Darmstadt, Germany). Recovery compounds bis(2- ethylhexyl)phthalate-3,4,5,6-d4, dicyclohexyl phthalate-3,4,5,6-d4, diethyl phthalate-3,435,6-d4, dimethyl phthalate-3,4,5,6-d4 and diisobutyl phthalate-3,4,5,6-d4 were procured from Accustandard while bisphenol A-d16 was from Dr. Ehrenstorfer.

3.4.2.2 Dust samples preparation

Dust sample (20 mg) was placed in a conical bottom glass tube and 1 mL of 1:1 acetone:hexane mixture was added. Samples were kept overnight and the extraction process was then carried out for 30 min in an ultrasonic bath (Ultrasonic bath operating at 85 % power, temperature 25 ◦C). After centrifugation at 1500 rpm for 5 min, the supernatant was transferred into another glass tube. The extraction process was repeated once more using 1 mL of acetone: hexane mixture and then both extracts were combined. The extract was evaporated to dryness under a gentle nitrogen gas stream and dissolved in 0.2 mL Methanol, transferred in a LC-MS/MS vial and the volume was topped up to 1 mL in methanol. Di-isobutyl phthalate3,4,5,6-d4 (50 ng) was added to each sample as an internal standard.

3.4.2.3 Instrumental analysis

Analysis of compounds were performed using a Shimadzu 8040 triple quadrupole LC-MS/MS system. 10 µL volume of the extract was injected and the chromatographic separation was achieved on a Shimpack FC-ODS (150 × 2 mm, Shimadzu, Kyoto/Japan) column. 10 mM ammonium acetate in water (solvent A) and acetonitrile (solvent B) were used as mobile phases. The gradient program was set as follows: 0.0–1.0 min; 80 % of solvent B, 1.0–5.0 min; a linear gradient to 100 % of solvent B; 5.0–13.5 min 100 % solvent B; 13.50–13.51 min gradient to 75 % solvent B; 16.0 min stop. The column flow rate and oven temperature were set 0.3 mL/min and 40 ◦C, respectively. The capillary voltage was kept with 4.0 kV, and vaporizer temperature was 350 ◦C. The m/z ions used for the quantification are presented in

Table 3-10.

Table 3-10. MRM conditions and retention times for target chemicals

Analytes	CAS Number	MRM Transitions			RT (min.)
		Q (m/z)	q (m/z)	CE (V)	
4-t-OP	140-66-9	205.1	133	-28	4.38
			117	-24	
4-n-NP	104-40-5	274.2	70.15	-28	4.79
			88.1	-24	
(di-NPE)	200-662-2	326.2	121.1	-22	5.46
			71.1	-21	

MRM: multiple reaction monitoring; Q: quantifier ion; q: qualifier ion; CE: collision energy, RT: retention time

3.4.3 Quality assurance and control

In analytical laboratories, the importance of QA/QC has grown. QA/QC is crucial in order to compare the results with those from other laboratories across the globe. The investigation used analytical grades of standards and solvents that were all acquired from reputable vendors like Sigma Aldrich Inc. After being cleaned using the standard washing procedures, glassware, including centrifuge tubes and Pasteur pipettes, was roasted at 150℃ for at least four hours. All glassware, including glass syringes, were cleaned before use by rinsing them in acetone and hexane. This process was followed by ultrasonicating the items in acetone for approximately half an hour.

To prepare the blank samples (n=12), a combination of solvents was used only for sample extraction. After that, these blank samples underwent the same preparation and analysis as the dust samples. Since no target substances were found in the process or solvent

blanks, LODs were determined by extrapolating the concentration at which injecting the extracted spiked sample of lowest concentration would result in a signal-to-noise ratio of 3:1 (S/N = 3). The method detection limit (MDL) was calculated as follows: MDL = average concentration of target chemical in blank+3*std dev. Table 3-11 displays LODs (in ng/µL) and MDLs (in ng/g d. w.).

Table 3-11. Limits of Detection (ng/µl) and Method Detection Limits (ng/g d.w.) of target analytes

PAHs	LOD	MDL	PCBs	LOD	MDL	PBDEs	LOD	MDL	Alkylphenols	LOD	MDL
Nap	0.020	0.040	PCB28	0.001	0.002	BDE15	0.000	0.001	4-t-OP	0.003	0.005
Acy	0.001	0.003	PCB52	0.001	0.002	BDE17	0.001	0.002	4-NP	0.001	0.002
Ace	0.008	0.016	PCB101	0.001	0.002	BDE28	0.001	0.002	di-NPE	0.002	0.004
Flu	0.002	0.004	PCB118	0.001	0.002	BDE47	0.002	0.005			
Phe	0.001	0.003	PCB138	0.001	0.002	BDE66	0.002	0.003			
Ant	0.002	0.004	PCB153	0.001	0.002	BDE71	0.002	0.003			
Flt	0.002	0.005	PCB166	0.002	0.004	BDE85	0.003	0.005			
Pyr	0.001	0.003	PCB180	0.001	0.002	BDE99	0.003	0.005			
BaA	0.003	0.006				BDE100	0.004	0.008			
Chr	0.000	0.001				BDE138	0.004	0.007			
BbF	0.015	0.030				BDE153	0.005	0.009			
BkF	0.000	0.001				BDE154	0.002	0.004			
BaP	0.000	0.000				BDE183	0.002	0.004			
Ind	0.000	0.000				BDE190	0.002	0.005			
DahA	0.001	0.003				BDE209	0.064	0.128			
BgP	0.002	0.004									

A recovery standard was added to all samples to find the recovery efficiency before the extraction process. The analytical recycling efficiencies of the recovery standard are 88.73 ± 14% (Asenaftin-d10), 80.98 ± 17% (Phe-d10), 81.90 ± 12% (Perylene-d12), 90.50 ± 19% (Chr-d12), 92 ± 9%, 11% (BDE15), 68 ± 6 (BDE128), 85.50 ± 9 (PCB 14) and 75.40± 10 (PCB 65). Due to financial constraints in our laboratory facilities, deuterated or ^{13}C labelled APs or APEs could not be obtained. Therefore, as recovery surrogate chemicals, 100 ng each of dicyclohexyl phthalate-3,4,5,6-d4 and dimethyl phthalate-3,4,5,6-d4, which are thought to be representative of the chemicals of interest in terms of molecular weight, were used. The recovery ratio for dimethyl phthalate-3,4,5,6-d4 was 77.5±8.56%, but the recovery ratio for dicyclohexyl phthalate-3,4,5,6-d4 ranged between 87.8±9.80%.

Organic pollutants in house dust reference - SRM2585 obtained from the American National Institute of Standards and Technologies (NIST) was used for method validation.

All procedures applied to the samples were also applied to the reference material. Measured values, certificate values and % recovery results of the elements in the reference material are given in Table 3-12. As depicted in Table 3-12, the recovery rates range from 65% for Dibenzo(a,h)anthracene (DahA) to 120% for Benzo(g,h,i)perylene (BgP).

The analyte recoveries of alkylphenols were examined using spiking. 150 ng of APs and APEs each were added to 1 milliliter of the extraction solvent combination. The six spike samples underwent the same treatment as the other samples. Following is the average recovery ratio for substances that were spiked: The results show that the percentages for, 4-t-OP, 4-NP, and di-NPE were 99.7%, 87.7%, and 92.6%, respectively.

Table 3-12. Recoveries of the elements in the reference material (SRM2585)

	Certificate values (ng/g)	Measured values (ng/g)	Recovery (%)
Naphthalene (Nap)	0.266	0.231	87%
Anthracene (Ant)	0.096	0.078	81%
Phenanthrene (Phe)	1.92	1.461	76%
Pyrene (Pyr)	3.29	3.04	92%
Fluoranthene (Flt)	4.38	3.705	85%
Chrysene (Chr)	2.26	1.532	68%
Benzo(a)anthracene (BaA)	1.16	1.023	88%
Benzo(b)fluoranthene (BbF)	2.7	2.45	91%
Benzo(k)fluoranthene (BkF)	1.33	1.51	113%
Benzo(a)pyrene (BaP)	1.14	1.13	99%
Indeno(1,2,3-c,d)pyrene (Ind)	2.08	1.93	93%
Dibenzo(a,h)anthracene(DahA)	0.31	0.19	65%
Benzo(g,h,i)perylene (BgP)	2.28	2.73	120%
PCB 28 (2,4,4′-trichlorobiphenyl)	13.4	10.58	79%
PCB 52 (2,2′,5,5′-tetrachlorobiphenyl)	21.8	18.53	85%
PCB 101 (2,2′,4,5,5′-pentachlorobiphenyl)	29.8	24.43	82%
PCB 118 (2,3′,4,4′,5-pentachlorobiphenyl)	26.3	18.93	72%
PCB 138 (2,2′,3,4,4′,5′-hexachlorobiphenyl)	27.6	28.98	105%
PCB 153 (2,2′,4,4′,5,5′-hexachlorobiphenyl)	40.2	31.7	79%
PCB 180 (2,2′,3,4,4′,5,5′-heptachlorobiphenyl)	18.4	15.27	83%
BDE 17 (2,2′,4-tribromodiphenyl ether)	11.5	10.58	92%
BDE 28 (2,4,4′-tribromodiphenyl ether)	46.9	47.36	101%
BDE 47 (2,2′,4,4′-tetrabromodiphenyl ether)	497.1	442.33	89%
BDE 66 (2,3′,4,4′-tetrabromodiphenyl ether)	29.5	28.02	95%
BDE 85 (2,2′,3,4,4′-pentabromodiphenyl ether)	43.8	35.91	82%
BDE 99 (2,2′,4,4′,5-pentabromodiphenyl ether)	892.1	802.8	90%
BDE 100 (2,2′,4,4′,6-pentabromodiphenyl ether)	145.1	147.9	102%
BDE 138 (2,2′,3,4,4′,5′-hexabromodiphenyl ether)	15.2	11.85	78%
BDE 153 (2,2′,4,4′,5,5′-hexabromodiphenyl ether)	119.1	97.58	82%

BDE 154 (2,2′,4,4′,5,6′-hexabromodiphenyl ether)	83.5	72.64	87%
BDE 183 (2,2′,3,4,4′,5′,6-heptabromodiphenyl ether)	43.1	31.39	73%
BDE 209 (decabromodiphenyl ether)	**2510.1**	**2334.3**	**93%**

3.4.4 Statistical analysis

All statistical analyses were performed with IBM SPSS software version 23.0 for Windows (SPSS Inc., USA). Shapiro-Wilk test was used to test whether the data were normally distributed. Since none of the parameters showed a normal distribution, all evaluations were made with non-parametric methods. Target chemicales from residential dust were compared between groups with various architectural qualities, family-related data, and lifestyle behaviors using the non-parametric Mann-Whitney U test. Descriptive statistics, frequency, percentage distributions, percentiles, considering that continuous data are not normally distributed, median (minimum, maximum) values are presented instead of arithmetic mean±standard deviation.

Spearman correlation test, Kruskal-Wallis analysis of variance and Positive matrix factorization (PMF) model (PMF 5.0, USEPA) were used to determine the possible sources of selected EDCs analyzed in this study in house dust.

Before performing the Kruskal-Wallis test or PMF, you might explore relationships between variables using correlation analysis. This can help identify which variables might be grouped together or influence each other.

The Kruskal-Wallis test is a non-parametric method used to determine whether there are statistically significant differences between two or more groups. In this case, it seems like the test was applied to assess whether there are differences in the concentrations of target pollutants isomers in dust collected from homes based on various factors related to household conditions and living habits, as reported in the survey.

PMF is a multivariate factor analysis tool used to decompose a matrix of observed data into contributions from several underlying factors. After determining significant differences using the Kruskal-Wallis test, PMF can be applied to understand the underlying factors contributing to these differences (Details are given in section 3.4.3.1).

By combining these methods, you gain a comprehensive understanding of the data, identifying significant differences and uncovering underlying factors contributing to those differences.

Estimation of associations between the development of asthma and selected EDCs in dust was performed with a binary logistic regression model based on adjustments for significant covariates. Items identified by the survey as potential risk factors for childhood asthma were included in the models to control for confounders. Associations were evaluated by odds ratio (OR) and 95% confidential interval (CI) calculations (Details are given in section 3.4.3.2).

3.4.4.1 Positive matrix factorization (PMF) model

Positive matrix factorization (PMF) is a bilinear model with non-negativity constraints. A matrix of speciated sample data is broken down into two matrices using PMF, a multivariate factor analysis tool: factor contributions (G) and factor profiles (F). The user must interpret these factor profiles using measured source profile data and emissions or discharge inventories to determine the source types that might be contributing to the sample. The restriction that no sample can have considerably negative source contributions is used to generate the results.

The PMF 5.0 model produced by the United States Environmental Protection Agency (USEPA) was used to identify the sources of chemicals in dust samples. Two input files are required to run the model: the concentrations and uncertainty (Unc) values of each species [179]. With the use of this function, analysts can take measurement confidence into consideration. There were two scenarios for the uncertainty computation. The uncertainty was determined using Eq. (1) for chemicals with a concentration ≤ MDL (method detection limit), and Eq. (2) was applied if the concentration exceeded MDL [180].

$$\text{Unc} = \frac{5}{6}\,\text{MDL} \qquad \text{Eq. (1)}$$

$$\text{Unc} = \sqrt{(concentration\ x\ error\ fraction)^2 + (0.5\ x\ MDL)^2} \qquad \text{Eq. (2)}$$

To maximize the factor contributions and profiles in PMF under nonnegative restrictions, the objective function (Q) and signal-to-noise ratio (S/N) are used [181]. In case the S/N ratio exceeded 0.5 but remained below 1.0, the chemical species was deemed as a "weak" source contributor. Chemical species were categorized as "weak" if the S/N ratio was

less than 0.5 and as "strong" if it was greater than 1 [180]. In this study, all species of PAHs, PCBs , and BDEs were classified as “strong”. The scale of residual values for each PAH species in this study was between − 3 and +3, indicating that the PMF model matches the input data well.

By minimizing the objective function Q, the PMF model derives factor contributions and profiles. For PMF, Q is an essential parameter, and is determined using Eq. (3).

$$Q = \sum_{i=1}^{n} \sum_{j=1}^{m} \left[\frac{x_{ij} - \sum_{k=1}^{p} g_{ik} f_{kj}}{u_{ij}} \right]^2 \qquad \text{Eq. (3)}$$

The order of operations is essentially how the tabs and functions in the program are arranged (from left to right) is given in Figure 3-2. To begin using the program, the user must provide input files via the Model Data - Data Files screen before other operations are available. The first time PMF is performed on the data set, the user should analyze the input data via the Concentration/Uncertainty, Concentration Scatter Plot, Concentration Time Series, and Data Exceptions screens. Under the Base Model tab, Base Model Runs and Base Model Results often come after this step. These stages should be repeated as necessary until the user finds a workable solution. The Error Estimation choices are used to evaluate the solution, starting with DISP and moving on to BS and BS-DISP. The output of these methods of error estimation—DISP, BS, and BS-DISP—provides important details about the stability of the solution. It takes the use of all three error estimating techniques to comprehend the degree of uncertainty surrounding the solution.

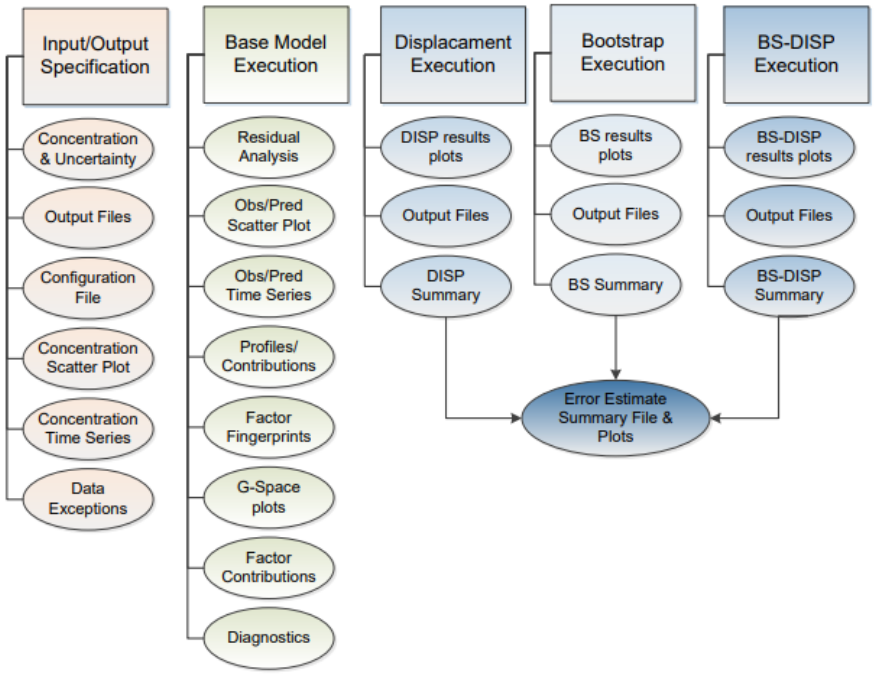


Figure 3-2. Flow chart of operation within EPA PMF- Base Model [180].

The method is briefly reviewed here, you can find detailed information in the pmf-5.0 user guide published by US EPA [37].

Numerous types of data have been subjected to PMF analysis, including size-resolved aerosol, air pollutants, 24-hour speciated PM2.5, high-time resolution observations from devices like aerosol mass spectrometers (AMS), and volatile organic compound (VOC) data [180]. In addition, the PMF model has been used in many studies to identify sources of indoor air pollutants [179,182]. In this study, PMF was preferred as a reliable analysis to determine possible sources of PAHs, PCBs and BDEs in the home environment.

3.4.4.2 Binary Logistic Regression Model

Binary logistic regression is a statistical method used to model the relationship between a binary dependent variable (outcome) and one or more independent variables (predictors). The dependent variable is dichotomous, meaning it takes on two possible outcomes. The primary purpose of binary logistic regression is to predict the probability that a given observation falls into one of the two categories of the dependent variable based on the values of the independent variables [183].

Binary logistic regression is one method that is particularly appropriate for analyzing survey data in the widely used cross-sectional and case–control research designs [175,184,185].

The logistic regression model estimates the log odds of the dependent event occurring. The model is given as follows:

$$\log\left(\frac{P(Y=1)}{1-P(Y=1)}\right) = \beta_0 + \beta_1 X_1 + \ldots + \beta_k X_k \qquad \text{Eq. (4)}$$

Where:

- P (Y=1) is the probability of the event occurring (e.g., success, yes, 1).
- β_0 is the intercept.
- $\beta_1, \beta_2, \ldots, \beta_k$ are the coefficients for the independent variables $X_1, X_2, \ldots, X_k$

Steps in Binary Logistic Regression:

1. **Specify the Model:**

- Identify the binary dependent variable.
- Select the independent variables (predictors).

2. **Estimate the Model Parameters:**

- Use a method such as Maximum Likelihood Estimation (MLE) to estimate the coefficients.

3. **Assess the Model Fit:**

- Use statistical tests (e.g., Wald test, Likelihood Ratio Test) and goodness-of-fit measures (e.g., Hosmer-Lemeshow test, pseudo-R-squared values) to evaluate the model.

4. **Interpret the Coefficients:**

- The coefficients represent the change in the log odds of the dependent variable for a one-unit change in the predictor variable.
- Exponentiating the coefficients gives the odds ratios, which are easier to interpret.

Assessing Model Fit:

- Goodness-of-fit tests: Hosmer-Lemeshow test, which assesses if the observed event rates match expected event rates in subgroups of the model population.
- Pseudo R-squared: Measures like McFadden's R-squared to indicate how well the model explains the variability of the outcome.

Logistic regression is actually a classification algorithm. The objective of the algorithms is to determine the decision borders between the classes in supervised classification situations, when the classes are discrete. Decision boundaries demarcate classes from their examples. Decision boundaries can have complicated, nonlinear geometric shapes depending on the specifics of the problem.

The feature weights are approximately equivalent to the parameters of the logistic regression model. The S-shaped logistic function is used to map each weighted feature vector to a value between 0 and 1. This number represents the likelihood that an example falls into a specific class. For the learning instances to be correctly classified, the learning algorithm adjusts the weights. For adjusting the weights, the gradient descent method and its various variations are widely used. The logistic function is then used to each unseen example to determine its chance of belonging to a class once the weights have been determined. Frequently, logistic regression is the first technique used for classification issues because of the oversimplified assumption of linear decision boundaries. As a conclusion, Binary logistic regression is a powerful tool for binary classification problems, allowing for the modeling and prediction of outcomes based on multiple predictors. By interpreting the coefficients and predicting probabilities, you can gain insights into the factors influencing the likelihood of an event and make informed decisions based on the model's predictions.

4 RESULTS AND DISCUSSION

In this section, the results of a face-to-face survey conducted with families of asthmatic and non-asthmatic children who voluntarily participated in the study, the levels, sources and mechanisms of endocrine disrupting chemicals (EDCs) in the collected house dust samples are reviewed and the effects of exposure to these chemicals on the prevalence and severity of asthma in children are reviewed. Evidence that it affects many aspects is being examined.

4.1 Evaluation of Survey Questions

The survey results were analyzed using SPSS 23.0 (SPSS Inc., Chicago, IL, USA) and underwent additional statistical analysis. The variables include residential characteristics and the lifestyle of occupants. The answers to the survey questions collected from both the asthma and control groups were compared, and the results are summarized in Table 4-1.

Table 4-1. Characteristics of study population

		Case	**Control**	***P* value**
		Number (%)	Number (%)	
Gender	Female	55 (50.5)	65 (50)	0.688
Age, years	Median (IQR)	8 (7-10)	8.5 (7-10)	0.366
Level of House	≤1 Floor	36 (33)	78 (60)	**<0.001**
	≥2 Floor	73 (67)	52 (.40)	
Dwelling age	≤20 Years	63 (57.8)	93 (71.5)	**0.019**
	>20 Years	46 (42.2)	37 (28.5)	
Residential area (m^2)	≤100	31 (28.4)	44 (33.8)	0.481
	>100	78 (71.6)	86 (66.2)	
Flooring material	Laminate	97 (89)	109 (83.8)	0.169
	polished wood	12 (11)	21 (16.2)	
Carpet type	Synthetic	99 (90.8)	120 (92.3)	0.681
	Wool	10 (9.2)	10 (7.7)	
Type of wall paint	Plastic Paint	72 (66.1)	97 (74.6)	**0.043**
	Oil Paint	13 (11.9)	9 (6.9)	
	Whitewash	10 (9.2)	18 (13.8)	
	Wallpaper	14 (12.8)	6 (4.6)	
Separate kitchen (Yes)		97 (89)	127 (97.7)	**0.006**
Near main street	No	32 (29.4)	38 (29.2)	0.548
	Yes	77 (70.6)	92 (70.8)	

		Case Number (%)	Control Number (%)	*P* value
Number of occupants	≤4	92 (84.4)	93 (71.5)	**0.013**
	>4	17 (15.6)	37 (28.5)	
Heating system	Natural gas	70 (64.2)	104 (80)	**0.005**
	Central heating	37 (33.9)	24 (18.5)	
Frying frequency	Once a week	56 (51.4)	42 (32.3)	**0.002**
	Twice a week or more	53 (48.6)	88 (67.7)	
Smoking at home	No	46 (42.2)	74 (56.9)	**0.016**
	Yes	63 (57.8)	56 (43.1)	
Repairs and painting done in the last year		25 (22.9)	41 (31.5)	0.138
New furniture		40 (36.7)	29 (22.3)	**0.011**
Pet keeping		13 (11.9)	36 (27.7)	**0.002**
Cleaning frequency	Once a week or less	42 (38.5)	35 (26.9)	**0.038**
	Twice a week or more	67 (61.5)	95 (73.1)	
Frequency of using bleach	Once a week or less	47 (43.1)	38 (29.2)	**0.018**
	Twice a week or more	62 (56.9)	92 (70.8)	
The material of the child's mattress	Viscos elastic	84 (77.1)	88 (67.7)	**0.026**
	Cotton	14 (12.8)	12 (9.2)	
	Wool	11 (10.1)	30 (23.1)	
Air conditioning	No	125 (93.6)	125 (96.2)	0.364
	Yes	7 (6.4)	5 (3.8)	

Participants in the asthma group were more likely to reside on the 2nd floor and above (67%) compared to the control group (40%). According to the data, 42.2% of families in the asthma group live in houses built over 20 years, compared to 28.5% in the control group. Both groups preferred to live in houses larger than 100 m^2 in terms of the area of the house, and laminate flooring and synthetic carpet were predominantly used. The survey results indicate that plastic paint was the most preferred type of paint for walls, and oil paint is more commonly used in the asthma group (11.9%) compared to the control group (6.9%). Additionally, wallpaper was used more frequently in the asthmatic group compared to the control group (12.8% for asthma and 4.6% for control group). A statistical difference was found between the two groups in terms of the presence of a kitchen independent of the living room at home. This difference was higher in the control group (97.7%) compared to the asthma group (89%). Based on the results, it was found that houses in both groups were situated near the main street.

The surveys included not only the characteristics of the houses but also the lifestyles of the families. One of the factors that may trigger asthma symptoms is the number of people living in the house. In families with children with asthma, the rate of households with four or fewer people was 84.4%, compared to 71.5% in the control group. Additionally, the rate of households with more than four people was 28.5% for the control group and 15.6% for the asthma group. Both groups preferred natural gas for heating, and while the rate of houses using central systems was 33.9% in the asthma group, this rate was 18.5% in the control group. The frequency of frying at home is an important risk factor, especially in homes that do not have an independent kitchen. In the asthma group, 48.6% of households engage in frying twice a week or more, while this proportion is 67.7% in the control group. One of the very important factors that increases the severity of asthma is cigarette smoke. Even though the families of children with asthma are aware of this issue, they continue to smoke during the day, even on the balcony of the house. The smoking rate of the families participating in this study at home (including the balcony) was 57.8%. On the other hand, the proportion of families who did not smoke at home was 42.2%. For the control group, these rates were 43.1% and 56.9%, respectively. In the last year, new furniture was purchased for the homes of 36.7% of children with asthma. In comparison, this rate is 22.3% for children without asthma. Families in both groups keep pets such as cats, dogs, birds, and goldfish. In the group of children with asthma, this rate was 11.9%, compared to 27.7% in the control group. Regarding the frequency of house cleaning, houses that are cleaned twice a week or more have higher rates in both groups (61.5% in the asthma group and 73.1% in the control group). However, this rate is higher in the control group. Families in the asthma group were aware of the impact of bleach on asthma severity, leading to a reduced frequency of its use in cleaning.

In both groups, elastic viscose was preferred as the material for children's mattress due to its affordability and practicality. Families of children in the asthma group preferred wool bedding less, adhering to the recommendations of doctors (10.1% in the asthma group and 23.1% in the control group).

In Ankara, homes generally use less air conditioning, and this trend is observed in both groups included in this study. Instead, natural ventilation is preferred.

4.2 Detection frequencies and levels of selected EDCs in indoor dust:

The detection frequencies (DFs) of all targeted analytes were calculated based on instrument analyses, and the results are summarized in Table 4-2.

Table 4-2 . The detection frequencies (DFs) and the levels of all targeted analytes

	DF	25th	Median	75th	SD	Min	Max
PAHs							
Nap	66	130.7	170.9	254.8	138.5	42.1	854.6
Ace	66	92.6	119.9	173.6	121.1	15.3	1167
Flue	66	120.9	159.7	246.8	114.5	35.9	705.9
Acy	67	74.1	95.3	115.4	78.7	13.2	474.5
Phe	66	101.3	191.6	362.3	162.1	26.4	1023
Ant	65	99.4	156.1	224.8	106.7	12.7	814.7
Flt	100	185.1	262.8	304.5	128.8	9.3	980.4
Pyr	95	146.1	201.1	237.3	97.6	7.2	753.1
BaA	95	100	101.6	149.4	50.2	1.6	300.2
Chr	100	190.6	203.3	251.7	83.5	6.4	551.1
BbF	93	49.6	93.5	206.4	139.4	13.9	935.2
BkF	95	49.9	93.5	198.1	144.1	1.9	951.5
BaP	92	31.6	67.1	119.1	252.2	6.6	1302
Ind	91	39.3	102.1	122.8	60.7	8.9	311.1
DahA	96	14.2	28.3	44.8	59.6	1.4	646.1
BgP	95	36.8	93.8	159.4	96.1	2.1	590.1
LMW		636.2	934.6	1309	621.7	171.1	4374
HMW		1002	1301.7	1629	609.6	75.4	3918
∑ PAHs		1329	1970.2	2705	928.3	461.8	5215
PCBs							
PCB28	100	0.6	1.2	3.1	2.8	0.1	19.4
PCB52	100	0.5	1.3	2.4	1.6	0.3	8.2
PCB101	100	4.3	8.5	14.2	7.5	0.4	39.8
PCB118	100	1.5	4.9	10.9	5.9	0.1	12.9
PCB138	100	4.8	8.3	14.7	8.3	0.2	71.2
PCB153	100	2.8	5.8	8.5	4.4	0.3	27.7
PCB166	100	1.6	0.1	6.4	4.7	0.1	32.3
PCB180	100	0.2	0.4	1.3	2.2	0.04	18.8
Cl3	100	0.7	1.2	3.1	2.8	0.06	19.4
Cl4	100	0.6	1.3	2.4	1.6	0.03	8.2
Cl5	100	7.8	14.1	24.7	12.1	0.7	7.8
Cl6	100	12.2	18.3	29.1	13.3	1.2	77.3
Cl7	100	0.2	0.4	1.3	2.2	0.04	18.8
∑ PCBs	100	48.8	68.4	114.6	49.3	11.7	264.7
PBDEs							
BDE15	100	0.05	0.1	0.2	0.5	0.04	4.8
BDE17	100	0.2	0.3	0.6	1.1	0.03	10.4
BDE28	100	0.2	0.3	0.7	2.2	0.07	18.3
BDE47	100	0.2	0.4	0.7	2.5	0.02	16.1
BDE66	100	0.1	0.2	0.6	2.3	0.03	20.7
BDE71	100	0.2	0.4	0.8	3.3	0.04	25.7
BDE85	100	0.2	0.6	1.7	2.5	0.04	22.1
BDE99	100	3.5	6.7	11.6	8.4	0.1	63.3
BDE100	100	1.9	2.9	4.3	9.7	0.2	71.5
BDE138	100	0.4	1.1	4.5	11.2	0.01	101.1
BDE153	100	1.7	3.6	6.3	6.1	0.01	46.1
BDE154	100	2.1	4.8	7.8	7.8	0.02	76.7
BDE183	100	12.9	26.5	44.7	14.7	1.2	162.4
BDE190	100	2.4	5.4	12.6	14.7	0.04	109.1
BDE209	100	10.1	24.6	517.5	536.7	0.2	2348
Penta	100	10.8	16.6	29.1	58.7	2.1	522.8
Octa	100	26.9	44.1	72.4	38.9	5.6	240.9
Deca	100	10.1	24.6	517.5	536.7	0.2	2348
∑PBDEs	100	66.2	431.6	987.6	566.8	16.5	2544
PHENOLS							
4-n-nonylphenol	65	310	739	1323	1021	49	8737
4-tert-octylphenol	63	19	31	59	45.5	8	268
di-NPE	67	1215	1965.5	3575	1881	113	9072

As shown in the Table 4-2, sixteen PAHs were detected among 240 indoor dust samples with the range of detection frequency (DF) and the range of median concentration were 65-100% and 28.3 (DahA) - 262.8 ng/g (Flt), respectively. Following Flt, the highest PAH isomers measured in the collected dust were detected for Chr with a median value of 203.3 ng/g and Pyr with a median value of 201.1 ng/g, respectively. The lowest detection frequencies (DF) were observed for low molecular weight PAHs. 2- and 3-ring PAH isomers, which are more volatile and have lower molecular weight, are less abundant in dust than in air [186].

The median value of total PAHs was 1970 ng/g (461.8 - 5215 ng/g). This value was lower than the values measured in China [186,187], Canada [188], Australia [189] Germany (Berlin)[190], Saudi Arabia [191], Türkiye (Kocaeli) [192], Greece [193], Kuwait [194], and higher than the concentrations measured in Türkiye (Ankara) [195], Vietnam [196] , and USA [197] (Details are given in section 4.3 and Figure 4-3).

As given in Table 4-2, 100% DFs were observed in both PCB and BDE groups. Based on the results, the highest measured PCB in indoor dust was PCB101, with a value of 8.5 ng/g. PCB138 and PCB153 were detected as PCB isomers with high concentrations, measuring 8.1 and 5.8 ng/g, respectively. The least common isomer found in homes was PCB180, with a median value of 0.4 ng/g. ΣPCBs have a median value of 68.4 ng/g (11.7 ng/g-264.7 ng/g) in house dust. ΣPCBs have a median value of 68.4 ng/g (11.7 ng/g-264.7 ng/g) in house dust. While this value is lower than the ΣPCBs values measured in Canada [198], Texas [198], Hong Kong [199] and the Czech Republic [200], it is higher than the studies conducted in Guangzhou [201], Birmingham [198], New Zealand [198], Kocaeli [192], California [198], and Kuwait [202] (Details are given in section 4.4 and Figure 4-10. Comparison of total ∑ PCBs levels with other countries).

As a result, BDE-209 was measured as the highest pollutant with a median value of 26.6 ng/g. The PBDE isomer with the second highest concentration was identified as BDE-183 (hepta-BDE), which is the main component of commercial octa-BDE containing products, with a median value of 24.5 ng/g. The other pollutant with the highest concentration detected in the study was PBDE-99, with a median value of 6.7 ng/g. This is followed by PBDE-190, which has a median value of 5.4 ng/g, and PBDE-154, with a median value of 4.8 ng/g. While the $\sum_{15}$ BDE (431.6 ng/g) value measured in this study was lower than those measured in UK (Newcastle) [36], Türkiye (Istanbul) [203], Canada [204], and Türkiye (Kocaeli) [205], this was higher than the value measured in indoor

dust in Germany[36], and Kuwait [206] (Details are given in section 4.4 and Figure 4-18. Comparison of total PBDE levels with other countries).

The concentrations of 4-nonylphenol (4-n-NP), 4-nonylphenol diethoxylate (di-NPE), and 4-tert octylphenol (4-t-OP) in dust are presented in Table 4-2. The highest concentrations were observed for di-NPE in indoor dust, with a median value of 1910 ng/g (ranging from 113 to 9070 ng/g). Among the APs, 4-t-OP showed lower concentrations in all the settled dust (35 ng/g). There have been relatively few studies on the presence of alkylphenols (APs) and alkylphenol ethoxylates (APEs) in dust. In comparison to several studies documented in the literature, the concentrations of APs and APEs in indoor dust were found to be lower in our study [72–74,207,208] (Details are given in section 4.6 and Table 4-5. APs and APE concentrations (in ng/g) in indoor dust were compared to those found in other investigations in the current study.).

4.3 The levels and sources of PAHs in indoor dust

Concentrations of 16 PAHs in indoor dust samples are summarized in Table 4-2 and Figure 4-1. Levels of 16 PAHs of indoor dust. Among 16 PAHs, Flt was the most dominant with the median concentration of 262.8 ng/g, followed by Chr at 203.3 ng/g, and Pyr at 201.1 ng/g (11%). DahA was found to have the least value at 28.3 ng/g.

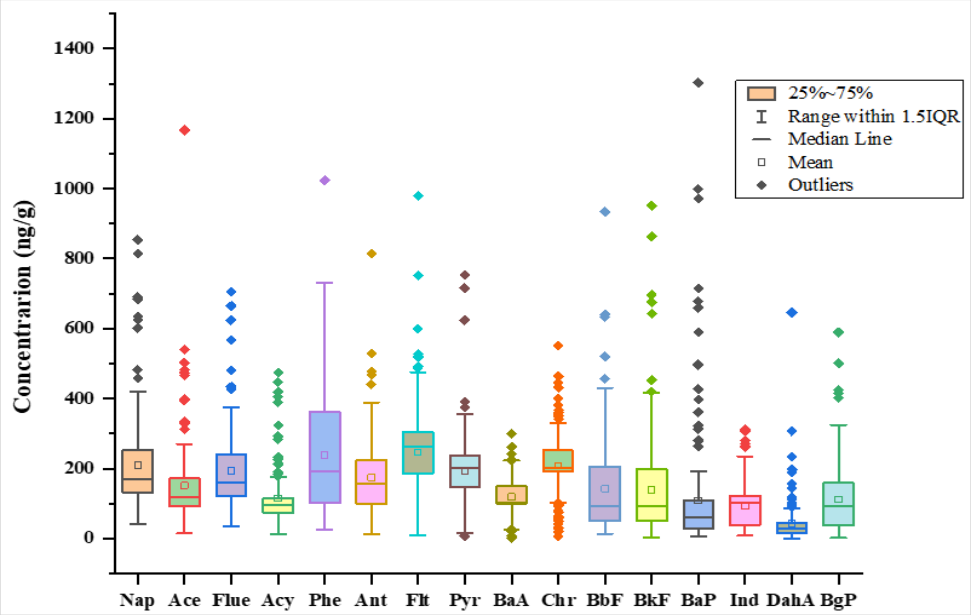


Figure 4-1. Levels of 16 PAHs of indoor dust.

The amount of $\sum_{16}$PAH was measured to be higher in houses with the following characteristics: age of the house over 20 years, proximity to the main street, indoor or balcony smoking, use of wood or coal as fuel, frying 2-3 times a week and frequent use of detergent for cleaning (more than 3 days a week).

The profile of the analyzed PAHs was described in Figure 4-2. Flt (11%), Chr (9%), and Pyr (9%) were determined as the dominant isomers in the $\sum_{16}$ PAHs. High molecular weight 4, 5 and 6 ring PAH isomers (BaA, BbF, BkF, BaP, IcdP, DahA, and BghiP) constitute 65% of the total measured PAH concentrations, and low molecular weight PAH isomers (Nap, Acy, Ace, Flu, Phe, Ant, Flt, Pyr, and Chr) constitute 45% of the total PAH amount. Due to their high volatility, LMW-PAHs were found to be highly concentrated in gaseous samples, whereas HMW-PAHs, which are more persistent and hazardous, were mostly identified in settled dust [191,209,210].

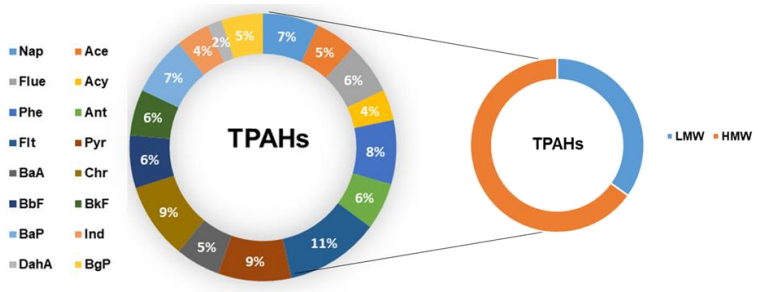


Figure 4-2. The profile of the analyzed PAHs

The results obtained from the PAH studies carried out on indoor dust samples from various parts of the world and the study conducted in Ankara are shown in Figure 4-3.

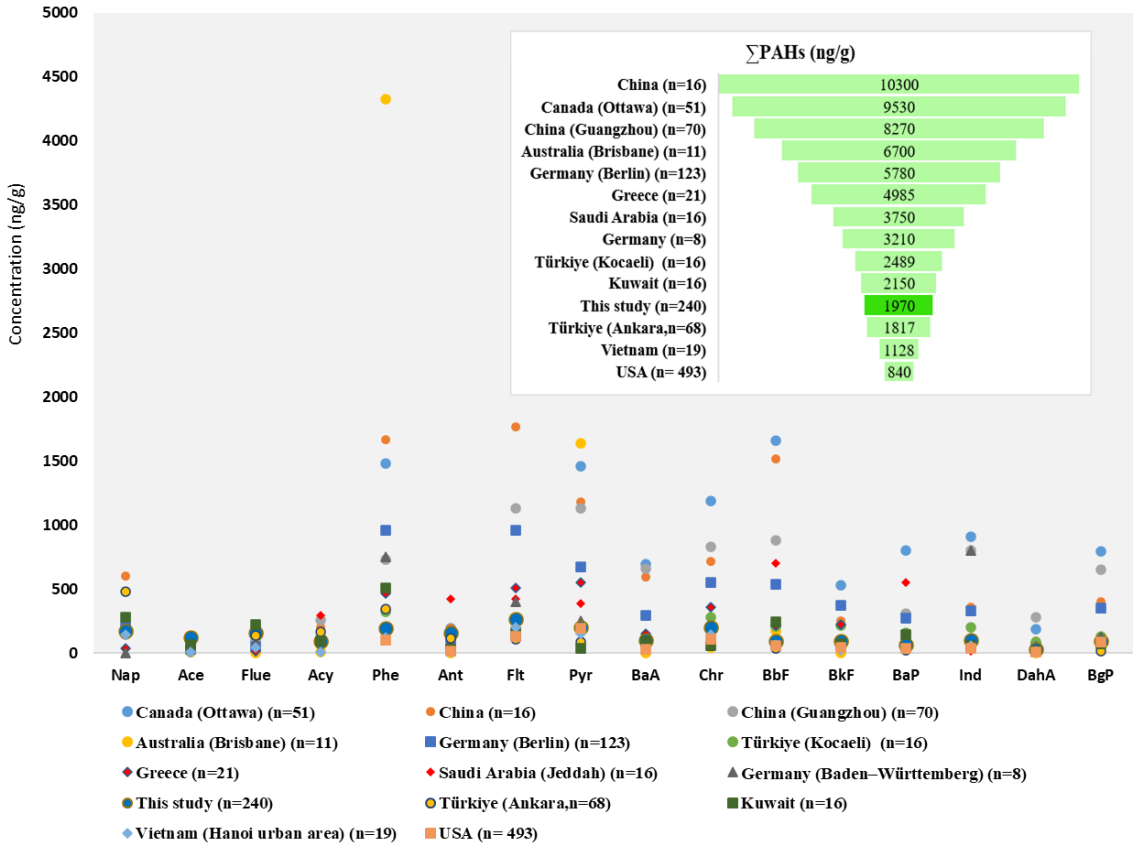


Figure 4-3. Comparison of PAH concentrations obtained in this study with data in the literature (ng/g)

In the winter of 2010, samples of indoor dust were taken from 45 private homes and 36 public buildings around China and 16 polycyclic aromatic hydrocarbons (PAHs) were identified in dust samples. The range of $\sum_{16}$PAHs values was 1000 ng/g to 47000 ng/g, with a median of 10300 ng/g. The most common type of PAHs in indoor dust are high-molecular weight (HMW) compounds (4–6 rings), which make up 84.6% of PAH concentrations in public buildings and 68% of total concentrations in private homes. The two main factors influencing PAH levels were cooking techniques and traffic conditions, particularly for sources of emissions from vehicles and coal combustion [186].

Maertens and colleagues conducted a study in 2008 to measure the concentrations of 13 polycyclic aromatic hydrocarbon (PAH) isomers in indoor dust samples collected from 51 houses in Ottawa, Canada. The $\sum_{13}$PAHs concentrations ranged from 1500 ng/g to 32500 ng/g. Among the PAH isomers, Benzo[b]fluoranthene had the highest median concentration at 1660 ng/g, while Acenaphthene had the lowest average concentration at 5 ng/g. Notably, the house with the highest total PAH concentration of 32500 ng/g was

18 years old and had 90% of its floor covered with carpet. This finding suggests that carpets can significantly trap indoor dust, potentially becoming a substantial source of PAHs indoors [188].

Polycyclic aromatic hydrocarbon (PAH) concentrations in the air and household dust were detected inside 123 homes as part of environmental monitoring in Berlin. In both the winter and the spring/summer, indoor air samples were taken in the residences of smokers and non-smokers. In the apartments of smokers, benzo(a)pyrene (BaP) median values were 650 ng/g (winter) and 270 ng/g (spring/summer), while in the apartments of non-smokers, they were 250 ng/g (winter) and 90 ng/g (spring/summer). Their findings imply that vehicle emissions may be the primary cause of the indoor PAH content in non-smoking apartments [190].

Nadeem and Iqbal determined the levels of PAH isomers in dust collected from 15 houses in Jadde in 2016. Total PAH level was measured in the range of 950 - 11950 ng/g. While the most dominant PAH isomer was Benzo[b]fluoranthene with a median value of 575 ng/g, the isomer with the lowest median value was determined to be Anthracene (50 ng/g) [191].

As a result of the measurement of PAH isomers in the study conducted on indoor dust samples collected from 90 houses in Kocaeli province, Phe was determined to be the most dominant isomer with a median value of 198.74 ng/g. This is followed by Flt with a median value of 126.85 ng/g, Pyr with a median value of 109.51 ng/g, Chr with a median value of 92.67 ng/g, and BbF with a median value of 83.98 ng/g. The most common isomers found in homes were the five-ring isomers, which constitute 46% of the total PAH isomers. The concentration of high molecular weight four, five and six ring PAH isomers in indoor dust was determined as 74% [192].

Twenty-one homes (H1–H21) had indoor dust samples taken during the colder months of 2017 (October to December). Median concentrations of PAHs contaminant was 4985 ng/g. Statistically significant correlations were also observed for PAH concentrations in dust, primarily with the number of smokers and secondarily with the presence of a fireplace [193].

Kurada and Güllü, Total PAH values measured in dust collected from living rooms and baby rooms of 68 houses in Ankara province ranged between 273 - 38259 ng/g and 137 - 39387 ng/g, respectively. While the concentration range of low molecular weight PAHs

is 147 - 36829 ng/g in the collected dust, the concentration range of high molecular weight PAHs is 126 - 18315 ng/g. According to these results, low molecular weight PAHs have higher values than high molecular weight PAHs. The highest concentrations were seen in Naphthalene in the vacuum cleaner and baby room samples, and the maximum PAH concentrations were calculated as 27944 ng/g and 14225 ng/g, respectively [195].

The other study offers first data on the levels of PAH and Me-PAH contamination and the distribution profiles of these pollutants in settled dusts from residential areas and workplaces of ELV production sites in northern Vietnam. Total PAH concentrations in dusts were found to be significantly higher in the ELV processing areas than in the corresponding living areas. This suggests that some ELV-related practices, such as burning vehicle waste in the open and improperly treating used engine oil, can release significant amounts of PAHs and Me-PAHs into the surrounding environment. In the meantime, PAHs from traffic emissions and the burning of biomass and coal, respectively, were the main sources of contamination in house dust in both urban and rural locations [196].

4.3.1 Distribute the Sources

Correlation analysis, Diagnostic ratios, Kruskal-Wallis test, and Positive matrix factorization (PMF) model were applied for the source apportionment analysis in this study.

4.3.1.1 Correlation analysis results between PAHs isomers

Correlation analysis was performed to examine the relationship between PAH isomers measured in 240 house dusts. According to the Kolmogorov-Smirnov and Shapiro-Wilk tests, the data are not normally distributed, so nonparametric correlation analysis (Spearman rank correlation) was applied. The results obtained are given in Figure 4-4.

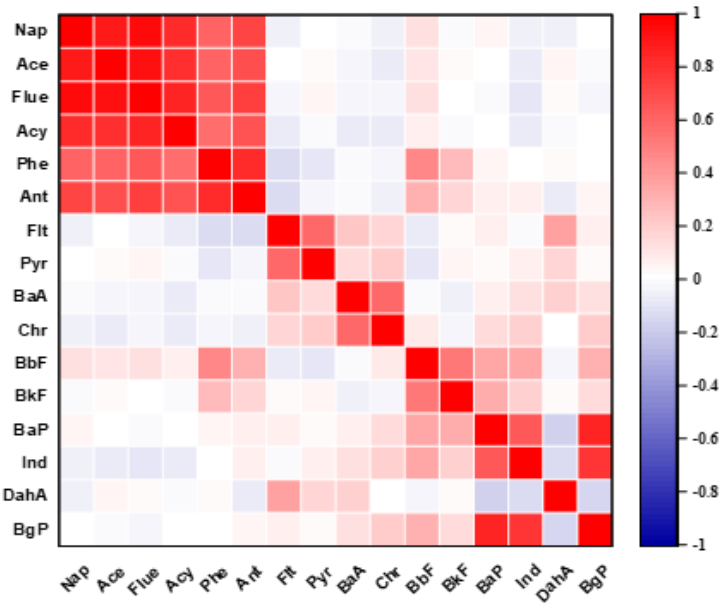


Figure 4-4. Spearman rank correlation for PAH isomers

The results presented in Figure 4-4 indicate a strong correlation among PAH isomers with low molecular weight (Nap, Ace, Flue, Acy, Phe, and Ant), suggesting that they originate from similar sources. Additionally, high correlations were observed among PAH isomers with high molecular weight. Specifically, correlations were noted between Ind and BgP, which are associated with vehicle exhaust emissions, and between BaA, BbF, BkF, and BaP isomers, which are linked to natural gas and cooking. Furthermore, correlations were found among BaA, Chr, BaP, DahA, and BkF isomers, indicating biomass combustion and smoking as potential sources. Detailed analysis of the possible sources of PAH isomers is discussed in the source identification analysis section.

4.3.1.2 Diagnostic ratio

Distribution indices, in conjunction with concentration ratios of specific PAHs, were employed for source apportionment diagnosis, following methods akin to those described in other investigations referenced in Table 4-3. Petrogenic sources consist of uncombusted petroleum products and are primarily composed of low molecular weight PAHs (2–3 rings), whereas high molecular weight PAHs (4–6 rings) are pyrogenic products resulting from the combustion of coal, fossil fuels, natural gas, diesel, gasoline, and the burning of tobacco products [211,212]. As a result, the LMW/HMW ratio can be utilized to identify anthropogenic sources of PAHs. A high concentration of LMW PAHs,

for instance, suggests a petroleum-based source of pollution, whereas a high concentration of HMW PAHs suggests a source of high-temperature burning [213].

In this study, PAH diagnostic rates, which are one of the analyzes used to determine potential PAH emission sources that may affect PAH accumulation in house dust, were used (Table 4-3).

Table 4-3. Source-identifying PAH ratios

PAH rates	Value range	Identification of emission source
BaP/BgP [214]	>1.25 <1.25	Biomass/coal combustion Traffic sources
Flt/(Flt+Pyr) [215]	< 0.4 0.4–0.5 > 0.5	Petrogenic, Petroleum emissions Burning of the fossil record, Natural gas burning mixed combustion
BaA/(BaA+Chr) [216]	< 0.2 0.2–0.35 > 0.35	Petrogenic Fossil fuels, vehicle emissions Biomass and coal burning
Ind/(Ind+BgP) [216]	< 0.2 0.2–0.5 > 0.5	Petrogenic Fossil fuels, vehicle emissions Biomass and coal burning
Flu/(Flu+Pyr) [217]	<0.5 >0.5	Gasoline emission Diesel emission

Diagnosis rate graphs are presented in Figure 4-5. When examining the BaA/(BaA+Chr) and Flu/(Flu+Pyr) ratios, as shown in Figure 4-5a, most of the BaA/(BaA+Chr) values measured in 240 dust samples are in the range of 0.2-0.35 or greater than 0.35. The results obtained reveal that the possible sources of PAH isomers are vehicle emissions and biomass combustion (burning wood and coal in cigarettes, barbecues, fireplaces or stoves). 54% of the Flu/(Flu+Pyr) values were calculated to be less than 0.5, and 46% were calculated to be greater than 0.5. These results indicate that a significant proportion of the pollutants in both groups, derived from vehicle emissions, are from diesel emissions [216,217].

The Ind/(Ind+BgP) and BaP/BgP ratios plotted in Figure 4-5b. When the values measured as Ind/(Ind+BgP) ratio are examined, 58% of the values are in the range of 0.2 – 0.5, while values greater than 0.5 account for 38%. According to the measured values for the BaP/BgP ratio, 73% of the houses remained below 1.25, indicating a significant influence of vehicle emissions and biomass combustion on the PAH levels in indoor dust [214,216].

As seen in Figure 4-5c, in the calculation of Flt/(Flt+Pyr) and Ind/(Ind+BgP) ratios, most of the values for the Flt/(Flt+Pyr) ratio are between 0.4 and 0.5 or greater than 0.5. This distribution suggests a strong presence of pyrogenic sources, such as combustion processes, contributing to the PAH levels in indoor dust. Similarly, for the Ind/(Ind+BgP) ratio, most of the values were calculated to be in the range of 0.2 – 0.5, with a significant proportion of values also exceeding 0.5. According to these results, the possible sources of these PAH isomers stand out as mixed combustion products such as vehicle emissions, the combustion of natural gas and fossil fuels used for heating and cooking, biomass combustion, and oil vapors used for frying [215,216].

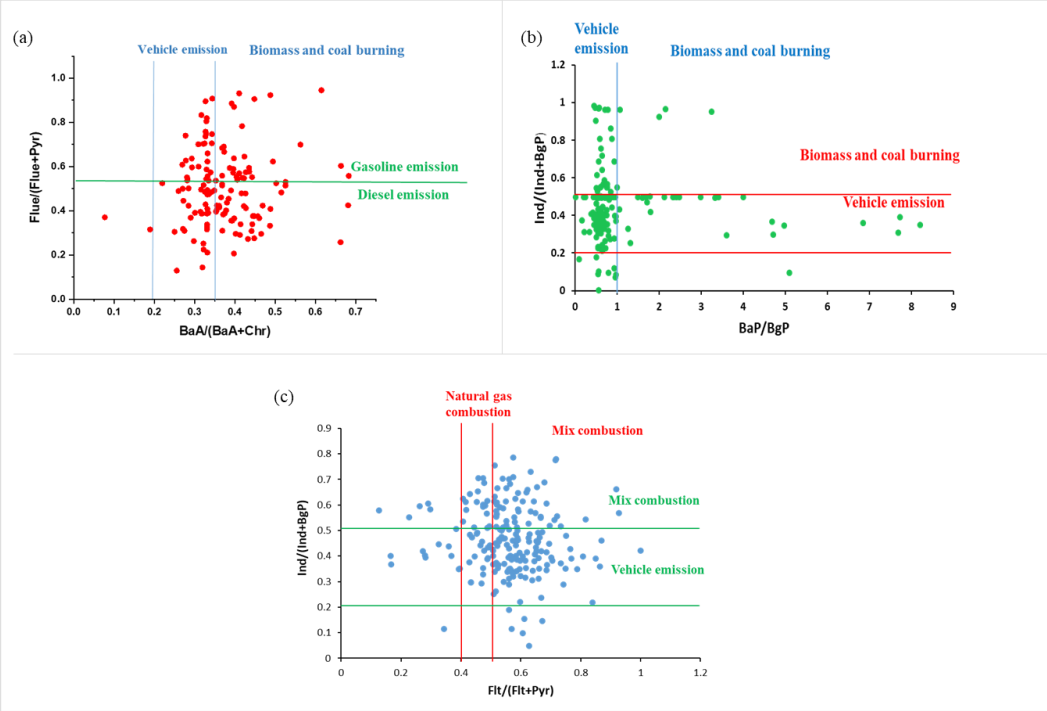


Figure 4-5. Source-determining PAH rates

4.3.1.3 Evaluation of the relationships between home conditions and PAH analysis results (Kruskal-Wallis tests)

A survey was conducted during the sampling period through personal interviews with families residing in the households of participating children. The survey encompassed inquiries about various aspects of the houses (e.g., floor type, age of the house, proximity to the main street) and the living habits of family members (e.g., smoking indoors, frequency of frying, frequency of house cleaning). Detailed explanations of the survey questions are provided in the survey evaluation section. The sources influencing the

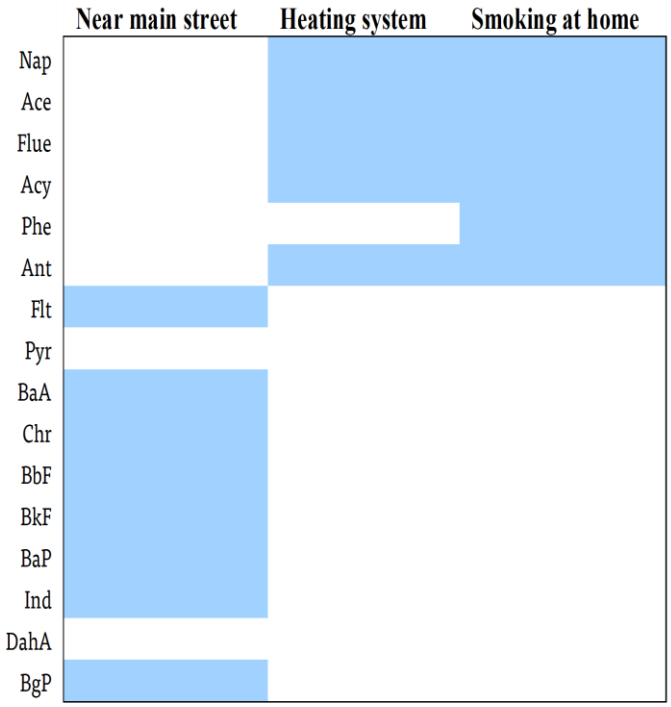
concentrations of PAH isomers measured in dust collected from homes were associated with the survey responses. To elucidate this relationship, a nonparametric Kruskal-Wallis test was employed. Based on the results of the Kolmogorov-Smirnov and Shapiro-Wilk tests, which indicated non-normal distribution of the data, nonparametric analyses (Kruskal-Wallis tests) were conducted in the statistical analysis. The Kruskal-Wallis H test, a nonparametric test based on ranks, is also known as the 'one-way ANOVA on ranks.' It is employed to evaluate whether there are statistically significant differences among two or more groups of an independent variable concerning a continuous or ordinal dependent variable. This test serves as an extension of the Mann-Whitney U test, enabling comparisons among multiple independent groups. Additionally, it serves as a nonparametric alternative to the one-way ANOVA.

The results are summarized in

Table 4-4. All survey responses were analyzed, and only associations between PAH isomers and home characteristics that were statistically significant are presented in

Table 4-4. According to the findings, 4-5-6 ring PAHs were linked with traffic emissions, likely due to proximity to the main street. 2 and 3-ring PAHs showed associations with different factors, including the heating system, and smoking at home.

Table 4-4. Correlation of PAH isomers with home conditions



The proximity of houses to the main street is categorized into two groups: <100 m and >100 m. According to the Kruskal-Wallis analysis results, proximity to the main street significantly affects the concentrations of Flt, BaA, Chr, BbF, BkF, BaP, Ind, and BgP ($P < 0.01$). Distributions revealing the relationship between proximity to the main street and PAH isomers are shown in Figure 4-6a. High molecular weight PAH isomers have been associated with vehicle exhaust emissions [218,219].

Home heating type is defined as floor heater, central system and stove (wood-coal). According to the results obtained, low molecular weight PAHs (Nap, Ace, Flue, Acy, and Ant) were measured higher in houses where stoves were used for heating. In other studies, PAH isomers with low molecular weight (anthracene, acetanafthene, phenanthrene, fluoranthene, chrysene, pyrene) have been identified as indicators of coal and wood combustion [220,221]. Figure 4-6b shows the graphical distribution of PAH isomers resulting from heating system.

A significant difference was observed in the PAH levels measured in dust collected from houses categorized as non-smoking and smoking ($P < 0.01$). The results of the Kruskal-Wallis analysis indicate that low molecular weight PAH isomers were higher in the dust of houses where smoking was conducted indoors. Low molecular weight PAHs have been detected in cigarette smoke in different studies [222,223]. Graphical relationships of smoking and values of PAH isomers are given in Figure 4-6c.

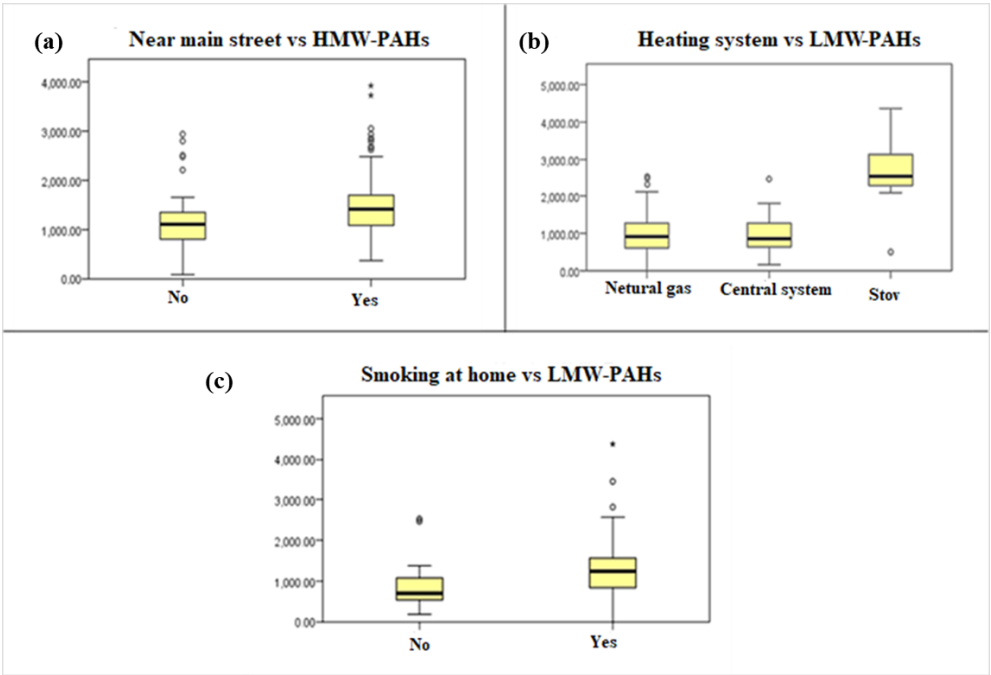


Figure 4-6. Graphical distribution of PAH isomers resulting from home characteristics

While the Kruskal-Wallis analysis may not fully explain all sources of PAH isomers in the house, the PMF model was utilized to enhance source identification. The findings from the Kruskal-Wallis analysis will aid in interpreting the results of the PMF model.

4.3.1.4 Positive matrix factorization (PMF) model

The PMF model was used to quantitatively identify possible sources of PAHs in the examined indoor dust samples to better elucidate the sources of PAHs. The "robust" mode was used to compute the elimination of each extreme value's influence. All 16 PAH congeners were regarded as "strong" in this sense. A suitable degree of uncertainty in the modeling input was indicated by the Q value of 92.7, which the four factors modeling provided. Thus, the PMF model was able to identify the proper four factors. A higher correlation was found (R2 = 0.998), indicating that the PMF model did a good job of allocating PAHs. Figure 4-7 displayed four factor profiles of PAHs. In the Factor Fingerprints screen, a stacked bar chart showed the concentration (in percent) of each species contributing to each factor. This figure is used to calculate the distribution of the factors for each species and confirm factor names.

The first factor (F1) was predominantly loaded by Flt, Pyr, BbF, BkF, and DahA, which are related to traffic sources, accounting for 19% of the total PAHs. BkF, BbF, and DahA are common tracers of gasoline and diesel combustion [224]. Also, natural gas combustion for cooking and heating is associated with the presence of Flt and Pyr among PAH compounds [47,48]. Consequently, a combined source of vehicle emissions and natural gas burning can be classified as factor 1.

The second factor (F2) was dominated by Nap, Ace, Flue, Acy, Phe, and Ant contributing with 48% of the total PAHs. F2 is acknowledged as a petrogenic source that is typically abundant in 2- and 3-ringed PAHs that are discharged into the environment because of incomplete combustion, crude and fuel oil spills, and other incidents [225]. Most gaseous 2–3 rings PAHs released during cooking oil use can enter the air directly, in contrast to their presence in dust [226–228]. Thus, cooking oil fumes as well as petrogenic sources affect factor 2.

The third factor (F3) contained 12% of the total PAHs and was characterized by BaA and Chr, which are typical indicators of traffic sources. Chr is indicative of diesel combustion [229].

The fourth factor (F4) showed the dominance of BaP, Ind, and BgP and made 21% of total PAHs. High-ring PAHs were previously found to be primarily released outdoors, such as from gasoline combustion engines [230]. BgP and Ind was the marker of gasoline vehicle emissions [231,232]. Conversely, BaP is a common genotoxin found in cigarette smoke [233]. As a result, emissions from vehicle emissions and cigarette smoke revealed a mixed source for factor 4.

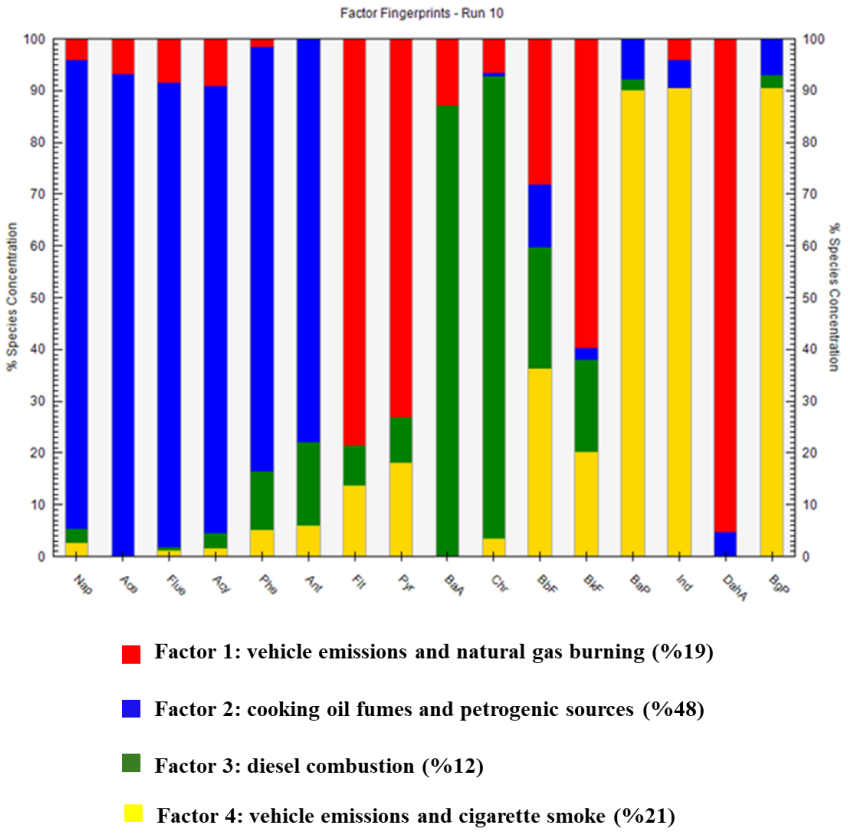


Figure 4-7. Determination of PAH source in house dust using PMF model (Profile of factors)

4.4 PCBs levels in indoor dust

PCB concentrations measured in a total of 240 indoor dust samples collected from homes are given in Table 4-2 and Figure 4-8. Based on the results, the highest measured PCB in indoor dust was PCB101, with a value of 8.5 ng/g. PCB138 and PCB153 were detected as PCB isomers with high concentrations, measuring 8.1 and 5.8 ng/g, respectively. The least common isomer found in homes was PCB180, with a median value of 0.4 ng/g.

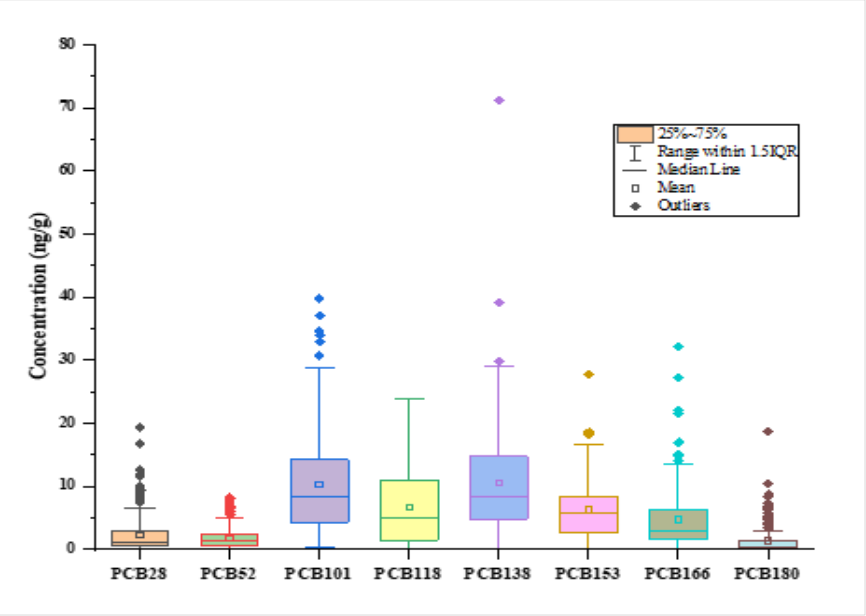


Figure 4-8. Levels of PCBs in house dust

PCBs measured in dust samples taken from the houses in this study are classified according to their homologous groups in Figure 4-9. According to the results, the most dominant PCB homolog group is 6-chlorinated biphenyls, comprising 51.18% of the total. This is followed by 5-chlorinated biphenyls at 31.99%, 3 and 4-chlorinated biphenyls at 4.09%, and 7-chlorinated biphenyls at 1.06%. Additionally, the measured PCBs were found to be like Arochlor 1254 and 1260, which are commercial production PCBs. Similar results were obtained in different previous studies [234–236].

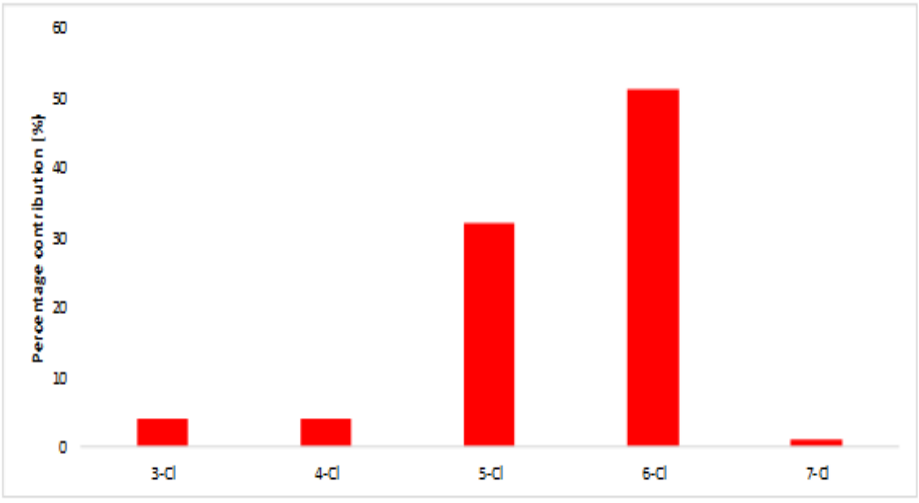


Figure 4-9. Contributions of homologous groups to total PCB concentrations in indoor dust

The median value of $\sum_8$ PCBs measured in dust samples collected from a total of 240 houses was 68.4 ng/g (11.7 – 264.7 ng/g). Figure 4-10 displays the median results obtained in this study alongside PCB measurement studies conducted on indoor dust samples from various parts of the world.

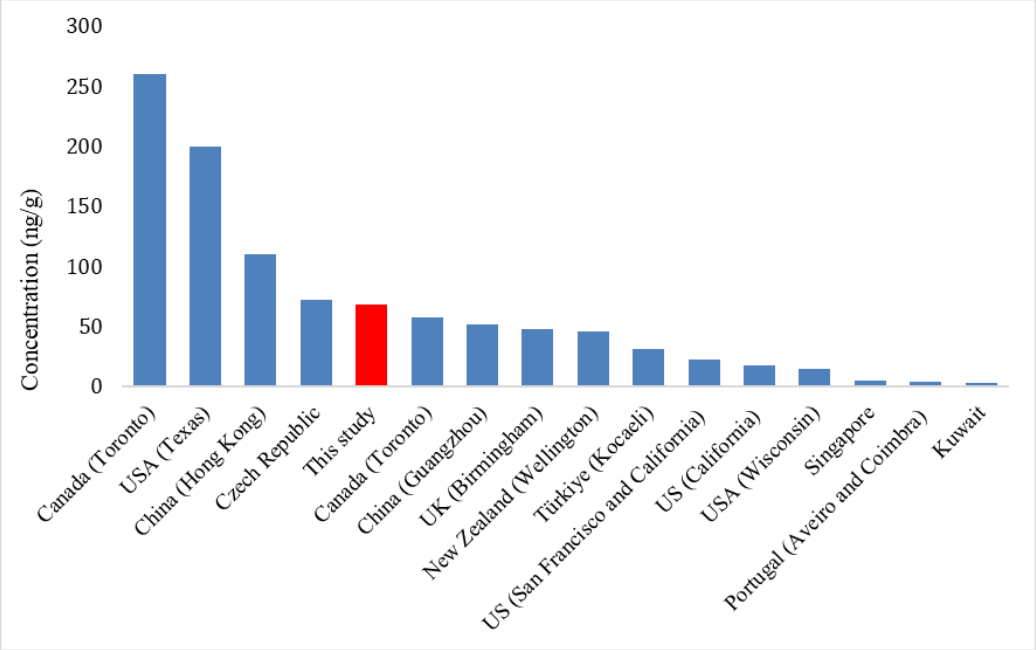


Figure 4-10. Comparison of total ∑ PCBs levels with other countries

While the results of studies conducted in countries such as Canada (Toronto) [198], America (Texas) [198], Czech Republic [200], and Hong Kong [199] are higher than the values measured in this study, they are lower than the values reported in studies conducted in countries such as Canada (Toronto) [237], China (Guangzhou)[201] Birmingham [198], New Zealand [198], USA (Wisconsin) [238], Türkiye (Kocaeli) [192], US (San Francisco and California) [239], US (California) [198], Singapore [240], Portugal [241], and Kuwait [202].

The average PCB concentration in dust samples collected from 20 homes in Texas/USA is 200 ng/g, the average PCB concentration in dust collected from 20 homes in Birmingham/UK is 48 ng/g, and the average PCB concentration in dust collected from 10 homes in Toronto/Canada is 260 ng/g. g, and the average PCB concentration in dust collected from 20 homes in Wellington, New Zealand was measured as 46 ng/g. According to the analysis results, PCB-28+31, PCB-52, PCB-101, PCB-118, PCB-138,

PCB-153 and PCB-180 were statistically similar in Canada and America, while they were found in New Zealand and the UK. It is different. No statistical similarity was found between New Zealand and UK dust concentrations. They could not explain the reasons for the low concentrations in the homes in the USA in the study. However, they explained the differences in concentrations as variations in the age of the buildings, differences in the regions where PCBs are used, decreases in PCB concentrations due to the replacement of PCB-containing building materials with those that do not contain PCBs over time, differences in sampling methods and differences in the number of samples [198].

Wang and colleagues conducted a study in Guangzhou and Hong Kong, where they collected indoor dust samples from a total of 40 houses and examined the PCB concentrations. The study found that the concentration of PCBs in indoor dust in Guangzhou city ranged from 51.8 ng/g to 264 ng/g, which was higher than the concentration measured in Hong Kong, ranging from 17.4 ng/g to 137 ng/g. The dominant PCB isomers in indoor dust were identified as PCB-18, 28, 77, 101, 126, 138, 153, 157, and 183, constituting 66% of the total concentration. Analysis of the PCB profile revealed variations in the main sources of the pollutant [199].

In the study conducted by Civan et al. in Kocaeli, PCB levels were determined in dust collected from the indoor environment in a total of 96 houses. While the median value of a total of 15 PCB isomers was 31.27, the highest measured PCB isomer was PCB153 with a median value of 4.95 ng/g, and the lowest measured PCB isomer was PCB 180 with a median value of 0.09 ng/g [192].

Tan et al. detected pesticide and PCB isomers in indoor dust samples collected from 31 houses in Singapore in 2007. In the study, the total PCB level was found to be 5.6 ng/g. PCB-101 was identified as the most abundant isomer. This isomer was followed by PCB-153. Most sample profiles were found to be penta- and hexa-PCB and match commercial Arochlor 1254. Arochlor 1254 is used in transformers and capacitors. As a result, it has been commented that more studies should be done to reduce exposure for children and people who are more sensitive to environmental pollutants [240].

In 2013, Ali and colleagues detected PCB levels in dust samples collected from a total of 30 houses in Kuwait and Pakistan. The average PCB concentration in samples collected from homes in Kuwait was found to be 3.6 ng/g, while in Pakistan, it was measured as

2.7 ng/g. Interestingly, the house with the highest PCB levels was determined to have been built in 1984 [202].

4.4.1 Distribute the Sources

In this study, correlation analysis, Kruskal-Wallis test and Positive matrix factorization (PMF) model were applied for source distribution analysis of PCB isomers.

4.4.1.1 Correlation analysis results between PCB isomers

Examining the correlations between various PCB (polychlorinated biphenyl) forms in terms of their concentrations or other pertinent characteristics is known as correlation analysis between PCB isomers. PCB isomers are PCB molecules with different biphenyl backbone chlorine replacements arranged in different structural configurations.

In this study, correlation analysis was performed to understand patterns in PCB contamination, identify potential sources or transformation pathways, or evaluate the effectiveness of remediation efforts. This analysis can offer insightful information on how PCB mixtures behave in the environment and can direct management or regulatory approaches to lessen their effects.

According to Kolmogorov-Smirnov and Shapiro-Wilk tests, the data are normally distributed, so nonparametric correlation analysis (Spearman rank correlation) was applied. The results obtained are given in Figure 4-11. Spearman rank correlation of PCBs isomers

The correlation between PCB isomers generally indicates that they have similar sources in the indoor environment.

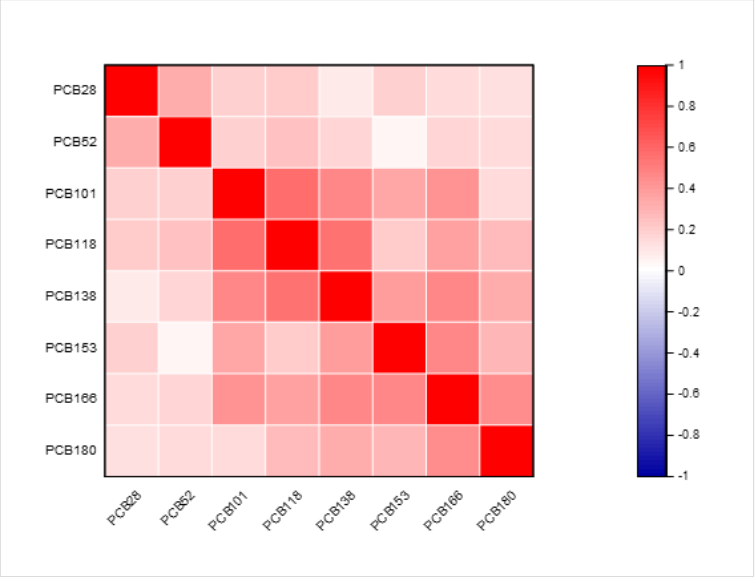


Figure 4-11. Spearman rank correlation of PCBs isomers

4.4.1.2 Evaluation of the relationships between home conditions and PCB analysis results (Kruskal-Wallis tests)

A survey was conducted during the sampling period, wherein families were interviewed one-on-one about the conditions of the houses where the children in this study lived, including the floor of the house, age of the house, proximity to the main street, etc. Additionally, information about the living habits of the family members, such as smoking in the house, frequency of frying, frequency of house cleaning, etc., was gathered. The questions asked in the survey are explained in detail in the survey evaluation section. Sources affecting the concentrations of PCB isomers measured in dust collected from homes were associated with the answers provided in the survey. To reveal this relationship, a nonparametric Kruskal-Wallis test was applied.

The proximity of the houses to the main street was categorized into two groups as < 100m and >100m. Distributions revealing the relationship between proximity to the main street and PCB isomers are shown in Figure 4-12.

According to the results, the levels of 5, 6, and 7 chlorinated PCBs exhibited statistically significant differences between houses located close to the main street and those not situated on the main street ($P < 0.05$). As seen in Figure 4-12, higher values were measured in houses close to the main street. These results suggest that vehicle emissions may be one of the potential sources of PCBs in areas with heavy traffic [199,242].

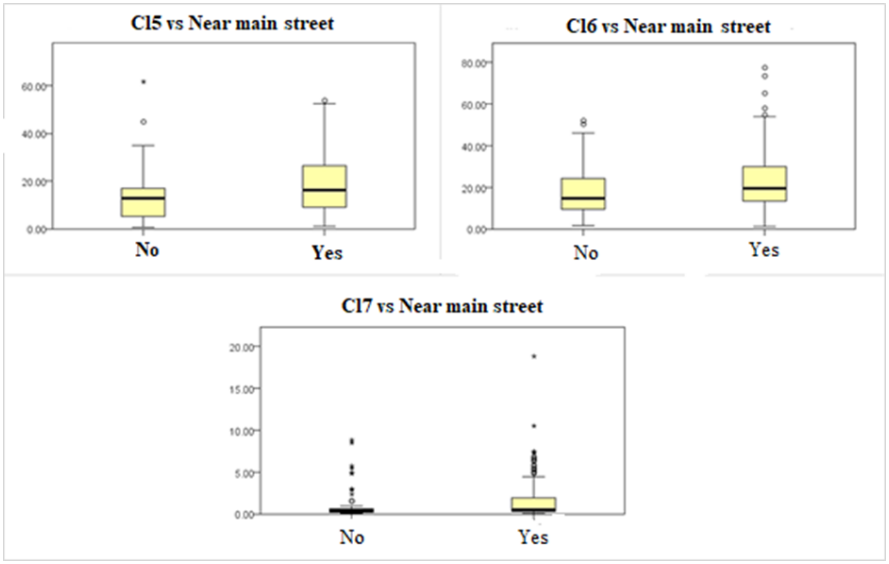


Figure 4-12. Effect of proximity to the main street on PCB values

The relationship between the cleaning frequency of the houses where dust samples were collected, and the measured PCB101, PCB138 and PCB153 values was examined with the Kruskal-Wallis Test. Distributions revealing the relationship between house cleaning frequency and PCB isomers are shown in Figure 4-13. The results reveal that the frequency of cleaning of houses affects the measured PCB values ($P < 0.05$). The houses with the highest PCB values were determined to be those that were cleaned once a week or less frequently. Less frequent cleaning of houses has led to the accumulation of pollution.

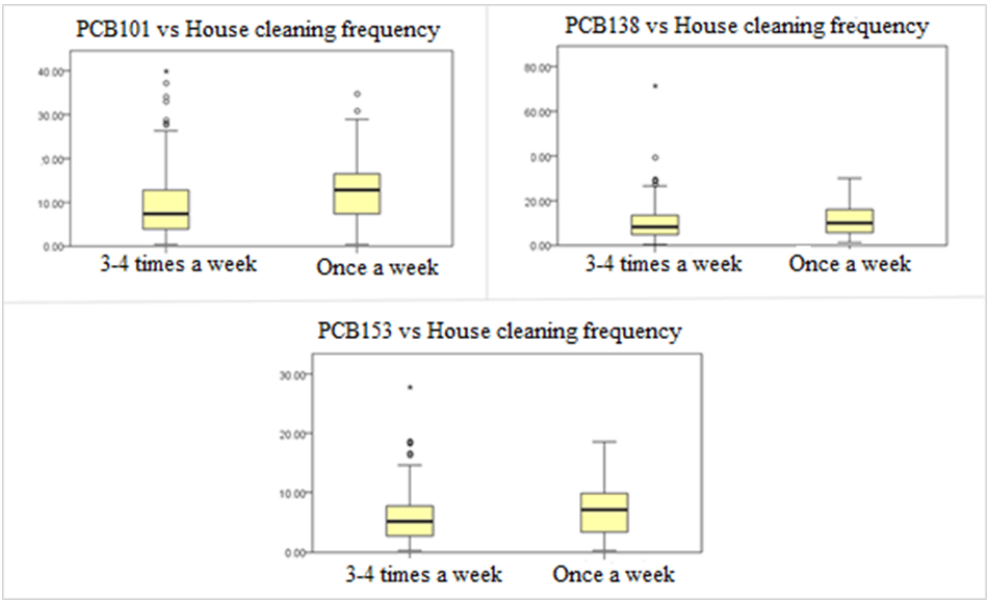


Figure 4-13. Effect of house cleaning frequency on PCB levels measured in house dust.

In the houses included in this study, PCB isomers released into the environment from new items purchased within the last year significantly affect the PCB values measured in the collected dust ($P < 0.01$). Figure 4-14 illustrates the distributions that reveal the relationship between house cleaning frequency and PCB isomers.

Penta and hexa PCBs, which are particularly found in textile products and exhibit a profile like commercial Aroclor 1254, may have been introduced into the indoor environment from new items purchased within the last year [240,243].

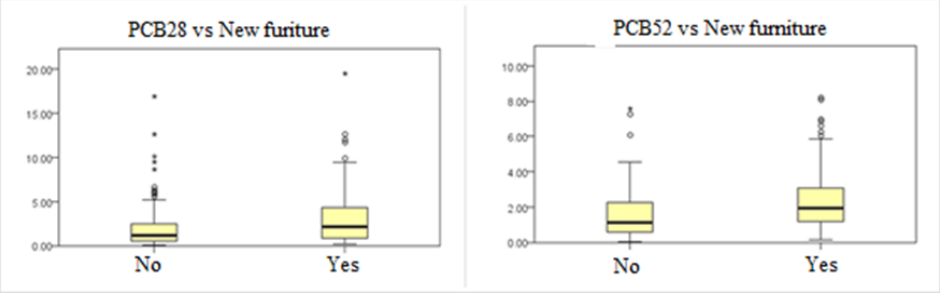


Figure 4-14. Effect of new furniture on PCB levels measured in house dust.

Repairs and painting in last year conducted in houses during the last sampling period have been found to influence the levels of PCBs measured in the collected dust. As indicated in Figure 4-15, PCB 28, PCB 52, and PCB 118 values were observed to be higher in houses where repairs and painting were conducted compared to houses without such activities. It has been demonstrated that PCBs can spread into nearby materials like wood or concrete as well as indoor air. Paint, caulk, floor sealants, and ballasts in lighting fixtures are the main sources of PCBs. Joint sealants are becoming more widely acknowledged as significant diffuse sources of PCB air pollution indoors [47,244].

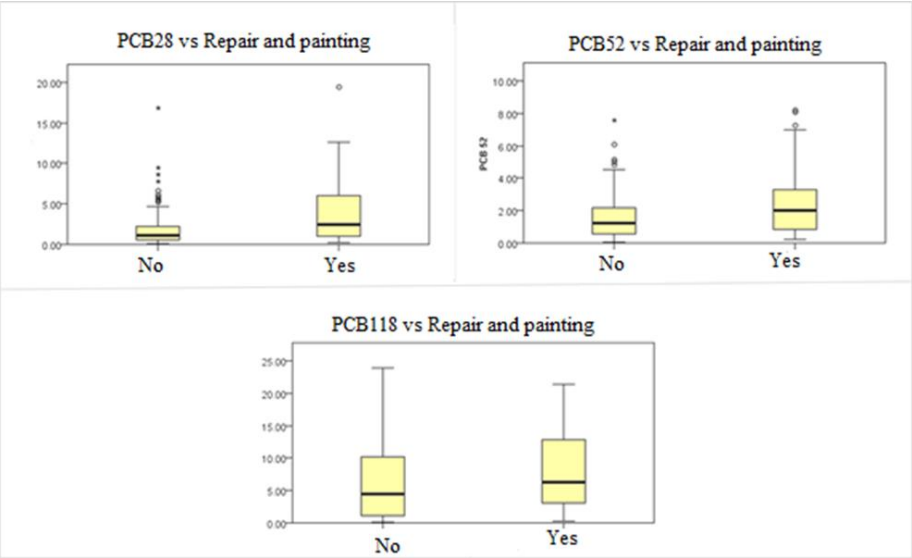


Figure 4-15. Effect of repairs or painting in the last year on PCB levels

According to the results of the Kruskal-Wallis Test, the concentrations of PCB isomers in the collected and analyzed dust samples were influenced by several factors, including proximity to the main street, frequency of house cleaning, recent purchases of new items (including electronic devices), and home repairs and painting conducted within the last year.

4.4.1.3 Positive Matrix Factorization (PMF) model:

An extensive study of PCB sources in indoor dust is possible using PMF. It facilitates the identification of sources and their contributions as well as the comprehension of how behaviors and features of the home affect PCB contamination levels. To lower indoor PCB exposure, targeted mitigation techniques can be informed by this integrated strategy. In this study, the PMF model was used to quantitatively identify possible PCB sources in the examined indoor dust samples to further elucidate the PCB origins. Results of the PMF analysis indicated two identified factors that are characteristic for PCBs found in indoor dust (

Figure 4-16. Profile of factors for deternination source of PCBs isomersThe first factor (F1) was predominantly loaded by Cl5 and Cl6 PCBs, accounting for 37.6% of the total PCBs. Although PCB isomers have been associated with gasoline exhaust emissions in previous studies, they are also identified as emissions from coal and wood combustion [242,245,246]. For this reason, this factor can be defined as vehicle exhaust and

combustion emissions for heating purposes. The second factor consists of low molecular weight 3,4 and 5 chlorinated PCBs compared to the other factor and explains 62.4 % of the total variance. While low molecular weight PCBs are associated with gasoline exhaust emissions, PCB-28, 44, 52, 101, 118 are shown as emissions from construction and insulation materials.

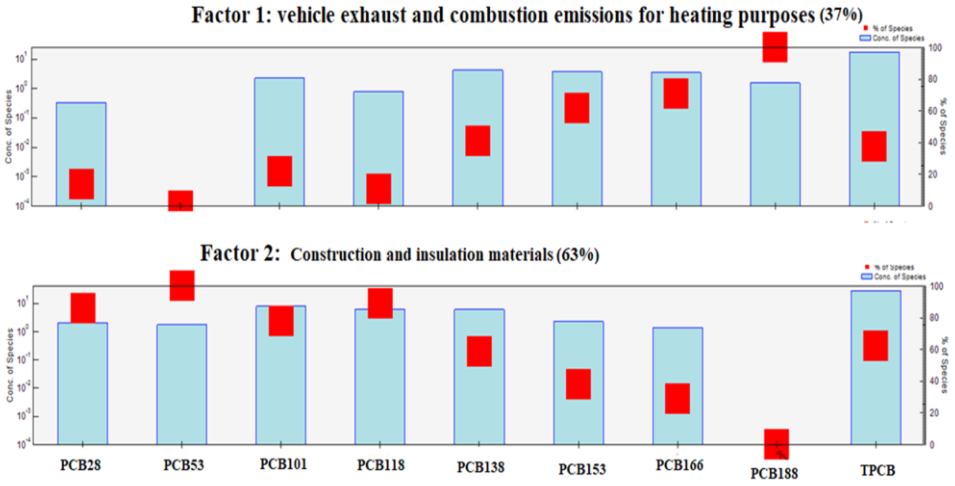


Figure 4-16. Profile of factors for deternination source of PCBs isomers

4.5 The levels of PBDEs in indoor dust

PBDEs concentrations measured in a total of 240 indoor dust samples collected from homes are given in Table 4-2 and Figure 4-17.

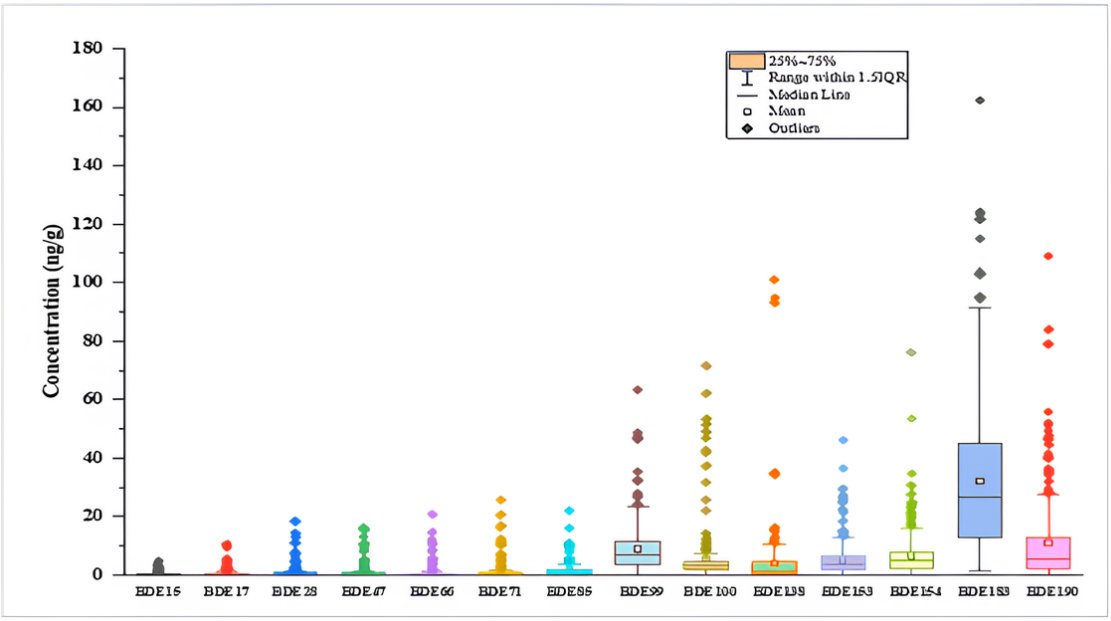


Figure 4-17. Leveles of PBDEs in house dust

As a result, BDE-209 was measured as the highest pollutant with a median value of 26.6 ng/g. The demand for the use of deca-BDE has increased due to the ban of penta and octa PBDEs in the European Union and other countries [247]. The PBDE isomer with the second highest concentration was identified as BDE-183 (hepta-BDE), which is the main component of commercial octa-BDE containing products, with a median value of 24.5 ng/g. The other pollutant with the highest concentration detected in the study was PBDE-99, with a median value of 6.7 ng/g. This is followed by PBDE-190, which has a median value of 5.4 ng/g, and PBDE-154, with a median value of 4.8 ng/g.

The major congener detected in two Deca-formulations, Saytex 102E and Bromkal 82-0DE, was BDE-209, comprising 96.8% and 91.6%, respectively. BDE-209 was also the major congener (49.6%) in the 79-8DE (Octa-BDE) formulation, while BDE-183 (42%) dominated in the other Octa-BDE formulation, DE-79. Finally, the Penta-formulations DE-71 and Bromkal 70-5DE were found to contain six major congeners: BDE-99, BDE-47, BDE-100, BDE-153, BDE-154, and BDE-85. The most prominent of these were BDE-47 and BDE-99, each comprising 38-49% of the total congener composition [248]. In this way, the order of penta-BDE, octa-BDE, and deca-BDE isomers detected in the indoor environment are utilized under the trade names DE-71 and Bromkal 70-5DE for penta-BDE formulations, DE-79 for octa-BDE, and Saytex 102E and Bromkal 82-0DE for deca-BDE and were detected in studies conducted in indoor and outdoor environments.

The median results obtained in this study with PBDE measurement studies carried out in indoor dust samples from various parts of the world are shown in Figure 4-18.

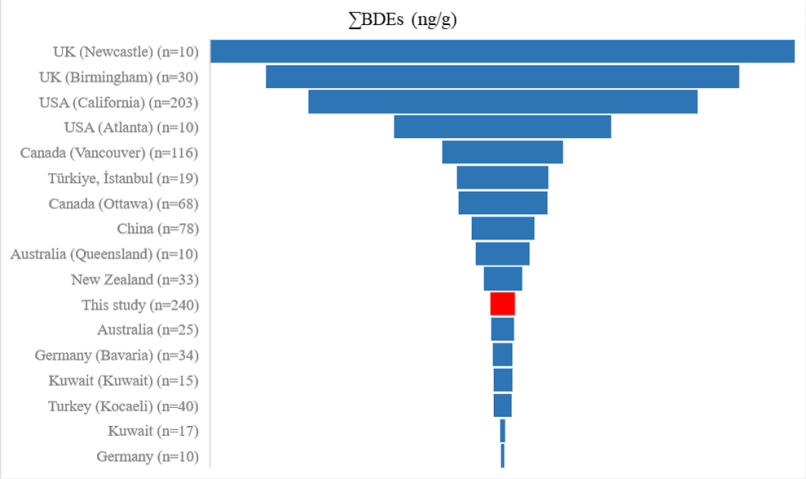


Figure 4-18. Comparison of total PBDE levels with other countries

The median value of total PBDEs measured in 240 dust samples in this study was 431.6 ng/g (ranging from 16.5 to 2544 ng/g). This value is lower than those reported in several other countries, including the UK (10067 ng/g and 8145 ng/g), USA (6707 ng/g and 3750 ng/g), Canada (2083 ng/g and 1541 ng/g), Türkiye (Istanbul) (1580 ng/g), China (1096 ng/g), Australia (944 ng/g), and New Zealand (672 ng/g). However, it is higher than the values measured in Germany (345 ng/g and 73 ng/g), Kuwait (337 ng/g and 93 ng/g), and Türkiye (Kocaeli) (316 ng/g).

These comparisons highlight significant geographical variations in PBDE concentrations in indoor dust, which could be attributed to differences in the usage of PBDE-containing products, regulatory policies, and environmental factors across different regions.

In the study conducted by Civan and Kara in 40 houses in Kocaeli, Türkiye, the total PBDE concentration was determined to be 316.1 ng/g [249]. Another study conducted by the same research group in 90 houses in the same city measured PBDE isomer concentrations, revealing that levels ranged from 1 (BDE-209) to 15 times (BDE-71) lower than those found in the earlier study [192]. In both studies, no significant relationship was identified between the survey data (such as building age, location, and lifestyle) collected from the houses where samples were taken. This discrepancy in PBDE levels between the two studies might be attributed to differences in sampling periods or methodologies used. The other study, conducted in February and March (winter and spring seasons), showed that indoor ventilation was effective in diluting other PBDE isomers except BDE-209 due to seasonal differences. The initial study was carried out in December and January (winter season). This indicates that seasonal variations in indoor ventilation can significantly affect the concentrations of PBDE isomers, demonstrating the importance of considering seasonal factors when assessing indoor air quality and pollutant levels. In a study conducted in a different city in Türkiye (Istanbul), the median concentrations of all BDE isomers except BDE-85 and BDE-154 were measured at higher values [250]. Differences in dust particle diameters, sampling seasons, economic levels, lifestyles, and outdoor pollution levels may account for this variation, despite both studies being conducted within the same country. These factors highlight the complexity of pollutant distribution and the need for comprehensive assessments when comparing data across different regions.

Sjödin and olleagues carried out their studies in 4 countries in 2008: Germany, Australia (Queensland), the United States (Atlanta) and UK (Newcastle). In this study, indoor dust samples were collected from 10 houses per country to determine PBDE concentrations.

A total of 7 PBDE concentrations in indoor dust in Germany range from 17 ng/g to 550 ng/g, a total of 7 PBDE concentrations in indoor dust in Australia range from 500 ng/g to 13000 ng/g, and in indoor dust in America PBDE concentrations have been estimated to range from 520 ng/g to 29000 ng/g and the total PBDE concentration in Great Britain to range from 950 ng/g to 54000 ng/g. The isomer with the highest concentration in all countries was found to be BDE 209. The median value of BDE 209 concentration is 63 ng/g (83% to ΣPBDEs) in Germany, 730 ng/g (61% to ΣPBDEs) in Australia, 10000 ng/g (61% to ΣPBDEs) in Great Britain 100%) and 2000 ng/g in America (48% to ΣPBDEs). The fact that BDE 209 was detected mostly in Great Britain indicates that it may be used more as a fire retardant here compared to other countries [251].

4.5.1 Source apportionment of PBDEs

4.5.1.1 Correlation analysis results between PBDEs isomers

Correlation analysis was performed to examine the relationship between PBDE isomers measured in dust samples collected from the houses in this study. According to the Kolmogorov-Smirnov and Shapiro-Wilk tests, the data are not normally distributed, so nonparametric correlation analysis (Spearman rank correlation) was applied. The results obtained are given in Figure 4-19. The results indicate a significant correlation between PBDE isomers. Strong correlations between penta BDEs suggest that these isomers likely share common sources. Furthermore, the correlation between the commercial PBDE 209 isomer and other isomers highlights televisions, computers, plastic toys, and textile products as the primary sources of PBDEs in indoor environments [252].

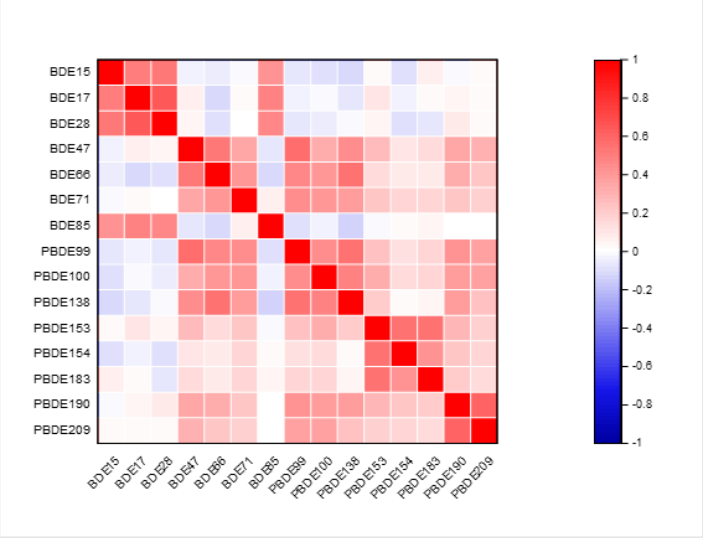


Figure 4-19. Spearman rank correlation of PBDEs isomers

4.5.1.2 Evaluation of the relationships between home conditions and PCB analysis results (Kruskal-Wallis tests)

A survey was conducted during the sampling period by personally interviewing the families about the conditions of the houses where the children participating in this study lived (e.g., floor of the house, age of the house, proximity to the main street) and the living habits of the family members (e.g., smoking in the house, frequency of frying, frequency of house cleaning). The questions asked in the survey are explained in detail in the survey evaluation section. Sources affecting the concentrations of PBDE isomers measured in dust collected from homes were associated with the answers given in the survey. To reveal this relationship, a nonparametric Kruskal-Wallis test was applied.

The age of a house can significantly influence the levels and types of PBDEs present. In this study, the age of the houses was grouped into two categories: under 20 years and over 21 years. Higher levels of PBDE isomers were found in older homes (Figure 4-20). This may be related to several factors. Due to the flame retardants employed in their manufacture, older electrical devices—which are more likely to be found in older homes—are important sources of PBDEs. Because older homes may have less effective insulation, contaminants like PBDEs in indoor dust may accumulate and persist longer. Additionally, the concentration of PBDEs might vary depending on a home's construction materials, ventilation system, and maintenance routine [170,253].

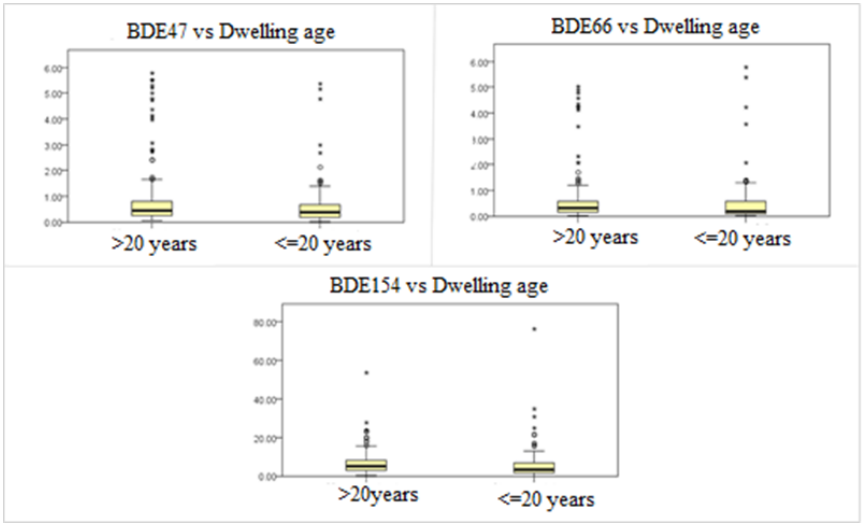


Figure 4-20. Effect of house age on PBDE levels

In this study, PBDE isomers released into the environment from new items purchased in the last year affected the PBDE values measured in the collected dust ($P<0.05$). The results of the Kruskal-Wallis Test are given in Figure 4-21. This suggests that recent additions to the household, such as new electronics, furniture, or textiles, which often contain PBDEs as flame retardants, contribute to the overall levels of these chemicals found in indoor dust. The continuous introduction of new items with PBDEs can lead to an increase in the accumulation of these harmful chemicals within the home environment [251]. The relationship between new items purchased in the last year and Okta-BDEs revealed that the substance used as a flame retardant under the trade name DE-79, which has a similar profile to Okta-BDEs, was used more in the new items found in the sampled houses [248].

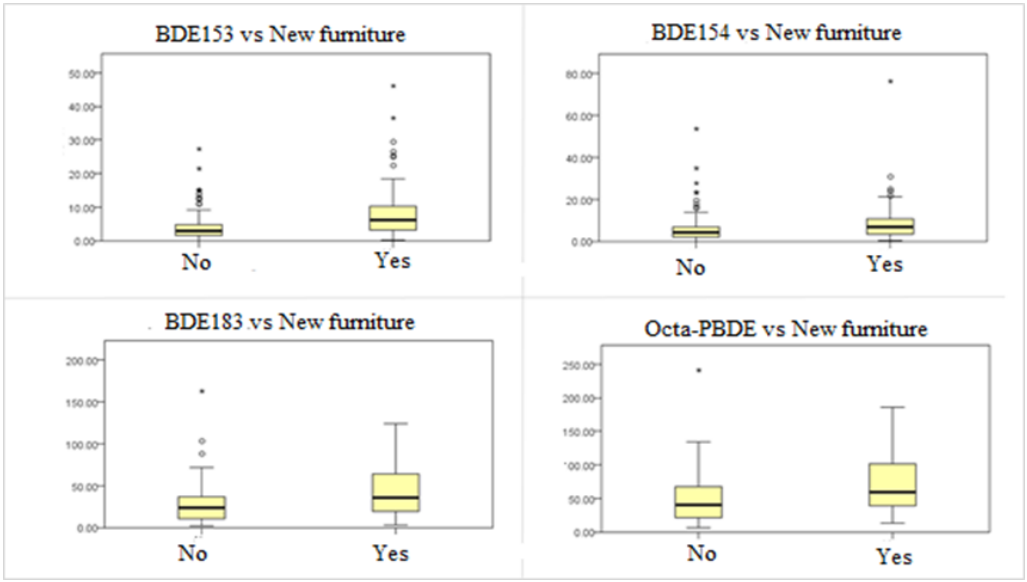


Figure 4-21. Effect of new furniture purchased in the last year on PBDE values

In this study, the type of bed the child sleeps on was evaluated in two groups: wood/plastic and metal. Kruskal-Wallis test results reveal that there is a statistical difference in measured PBDE levels between the two groups ($P<0.05$). As can be seen in Figure 4-22, especially BDE 209 was measured higher in the dust samples collected from the homes of children whose beds were made of wood/plastic. According to these results, it has been estimated that the use of PBDE isomers as fire retardant in products made of wood or plastic materials, such as the child's bed, is one of the sources of PBDE in houses made of this material [248,253].

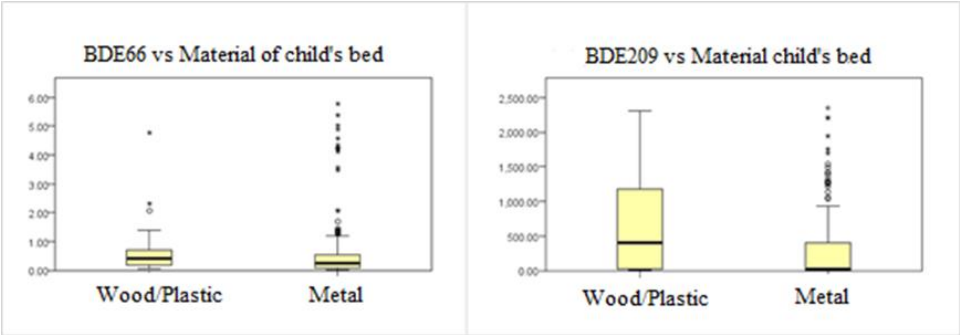


Figure 4-22. Effect of child's bed material on PBDE values

4.5.1.3 Positive Matrix Factorization (PMF) model

The positive matrix factorization (PMF) approach is used to assign sources of PBDE homologues and ∑8PBDEs in home dusts, as shown in Figure 4-23. The most likely results are cited, and the change in Q value is used to draw the conclusion that there were two main sources of PBDEs in home dusts.

Factor 1 is loaded by tri-to-penta-BDEs including BDE15, BDE17, BDE28, BDE71, BDE85, and BDE99 as the main components of the commercial mixtures of penta-BDE. More than 85% of these 6 PBDE homologues are contributed by factor 1 (Figure 4-23). Penta-BDE is mostly added to polyurethane foams that are used to make textiles and furniture [254]. The predominance of penta-BDE isomers DE-71 and Bromcal 70-5DE indicates the use of commercial PBDEs [170]. Therefore, the application of penta-BDE in domestic microenvironments may be responsible for factor 1. Factor 2 is dominated by BDE100, BDE138, BDE153, BDE154, BDE154, BDE183, BDE190 and BDE209, which are the main components of the commercial mixtures of octa-BDE and deca-BDE (Figure 4-23). In polymer polymers used to make electrical and electronic equipment, the two mixes are added [255]. As a result, factor 2 is associated with home appliance use. The contribution frequencies of factors 1 and 2 to ∑PBDEs are 6.9% and 93.1%, respectively, suggesting that the primary source of ∑PBDEs in household dust is the use of electrical and electronic equipment.

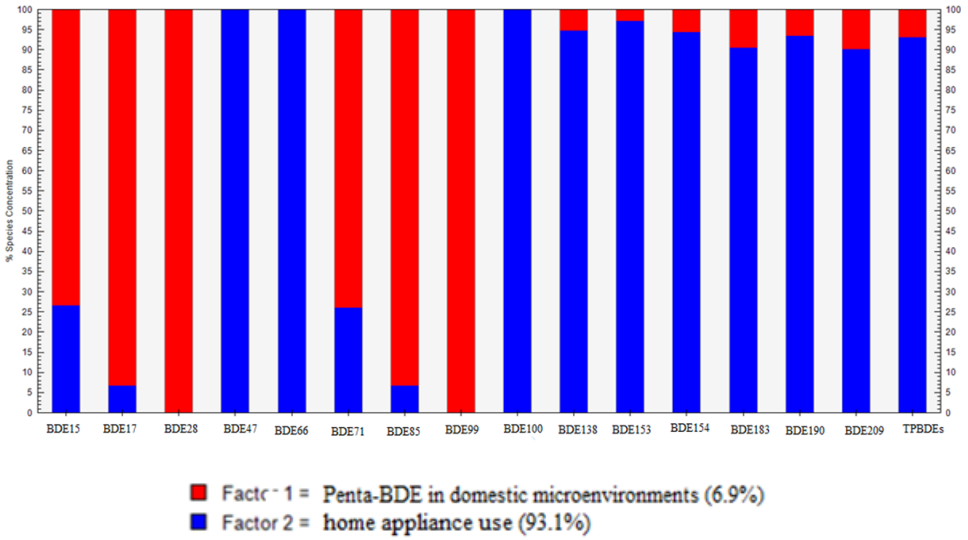


Figure 4-23. PMF factor profiles of household dust of PBDEs.

4.6 Alkylphenols, and alkylphenol ethoxylate

As stated earlier, OP or 4-OPME was not present in the analyzed samples. The concentrations of 4-nonylphenol (4-n-NP), 4-nonylphenol diethoxylate (di-NPE), and 4-tert octylphenol (4-t-OP) in dust from 148 houses are presented in Table 4-2. The range of 4-n-NP concentration in residential dust was 49–8740 ng/g , with a median of 520 ng/g. The highest concentrations were observed for di-NPE in indoor dust, with a median value of 1910 ng/g (ranging from 113 to 9070 ng/g). This finding can be attributed to the that the production of nonylphenol ethoxylates accounts for around 80 % of the total volume of alkylphenol ethoxylates [62]. Among the APs, 4-t-OP showed lower concentrations in all the settled dust (35 ng/g, ranging from 8 to 227 ng/g). There have been relatively few studies on the presence of alkylphenols (APs) and alkylphenol ethoxylates (APEs) in dust. Table 4-5. displays the concentration of APs and APEs reported in previous studies. In comparison to several studies documented in the literature, the concentrations of APs and APEs in indoor dust were found to be lower in our study. For instance, Kubwabo et al. (2016) reported mean concentrations of 8970 ng/g for 4-n-NP and 3960 ng/g for di-NPE in residential dust from Canada, which were higher then the mean concentrations (854 ng/g and 2520 ng/g, respectively) observed in our study [73]. Wilson et al. (2003) found higher 4-n-NP concentrations in dust samples collected from a nursery in North Carolina, USA, compared to the concentrations in our study [72]. In another study, Rudel et al. (2003) reported median concentration for NP,

di-NPE, and OP in house dust as 2580 ng/g, 5330 ng/g, and 130 ng/g, respectively, which were higher than the values measured (520 ng/g, 1910 ng/g and 35 ng/g, respectively) in our study. However, it's worth noting that our study yielded higher values when compared to some previous research [65]. The mean values of di-NPE (1860 ng/g), NP (341 ng/g) and OP (26 ng/g) in dust collected from home in South Africa is lower than the amount measured in this study [74]. Similarly, Lu et al. (2013) reported lower median concentrations of OP and NP in house dust from China [64] (Table 4-5.).

Table 4-5. APs and APE concentrations (in ng/g) in indoor dust were compared to those found in other investigations in the current study.

Location	Isomers	Min.	Max.	Mean	Median	Reference
Ankara, Türkiye	4-n-NP di-NPE 4-t-OP	49 113 8	8740 9070 465	854 2520 59	520 1910 35	This study
Canada	NP di-NPE	1000 nd	84200 28100	8970 3960	6840 28400	[73]
Cape Cod, MA, USA	NP di-NPE OP	nd nd nd	8680 49300 1990		2580 5330 130	[65]
Dueban, South Africa	NP di-NPE OP	127 85 15	686 4810 47	341 1860 26		[74]
North Carolina	NP	3280	9620	7220		[72]
China	NP OP	nd nd	9 20		3 5	[64]
Japan	NP	nd	42300		3100	[71]

4.6.1 Influencing factors of Alkylphenols and alkylphenol ethoxylates in household dust

The Shapiro-Wilk U test was used to determine whether the data were normally distributed. As the data were not normally distributed, the Mann-Whitney U test, a nonparametric equivalent of the t-test, was employed to compare median values of data sets containing two groups/conditions at a 95% confidence level. The p-value of <0.05 indicates significant differences between the median of the groups being compared.

Target chemicals included in consumer products and building materials, such as electronics, furniture, floor and wall coverings, and personal care and cleaning supplies, can migrate, leach, abrade, or off-gas from these items, resulting in human exposure [256–258]. Therefore, house characteristics and the lifestyle of the people living in the homes may affect the levels of pollutants measured in this study. In order to reveal this effect, the results of the face-to-face survey conducted with the families who voluntarily participated in the study were used.

Factors that may affect 4-n-NP, di-NPE and 4-t-OP concentrations are evaluated in Figure 4-24.

Of the families included in this study, 94.6% live in apartments, while 5.4% live in detached houses. These proportionsremain similar when we categorize the houses into those larger and smaller than 100 m^2. The study found that whether the houses were apartments or villas and whether they were large or small did not have an impact on the levels of target pollutants. However, the number of people living in a household appears to influence the concentration of 4-n-NP, particularly when the household exceeds four people ($p < 0.05$). Houses with more occupants exhibited a higher median value in di-NPE and 4-t-OP. Previous studies have indicated that these chemicals are prevalent in personal care products and household cleaning detergents [259–261]. Consequently, a larger number of residents might lead to increased usage of these products, thereby resulting in higher levels of these pollutants being measured.

Among other things, alkylphenols and alkylphenol exhalate are utilized as surfactants in paint and varnish hardeners and adhesives [262–264]. Furthermore, APE is still applied as emulsifying agents in latex paints [259,265,266]. A report prepared by Karakas shows that construction paints and varnishes constitute 58% of the total paint used in Türkiye. It is known that 60% of this amount contains NPE [267].

Among the sampled houses, those with polished wood flooring constitute 13.5% of the total. It was observed that dust samples taken from these houses had higher levels of di-NPE and 4-t-OP.

di-NPE levels were measured higher in dust collected from houses that used latex paint as wall paint ($p < 0.05$), and there is no statistically significant difference for 4-n-NP and 4-t-OP levels.

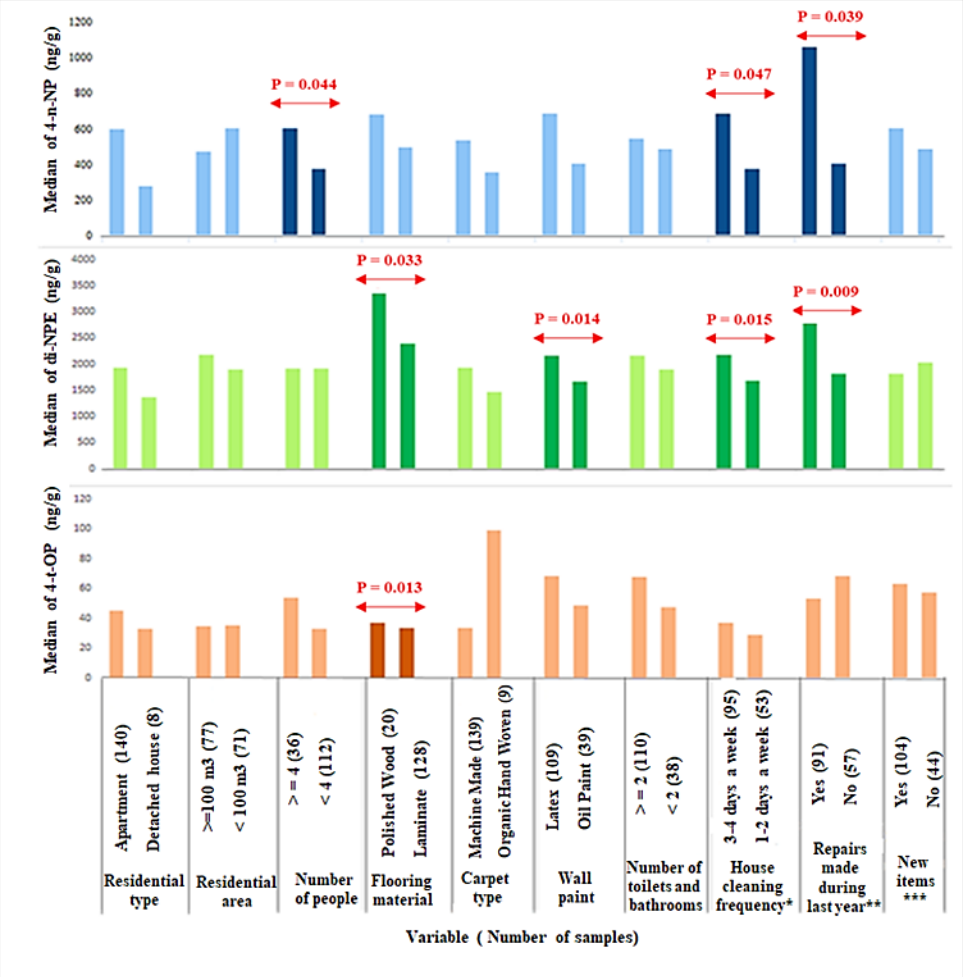
A total of 93.91% of the families preferred to use machine-made carpets due to their modernity and affordability, while only nine families chose hand-woven carpets. It was found that carpet type does not significantly affect pollutant levels.

Non-ionic surfactants, such as alkylphenol and alkylphenol ethoxylates, are commonly found in detergents and bleach. It was hypothesized that homes with more toilets and bathrooms might use more bleach disinfectants, potentially leading to higher concentrations of these chemicals. However, the study found no significant correlation between the number of toilets and bathrooms and the levels of these pollutants. This could be due to the common practice of rinsing toilets and bathrooms with water after disinfection, which may prevent some chemicals from integrating with house dust.

Families may engage in home cleaning routines for 1-2 days or 3-4 days a week, depending on their lifestyle and working hours. The cleaning products they use often contain significant amounts of alkylphenol and alkylphenol exhalates. Detergents or spray cleaning agents used for floor cleaning can accumulate in dust, potentially having a notable impact on human health. Dust samples collected from houses where families cleaned 3-4 days a week and used a higher quantity of chemical cleaning products revealed higher levels of 4-n-NP and di-NPE.

Some studies have shown that large amounts of nonylphenol and nonylphenol ethoxylates leach from paints, varnishes, and PVC products, such as plumbing pipes, window shutters, and gutters [268,269]. For these reasons, it was hypothesized that house repairs conducted during the final sampling period, including activities like painting, changing floor parquet, replacing doors or windows, and updating installation pipes, had an impact on the levels of 4-n-NP and di-NPE.

In summary, reported concentrations of APs and APE indicate the widespread use of NP- and NPE-based consumer applications and products in the studied microenvironments, including household cleaning products, personal care products, paints, sealants, and more.



*House cleaning frequency: Bleach is used for cleaning and parquet floors are wiped with detergent.
**Repairs made during the last year: Repairs included wall painting, floor parquet replacement, and plumbing pipe replacement.
***New household items purchased within the last year: Furniture, carpets, wooden cabinets ,and electrical appliances were purchased as household items.

Figure 4-24. Effect of home conditions and the lifestyle of the people living in the homes on the levels of APs and APE

4.6.2 Source apportionment of dust 4-n-NP, di-NPE, and 4-t-OP via a logistic regression model

The association between chemical exposure, home characteristics, and family lifestyle was investigated using the multivariate logistic regression model. The parameters included in this model were residential type and area, number of people in the household, flooring material, carpet type, type of wall paint, number of toilets and bathrooms, cleaning frequency, repairs made during the last year, and new household items purchased within the last year. Multivariate logistic regressions were used to explore relationships between higher (> median) concentrations of chemical markers and various home parameters. Odds ratio (OR) and adjusted odds ratios (aOR) with a 95% confidence

interval (Cl) were used to express the multivariate results. It is essential to check the sufficiency or goodness of fit of the model before fitting it. The estimated model's predictive capability might be used to find this. Model fit and accuracy in this study were assessed using the Chi-square goodness-of-fit test, Hosmer-Lemeshow test, Classification table, and Logistic regression R^2 value.

Table 4-6 and Figure 4-25 depict the results of multivariate logistic regression analyses for 4-nonylphenol (4-n-NP), 4-nonylphenol diethoxylate (di-NPE), and 4-tert-Octylphenol (4-t-OP). Concentrations of chemical markers were divided into two categories (≤ 50 percentile (median) vs > 50 percentile), and the reference category consisted of levels below the 50th percentile.

Multivariate logistic regression analysis revealed that high levels of 4-nonylphenol (4-n-NP) were associated with the frequency of house cleaning (OR = 3.2, 95% CI: 1.13-9.03), repairs made during the last year (OR = 4.33, 95% CI: 1.8-10.45), residential type (OR = 4.11, 95% CI: 1.32-12.78), and the number of people (OR = 3.01, 95% CI: 1.07-8.49). According to the odds ratio values, more frequent use of cleaning materials for cleaning is associated with approximately a 3-fold increase, repairs made in the last year are associated with approximately a 4-fold increase, having an apartment-type residence is associated with approximately a 4-fold increase, and having more than 4 people living in the house is associated with approximately a 3-fold increase in the likelihood of high levels of 4-n-NP exposure.

High levels of 4-nonylphenol diethoxylate (di-NPE) have been found to be associated with flooring materials (OR = 6.88, 95% CI: 1.9-24.89), the frequency of house cleaning (OR = 2.51, 95% CI: 1.1-5.71), repairs made during the last year (OR = 2.21, 95% CI: 1..04-4.71), and the purchase of new household items within the last year (OR = 2.52, 95% CI: 1.09-5.8). Specifically, polished wood flooring, cleaning 3-4 days a week with cleaning materials containing detergent, making repairs and painting within the last year, and purchasing new household items have all been associated with approximately 7, 2.5, 2, and 2.5 times higher concentrations of di-NPE, respectively.

Considering the results presented in the previous section, the high concentrations measured for 4-t-OP were not found to be associated with any of the household parameters, as predicted.

Table 4-6. Results of the binary logistic regression model to determine the relationship between APs and APE and building conditions and lifestyle behaviors.

		4-n-NP	di-NPE	4-t-OP
	Options (Number of sample)	**OR, 95 % CI (> median vs ≤ median; reference: ≤ median)**		
Residential type	Detached house (ref) Apartment	**4.11 (1.32-12.78)***	2.1 (0.42-10.35)	2.91 (0.05-19.66)
Residential area	< 100 m^3 (ref) >=100 m^3	1.36 (0.57-3.28)	1.64 (0.75-3.54)	1.24 (0.42-3.64)
Number of people	< 4 (ref) >= 4	**3.01 (1.07-8.49)***	1.19 (0.42-2.99)	2.6 (0.73-9.17)
Flooring material	Laminate (ref) Polished Wood	2.34 (0.63-8.61)	**6.88 (1.9-24.89)***	1.78 (0.46-7.99)
Carpet type	Hand Woven (ref) Machine Made	3.85(0.66-22.51)	4.08 (0.87-18.92)	3.45 (0.32-3.3)
Type of wall paint	Oil Paint (ref) Latex Paint	1.73 (0.63-4.97)	1.41 (0.59-3.37)	1.05 (0.32-3.3)
Number of toilets and bathrooms	< 2 (ref) >=2	1.15 (0.41-3.21)	2.14 (0.88-5.18)	1.55 (0.49-4.24)
House cleaning frequency[1]	1-2 days a week (ref) 3-4 days a week	**3.2 (1.13-9.03)***	**2.51 (1.1-5.71)***	2.1 (0.8-5.5)
Repairs made during the last year[2]	No (ref) Yes	**4.33(1.8-10.45)***	**2.21 (1.04-4.71)***	1.18 (0.42-3.3)
New house items purchased within the last year[3]	No (ref) Yes	1.29 (0.49-3.14)	**2.52 (1.09-5.8)***	1.08 (0.28-3.3)

***P value** < 0.05

1. House cleaning frequency: Bleach is used for cleaning and parquet floors are wiped with detergent.
2. Repairs made during the last year: Repairs included wall painting, floor parquet replacement, and plumbing pipe replacement.
3. New household items purchased within the last year: Furniture, carpets, wooden cabinets ,and electrical appliances were purchased as household items

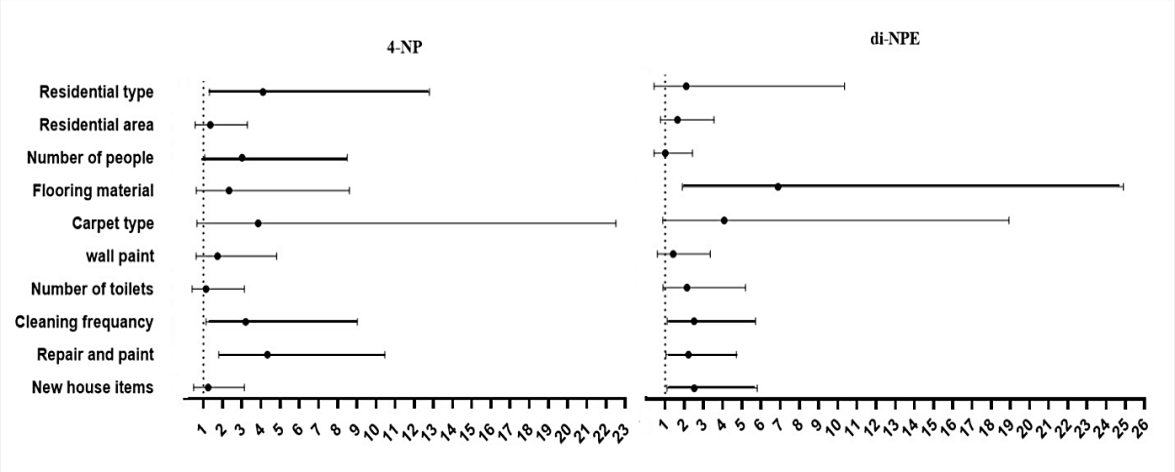


Figure 4-25. Relationships between 4- NP and di- NPE and building conditions, family-related information, and lifestyle behaviors.

4.7 Effect of target pollutants known as endocrine disrupting chemicals on childhood asthma development

The connection between the target pollutants analyzed in the dust samples collected from the homes of the children in the case and control groups included in the study and the development and severity of stroke is discussed in this section. As given in the Materials and Methods section, the children diagnosed with asthma, which constitute the case group, were selected from asthmatic children who were under follow-up at the Hacettepe University Pediatric Allergy Clinic, and the control group was selected from children in the similar age group who were declared to have no allergies or asthma. The hospital records of the children in the case group and the findings obtained from the survey form were used in the research. Routine follow-ups of children with asthma were carried out by their own doctors, and disease-related data were used in this study.

4.7.1 Characteristics of The Asthma Group

Given the strong relationship between air quality and health, it is crucial to collect data from a comprehensive medical assessment of patients.

Periodic health monitoring of the asthmatic and control group children included in the study was carried out throughout the study. The children diagnosed with asthma, which constitute the case group, were selected from children who were under follow-up at the Hacettepe University Pediatric Allergy Clinic or were newly diagnosed with asthma, and

the control group was selected from children in a similar age group who were declared to have no disease. The hospital records of the children in the case group and the findings obtained from the survey form were used in the research. The follow-up form given in Appendix C was used in the routine follow-up of children with asthma. If a health problem was observed in children selected for the control group, the child was seen by the research team, an evaluation was made regarding his disease, and he was removed from the control group. Routine check-ups of the children under follow-up were performed by their own doctors and hospital records were used in the research.

Criteria for inclusion in the study under the case (asthmatic) group.

• Presence of reversible airway obstruction in clinical and/or pulmonary function tests that would support the diagnosis of asthma (only patients with clinically reversible airway obstruction are also included),

• Being between the ages of 6-11,

• Volunteering to participate in the research.

“Pulmonary function tests” have been used to help diagnose asthma, determine disease severity, reveal pathogenetic mechanisms, reveal phenotypes, and observe and follow the effects of treatment. Spirometry and/or flow-volume loop, FEV1 and FEV1/FVC parameters are most commonly used to evaluate obstruction. If detected, showing that this obstruction is reversible brings us closer to the diagnosis of asthma. Again, in cases where obstruction is not detected, bronchial hyperreactivity tests are used. However, detection of bronchial hyperreactivity is not specific for asthma. It is also seen in chronic obstructive pulmonary disease, allergic diseases and especially allergic rhinitis. The use of standardized instruments in the application of pulmonary function tests, an accurate test using standardized methods by an experienced team dealing with this work and its good interpretation are very important for the consistency of the results. In this study, it is important for the reliability of asthma control data that all children with asthma were followed up in the same clinic by an experienced team.

In this study, patient follow-up forms of children in the asthmatic group were filled out by Hacettepe University Pediatric Allergy Department doctors who participated in the project, and the numerical summary of the medical data in these forms is given in Table 4-7.

Table 4-7. Characteristics of the asthma group

	n=110
Female, n (%)	55 (50.5)
Age, mean	8.32
Age of symptom, mean	2.98
Age of asthma diagnose, mean	4.24
Follow up duration, mean	4.82
Family history of atopic disease, n (%)	46 (44.7)
Asthma exacerbation within last month, n (%)	46 (44.7)
Asthma exacerbation within last year, n (%)	87 (84.5)
Emergency department visit within last year, n (%)	37 (35.9)
Allergic Rhinitis, n (%)	23 (22.3)
Atopy, n (%)	39 (37.9)
Atopic dermatitis, n (%)	20 (19.4)
FEV_1, mean	91.23
Wheezing, n (%)	12 (11.7)
Night cough, n (%)	46 (44.7)
Asthma severity	
Mild asthma, n (%)	64 (59.8)
Moderate/ Severe asthma, n (%)	46 (40.2)
ACT Score, (Median IQR)	23 (19-25)
ACT	
Controlled, n (%)	84 (76.14)
Uncontrolled, n (%)	26 (23.85)

In the asthma group, gender frequency was roughly equal (Girl: 50.5%, boy: 49.5%). The children in the asthma trial were eight years old on average. Of the children whose asthma was diagnosed, 44.7% experienced an asthma attack in the past month, 84.5% experienced an asthma attack in the past year, and 35.9% reported visiting the emergency department in the previous year. Atopy was present in 37.9% of children with asthma, and throughout the follow-up, 19.4% of children in the asthma group had an official diagnosis of atopic dermatitis. Of the children included in this study, 44.7% had a family history of atopic disorder. In the asthmatic child population, wheezing was recorded by just 11.7% of the children. Among those with asthma, 44.7% said they had a nighttime cough. The asthma group's mean forced expiratory volume, or FEV1, was 91.23%. Asthma Control Test (ACT) results show that over two thirds (76.14%) of children had asthma symptoms that are under control or potentially under control (ACT score >19). According to the GINA standards, the 110 children with asthma were divided into two asthma severity groups: mild asthma and moderate/severe asthma. Of the individuals with asthma, 40.2% have moderate/severe asthma, and 59.8% have mild asthma.

4.7.2 Results of a questionnaire survey

In parallel with the sample process, a questionnaire survey was carried out to assess the risk variables associated with pediatric asthma. The children's parents participated in in-person interviews and completed a specially created questionnaire.

Logistic regression models were employed to examine the impact of the lifestyles and environmental conditions of the families on the risk of asthma in children. ORs and 95 % CI were calculated, and they are summarized in Table 4-8. Variables containing the level (floor), age (year) and area (m3) of residential, number of occupants, flooring material, carpet type, wall paint, window material, separate kitchen, near main street, heating system, frying frequency, smoking at home, repairs and painting done in the last year, new furniture, pet keeping, cleaning frequency, frequency of using detergent and material of the child's mattress.

Table 4-8. Questionnaire survey (n_{Case} = 110, $n_{Control}$ =130) to identify childhood asthma risk factors

	Case and Control	
	Crude OR (95 % CI)	**Adjusted** OR (95 % CI)
Level of House		
≤1 Floor	**3.04 (1.78-5.17)**	**4.29 (2.24-8.24)**
≥2 Floor (reference)		
Dwelling age		
≤20 Years (reference)		
>20 Years	**1.83 (1.07-3.14)**	**2.42 (1.23-4.77)**
Residential area (m2)		
≤100	1.28 (0.74-2.23)	
>100 (reference)		
Number of occupants		
≤4 (reference)		
>4	**2.15 (1.13-4.01)**	**2.17 (1.02-4.65)**
Flooring material		
Laminate (reference)		
polished wood	1.55 (0.73-3.33)	
Carpet type		
Synthetic (reference)		
Wool	1.21 (0.48-3.03)	
Type of wall paint		
Plastic Paint (reference)		
Oil Paint	1.94 (0.79-4.8)	
Lime	0.75 (0.32-1.72)	
Wallpaper	**3.14 (1.15-8.57)**	**4.06 (1.23-13.37)**
Type of window material		
PVC (reference)		
Wood	1.03 (0.58-1.82)	
Separate kitchen		
Yes (reference)		
No	**5.23 (1.44-19.07)**	**4.63 (1.11-19.46)**
Near main street		
No (reference)		
Yes	1.01(0.57-1.76)	

Heating system		
Natural gas	**2.27 (1.25-4.11)**	
Central heating (reference)		
Frying frequency		
Once a week (reference)		
Twice a week or more	**2.21 (1.31-3.74)**	**2.31 (1.82-3.03)**
Smoking at home		
No (reference)		
Yes	**1.81 (1.08-3.03)**	**1.99 (1.06-3.76)**
Repairs and painting done in the last year		
No (reference)		
Yes	1.55 (0.86-2.76)	
New furniture purchased within the last year		
No (reference)		
Yes	**2.02 (1.14-3.56)**	**2.49 (1.26-4.95)**
Pet keeping		
No (reference)		
Yes	**2.82 (1.41-5.66)**	**4.5 (1.92-10.59)**
Cleaning frequency		
Once a week or less(reference)		
Twice a week or more	**1.71 (1.01-2.94)**	
Frequency of using detergent		
Once a week or less(reference)		
Twice a week or more	**1.83 (1.07-3.13)**	**2.05 (1.06-3.98)**
The material of the child's mattress		
Cotton (reference)		
Viscos elastic	**2.61 (1.22-5.52)**	
Wool	**3.18 (1.13-8.96)**	**2.88 (1.18-7.06)**

First, each variable was initially included in the model individually, and the crude model (Model 0) was used to assess the unadjusted association between the variable and asthma risk. The category with the lowest probability of being a risk factor for asthma was designated as the reference value for each variable. Asthma incidence was positively correlated with thirteen of the factors under investigation: Level of House, dwelling age, number of occupants, type of wall paint, separate kitchen, heating system, frying frequency, smoking at home, new furniture, pet keeping, cleaning frequency, frequency of using detergent and material of the child's mattress.

Model 1 adjusted for variables that were positively associated with asthma in the model 0. The adjusted model reveals the cumulative effect of the variables. Eleven of the factors investigated showed positive associations with asthma occurrence: living on the first or lower floors (OR = 4.29; 95%CI: 2.24-8.24), living in houses built more than 20 years ago (OR = 2.42; 95%CI: 1.23-4.77), more than 4 people at home (OR = 2.17; 95%CI: 1.02-4.65), using wallpaper for wall covering (OR = 4.06; 95%CI: 1.23-13.37), having no separate kitchen (OR = 4.63; 95%CI: 1.11-19.46), Frying two or more times a week (OR = 2.31; 95%CI: 1.82-3.03), parents' smoking at home (OR = 1.99; 95%CI: 1.06-3.76), new items purchased in the last year (sampling period) (OR = 2.49; 95%CI: 1.26-4.95), pet keeping (OR = 4.5; 95%CI: 1.92-10.59), using detergent for cleaning twice a

week or more (OR = 2.05; 95%CI: 1.06-3.98), using wool for child's mattress (OR = 2.88; 95%CI: 1.18-7.06).

When OR values are examined, living on the first or lower floors, using wallpaper as wall covering, not having a separate kitchen, and having pets can affect the development of asthma approximately 4 times. However, living in houses built before 20 years, having more than 4 people at home, frying two or more times a week, smoking at home, using new furniture, frequency of detergent use, and using wool for the child's mattress can affect the development of asthma approximately 2 times.

Compared with the above, several traditionally accepted risk factors, such as polished wood flooring, wool carpet, wood windows, near the main street, and repair and painting in the last year (sampling period), were not statistically associated with the occurrence of childhood asthma.

Meng et al. conducted a study revealing that the ventilation status of the kitchen and mildew on the walls were triggers for asthma disease. In this study also some risk factors such as parental smoking, home decoration, and proximity to a main street were not statistically associated with the occurrence of childhood asthma [175].

One important contributing factor was tobacco smoking (ETS). Several studies have shown a correlation between prenatal and postnatal ETS exposure and an increased prevalence of asthma [270–272].

4.7.3 Comparison of pollutant concentrations between case and control groups:

A total of 110 asthma cases and 130 matched controls were included in the final analysis. The concentrations of the included analytes were computed, and Table 4-9. Summarizes the inter-group variations in the concentrations by analysis using the Mann-Whitney U test. According to the analysis results, the distributions of target pollutants with a statistical difference between their median values in the case and control groups are presented in Figure 4-26.

The levels of polycyclic aromatic hydrocarbons (PAHs), including Acenaphthene ($Con._{case}$ = 130.5 ng/g, $Con._{control}$ = 106.8 ng/g, p value= <0.01) , Fluorene ($Con._{case}$ =168.1 ng/g, $Con._{control}$ = 142.7ng/g, p value=<0.01), Acenaphthylene ($Con._{case}$ = 104.7 ng/g, $Con._{control}$ = 87.8 ng/g, p value=<0.01), Phenanthrene($Con._{case}$ = 369.9 ng/g, $Con._{control}$ = 111.1ng/g, P value=<0.001) , Anthracene($Con._{case}$= 220.3 ng/g, $Con._{control}$= 108.8 ng/g, P value=<0.001), Benzo[b]fluoranthene ($Con._{case}$= 200.1 ng/g, $Con._{control}$ = 56.4 ng/g, P value=<0.001), Benzo[k]fluoranthene ($Con._{case}$ =115.8 ng/g, $Con._{control}$ = 70.1 ng/g, P

value=<0.001), and Indeno[1,2,3-cd]pyrene ($Con._{case}$ = 105.1ng/g, $Con._{control}$ = 100.2 ng/g, P value=<0.01), were found to be significantly higher in asthma cases compared to controls. Exposure to polycyclic aromatic hydrocarbons (PAHs) including benz(a) anthracene, benzo(a)pyrene, benzo(b)fluoranthene, benzo (g,h,i)perylene, chrysene, and dibenz(a,h)anthracene has been linked in the past to respiratory issues, asthma development, and increased cough and wheeze by the time a child is 12 months old [122,163].

While PCB118 ($Con._{case}$ = 1.49 ng/g, $Con._{control}$ = 9.46 ng/g, P value=<0.001), PCB138 ($Con._{case}$ = 7.02 ng/g, $Con._{control}$ = 10.77 ng/g, P value=<0.001), and PCB180 ($Con._{case}$ =0.39 ng/g, $Con._{control}$ = 0.5 ng/g, P value=<0.01) were measured higher in the control group, PCB153 ($Con._{case}$= 6.92 ng/g, $Con._{control}$ = 5.19 ng/g, P value=<0.001) was found to be higher in asthma cases (Table 4-9 and Figure 4-26).

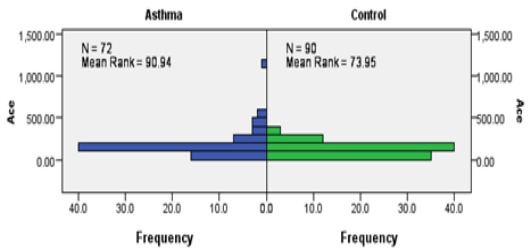
Along with a retrospective study on associations between typical POP exposure and childhood asthma in Shanghai, Meng et al collected indoor dust samples from the homes of asthmatic and non-asthmatic children (n = 60 each). Among the 27 PCBs, PCB52($Con._{case}$ = 0.05 ng/g, $Con._{control}$ = 0.08 ng/g, P value=0.072), PCB ($Con._{case}$ = 0.05 ng/g, $Con._{control}$ = 0.06 ng/g, P value=0.223), PCB118 ($Con._{case}$ = 0.05 ng/g, $Con._{control}$ = 0.04 ng/g, P value=0.562), PCB138 (Con.case = 0.08 ng/g, Con.control = 0.07 ng/g, P value=0.268), PCB153 ($Con._{case}$=0.08 ng/g, $Con._{control}$ = 0.09 ng/g, P value=0.789), PCB166 ($Con._{case}$=0.07 ng/g, $Con._{control}$ = 0.05 ng/g, P value= 0.022) were mesured [175].

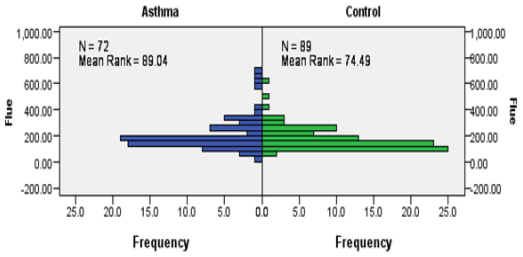
The measured concentration values of BDE-15 ($Con._{case}$ = 0.14 ng/g, $Con._{control}$ = 0.07 ng/g, p value= <0.001), BDE-17 ($Con._{case}$ = 0.39 ng/g, $Con._{control}$ = 0.26 ng/g, p value= <0.001) , BDE-28 ($Con._{case}$ = 0.38 ng/g, $Con._{control}$ = 0.34 ng/g, p value= <0.001), and BDE-85 ($Con._{case}$ = 0.86 ng/g, $Con._{control}$ = 0.5 ng/g, p value= <0.001) were higher in the asthma group compared to the control group, while the opposite trend was observed for BDE-154 ($Con._{case}$ = 3.02 ng/g, $Con._{control}$ = 5.37 ng/g, p value= <0.001) and BDE-190 ($Con._{case}$ = 28.09 ng/g, $Con._{control}$ = 23.1 ng/g, p value= <0.001) values (Table 4-9.

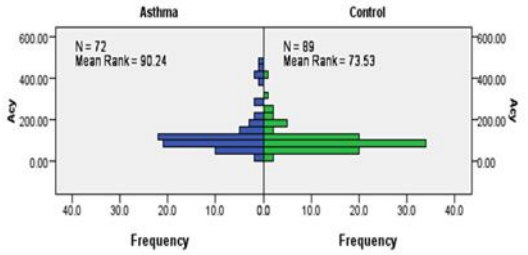
A study conducted in China by Meng et al. (2016) found higher measurements of BDE-28 ($Con._{case}$ = 0.53 ng/g, $Con._{control}$ = N.D.), BDE-85 ($Con._{case}$ = 1.57 ng/g, $Con._{control}$ = 1.02 ng/g), and BDE-153 ($Con._{case}$ = 1.02 ng/g, $Con._{control}$ = 0.15 ng/g) in the asthma group [175].

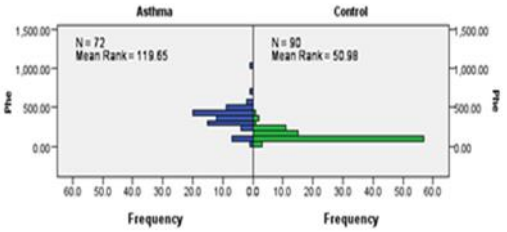
Table 4-9. Selected EDCs detection frequencies (DFs), concentrations (DFs), and intergroup variations in indoor dust

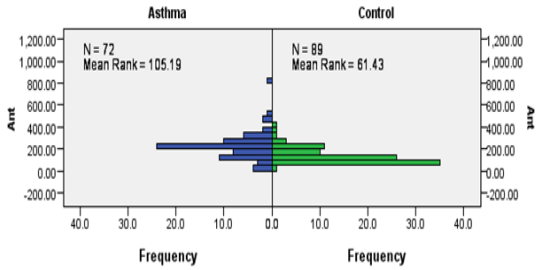
	Case Group (n=110)				Control Group (n=130)				P value
	DF	25th	Median	75th	DF	25th	Median	75th	
PAHs									
Naphthacene	66	137.01	179.32	263.88	68	120.41	155.22	253.68	0.293
acenaphthene	66	105.07	130.54	197.4	69	82.4	106.8	159.39	<0.01
fluorene	66	137.27	168.06	266.04	68	115.61	142.73	221.06	<0.01
Acenaphthylene	67	83.87	104.71	117.65	68	69.05	87.84	110.8	<0.01
phenanthrene	66	304.17	369.96	446.65	69	85.67	111.06	160.18	<0.001
anthracene	65	146.8	220.32	258.18	68	85.56	108.83	165.65	<0.001
fluoranthene	100	129.73	230.72	304.91	98	200.07	269.02	304.33	0.126
Pyrene	95	125.17	200.08	216.2	96	149.46	203.6	259.9	0.064
benz[a]anthracene	95	92.67	100.39	148.06	96	100.03	110.54	151.97	0.244
chrysene	100	149.56	202.19	238.25	100	200.22	203.8	258.07	0.123
benzo[b]fluoranthene	93	101.59	200.09	273.03	90	40.83	56.39	90.4	<0.001
benzo[*k*]fluoranthene	95	84.35	115.85	232.72	92	35.62	70.07	127.31	<0.001
benzo[a]pyrene	92	83.87	104.71	117.65	94	26.17	64.59	115.85	0.287
indeno[1,2,3-cd]pyrene	91	57.78	105.1	123.13	93	25.37	100.15	122.67	<0.01
dibenz[a,h]anthracene	96	12.33	26.74	50.36	93	15.01	28.9	44.15	0.882
benzo[g,h,i]perylene	95	53.22	105.01	161.16	92	32.11	87.24	150.1	0.303
PCBs									
PCB28	100	0.54	1.34	2.86	100	0.64	1.22	3.22	0.904
PCB52	100	0.57	1.39	2.32	100	0.6	1.3	2.55	0.659
PCB101	100	4.76	9.43	12.97	100	4.05	8.48	14.7	0.679
PCB118	100	0.64	1.49	4.42	100	4.88	9.46	14.75	<0.001
PCB138	100	4.30	7.02	10.47	100	5.72	10.77	16.26	<0.001
PCB153	100	2.81	6.92	9.92	100	2.75	5.16	7.30	<0.001
PCB166	100	1.53	3.21	6.42	100	1.71	2.93	6.38	0.765
PCB180	100	0.19	0.36	0.66	100	0.26	0.5	1.85	<0.001
PBDEs									
BDE15	100	0.07	0.14	0.34	100	0.04	0.07	0.18	<0.001
BDE17	100	0.23	0.39	0.63	100	0.12	0.26	0.53	<0.001
BDE28	100	0.21	0.38	0.81	100	0.13	0.34	0.69	<0.001
BDE47	100	0.21	0.4	0.95	100	0.23	0.41	0.72	0.981
BDE66	100	0.09	0.19	0.74	100	0.12	0.3	0.48	0.419
BDE71	100	0.16	0.4	0.95	100	0.16	0.46	0.75	0.928
BDE85	100	0.29	0.86	2.01	100	0.2	0.5	1.45	<0.001
BDE99	100	3.2	5.34	11.18	100	3.63	6.87	11.97	0.495
BDE100	100	1.91	3.11	4.31	100	1.84	2.94	4.38	0.498
BDE138	100	0.24	0.89	4.48	100	0.64	1.22	4.53	0.123
BDE153	100	1.44	2.82	6.56	100	1.93	4.04	6.12	0.188
BDE154	100	1.55	3.02	6.85	100	3.19	5.37	8.34	<0.001
BDE183	100	11.58	25.05	47.56	100	12.99	27.48	43.37	0.559
BDE190	100	2.28	4.96	9.29	100	2.53	5.91	21.75	<0.001
BDE209	100	10.09	28.09	536.6	100	10.03	23.10	507.3	0.682
Phenols									
4-tert-octylphenol	65	315.5	762	1485.5	64	299	644	1191	0.601
4-n-nonylphenol	63	17	27	61	60	23.5	33	65	0.378
di-NPE	67	1126	1915	3245	65	1359	2046	3735	<0.01

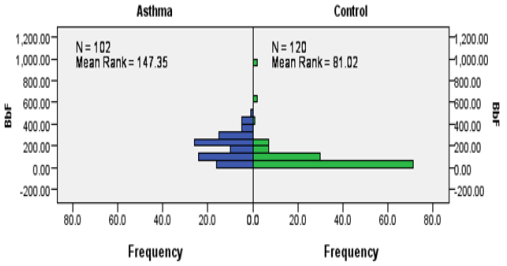

Asthma
Control
N = 72
Mean Rank = 90.94
N = 90
Mean Rank = 73.95
Ace
Frequency

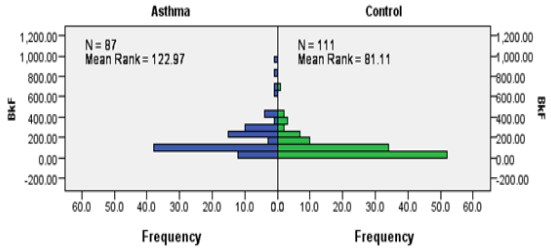

Asthma
Control
N = 72
Mean Rank = 89.04
N = 89
Mean Rank = 74.49
Flue
Frequency

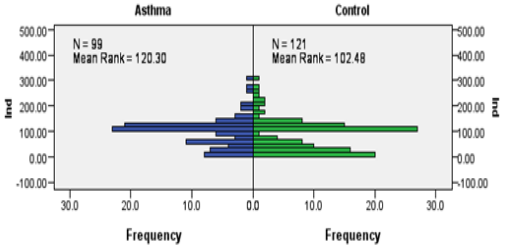

Asthma
Control
N = 72
Mean Rank = 90.24
N = 89
Mean Rank = 73.53
Acy
Frequency

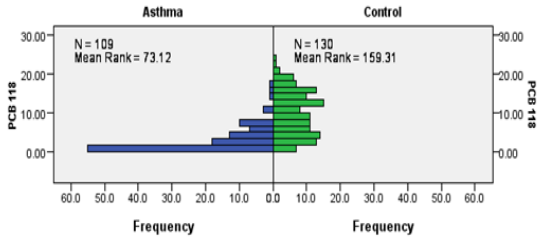

Asthma
Control
N = 72
Mean Rank = 119.65
N = 90
Mean Rank = 50.98
Flre
Frequency

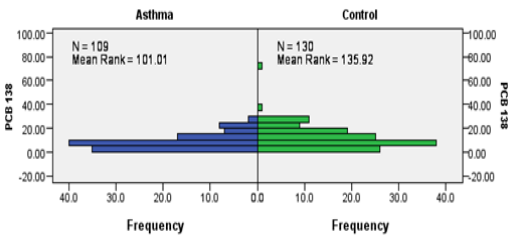

Asthma
Control
N = 72
Mean Rank = 105.19
N = 89
Mean Rank = 61.43
Ant
Frequency


Asthma
Control
N = 102
Mean Rank = 147.35
N = 120
Mean Rank = 81.02
BbF
Frequency


Asthma
Control
N = 87
Mean Rank = 122.97
N = 111
Mean Rank = 81.11
BkF
Frequency


Asthma
Control
N = 99
Mean Rank = 120.30
N = 121
Mean Rank = 102.48
Ind
Frequency


Asthma
Control
N = 109
Mean Rank = 73.12
N = 130
Mean Rank = 159.31
PCB 118
Frequency


Asthma
Control
N = 109
Mean Rank = 101.01
N = 130
Mean Rank = 135.92
PCB 138
Frequency

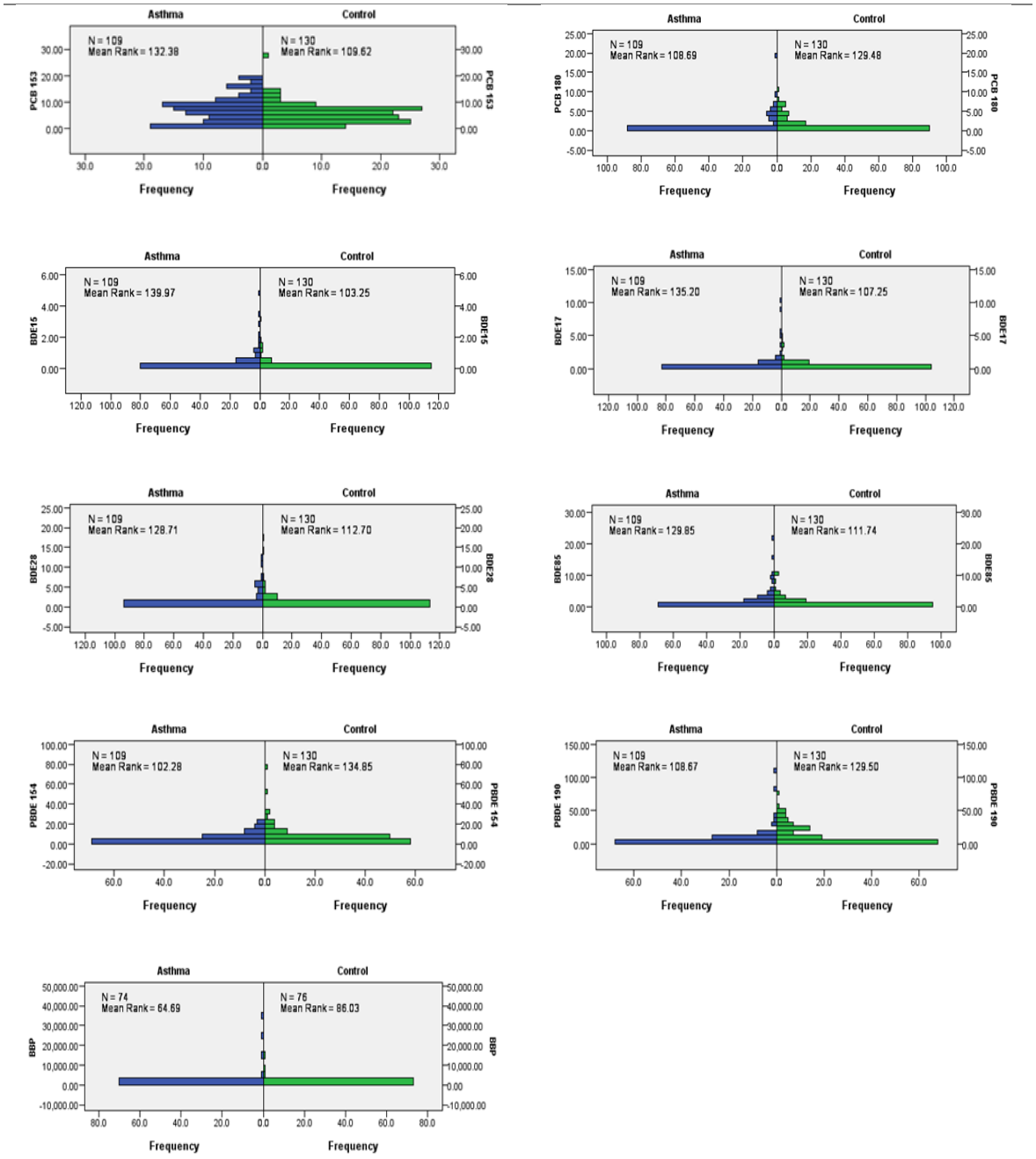


Figure 4-26. Distributions of target pollutants with a statistical difference between their median values in the case and control groups

The results of examining the differences between the case and control groups of homologous and total isomers, apart from individual isomers, are given in Figure 4-28. According to these results, a significant difference was observed in the median values of low molecular weight PAHs and TPAHs between the case and control groups. These isomers, as seen in Figure 4-28a, were measured higher in the case group than in the control group.

When we evaluated PCB isomers, a significant difference emerged between the two groups for Cl5 and TPCB. This shows that the most important isomers affecting the TPCB values measured for the case and control groups are the isomers in the homologous group Cl5 (PCB101 and PCB118). As shown in Figure 4-28b, isomers of PCBs and consequently TPCB values were measured higher in the control group than in the case group. The reason for this is that there were construction activities around some of the houses in the control group, which used materials that are main sources of PCBs. This affected the PCB concentrations in these houses. Images taken from the exteriors of some houses included in the control group are given in Figure 4-27.

Figure 4-27. Images taken around the houses in the control group

When Penta, Octa, Deca and TPBDE values are examined, although there is no significant difference between the case and control groups, the measured values are higher in the case group (Figure 4-28c).

As shown in Figure 4-28d, alkylphenol ethoxylate values differed between the case and control groups, with higher values measured in the control group. As explained in previous sections, the most important source of alkylphenol ethoxylates is detergents. Families of children in the case group prefer less chemical-intensive detergents for cleaning, as recommended by doctors. Therefore, it is thought that alkylphenol ethoxylates were measured lower in dust samples collected from the homes of children with asthma.

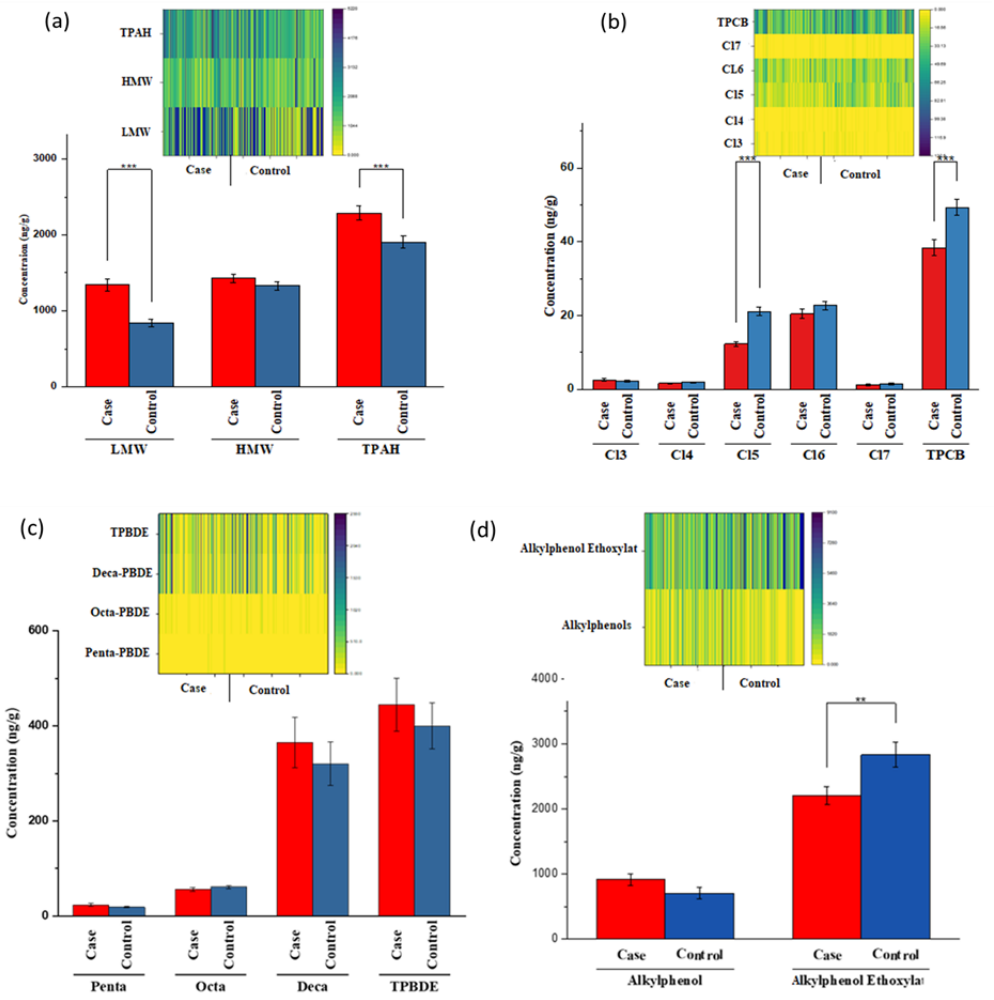


Figure 4-28. The results of examining the differences between the case and control groups of homologous and total isomers

4.7.4 Associations between exposure to typical EDCs and childhood asthma

This study employs logistic regression models to investigate the impact of specific endocrine-disrupting chemicals (EDCs) on asthma in school-aged children. To investigate the impact of target pollutants on the development and severity of childhood asthma, the concentrations of pollutants were categorized as either above or below the median value for each pollutant. Concentrations below the median served as the reference category and the effects of concentrations higher than the median on asthma outcomes were then analyzed.

Each variable was initially included individually to assess its unadjusted association with asthma risk and create model 0 (crude model). The multivariate adjusted model (model 1) was performed with further adjustment for factors that increase the risk of asthma in section 4.7.2. In the multivariable-adjusted model, we included a multiplicative term in

addition to the main impact terms for the two investigated variables to examine the interactions between the environmental factors and individual chemicals. Model 1, adjusted for level of House, dwelling age, number of occupants, type of wall paint, separate kitchen, frying frequency, smoking at home, new furniture, pet keeping, frequency of using detergent, and the material of the child's mattress. The results are given in Table 4-10 and Figure 4-29.

Table 4-10. Odds ratio (OR) of EDCs exposure on school-age children asthma

Case and Control		
	Model 0 (Crude model) OR (95 % CI)	**Model 1** (Multivariate adjusted model) OR (95 % CI)
Nap	1.14 (0.558-2.344)	1.19 (1.291-5.841)
Ace	1.34 (0.658-2.755)	1.01 (0.414-2.494)
Flue	1.51 (0.741-3.411)	1.25 (0.512-3.065)
Acy	1.98 (1.063-4.102)	1.98 (0.853-4.635)
Phe	**5.14 (2.241-7.159)**	**7.02 (4.843-9.191)**
Ant	**3.37 (1.229-5.972)**	**4.35 (2.934-7.738)**
Flt	1.09 (0.602-1.997)	1.04 (0.536-1.019)
Pyr	**1.94 (1.044-3.656)**	**2.62 (1.287-5.335)**
BaA	1.21 (0.659-2.231)	1.18 (0.594-2.373)
Chr	1.44 (0.778-2.688)	1.57 (0.791-3.148)
BbF	**2.51 (1.601-5.651)**	**4.41 (2.176-7.195)**
BkF	**2.54 (1.051-5.218)**	**4.52 (2.354-6.452)**
BaP	1.31 (0.709-2.409)	1.61 (0.818-3.182)
Ind	1.02 (0.545-1.892)	1.08 (0.549-2.117)
DahA	1.32 (0.706-2.486)	1.13 (0.565-2.279)
BgP	1.01 (0.604-2.016)	1.31 (0.676-2.526)
TPAH	**2.25 (1.233-4.112)**	**2.51 (1.262-4.991)**
PCB 28	1.16 (0.701-1.941)	1.36 (0.687-2.704)
PCB 52	1.09 (0.655-1.813)	1.22 (0.691-2.179)
PCB 101	1.48 (0.819-2.703)	1.46 (0.749-2.852)
PCB 118	**9.86 (3.902-12.17)**	**10.4 (5.681-12.85)**
PCB 138	**2.67 (1.421-5.039)**	**2.44 (1.215-4.921)**
PCB 153	**3.72 (1.978-6.988)**	**3.48 (1.751-6.937)**
PCB 166	1.11 (0.621-1.988)	1.29 (0.682-2.469)
PCB 180	1.79 (0.981-3.292)	1.45 (0.748-2.816)
TPCB	**2.18 (1.176-4.048)**	**1.96 (1.207-3.895)**
PBDE 15	**3.18 (1.723-5.901)**	**3.71 (1.863-7.367)**
PBDE 17	1.38 (0.771-2.488)	1.53 (0.796-2.944)
PBDE 28	1.53 (0.916-2.552)	1.26 (0.655-2.427)
PBDE 47	1.67 (0.708-3.944)	1.79 (0.933-3.447)
PBDE 66	1.38 (0.771-2.488)	1.29 (0.686-2.438)
PBDE 71	1.43 (0.794-2.587)	1.55 (0.821-2.951)
PBDE 85	1.51 (0.842-2.723)	1.83 (0.951-3.552)
PBDE 99	1.13 (0.627-2.038)	1.09 (0.574-2.091)
PBDE 100	1.16 (0.701-1.941)	1.03 (0.544-1.963)
PBDE 138	1.03 (0.574-1.859)	1.11 (0.581-2.127)
PBDE 153	1.26 (0.705-2.274)	1.24 (0.655-2.351)
PBDE 154	1.48 (0.819-2.703)	1.36 (0.714-2.606)
PBDE 183	1.06 (0.589-1.903)	1.08 (0.569-2.065)
PBDE 190	**3.47 (1.779-6.774)**	**3.58 (1.719-7.493)**
PBDE 209	1.06 (0.589-1.903)	1.21 (0.633-2.276)
TPBDE	1.11 (0.621-2.016)	1.18 (1.081-3.623)
Alkylphenols	1.38 (0.655-2.931)	1.09 (0.445-2.677)
NPDE	**2.39 (1.312-4.356)**	**2.08 (1.084-4.015)**

p value <0.001 are designated with a bold value.

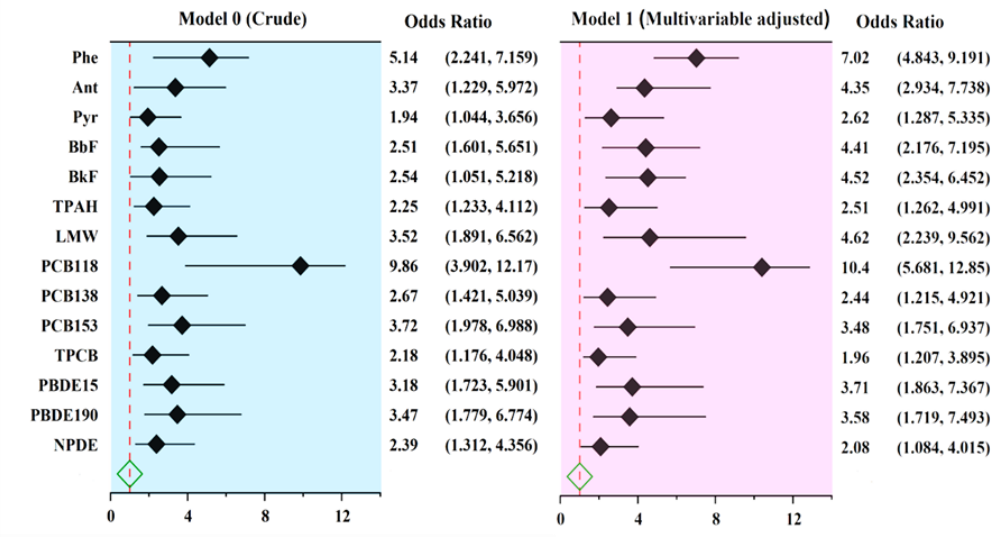


Figure 4-29. Odds ratio for risk of asthma in association with increase in household dust chemicals concentrations

As Table 4-10. Odds ratio (OR) of EDCs exposure on school-age children asthma shows, exposure to Phe, Ant, Pyr, BbF, BkF, TPAH, PCB118, PCB138, PCB153, TPCB, PBDE15, PBDE190, and NPDE were identified as having a significant relationship with the OR (95%CI) of 5.14 (2.241-7.159), 3.37 (1.229-5.972), 1.94 (1.044-3.656), 2.51 (1.601-5.651), 2.54 (1.051-5.218), 2.25 (1.233-4.112), 9.86 (3.902-12.17), 2.67 (1.421-5.039), 3.72 (1.978-6.988), 2.18 (1.176-4.048), 3.18 (1.723-5.901), 3.47 (1.779-6.774), and 2.39 (1.312-4.356), respectively. No significant relationship was found between the remaining chemicals and asthma.

When the cross-effects of chemicals adjusted for the factors specified in Model 1 on the development of asthma were examined, it was generally revealed that the effect increased. There is an approximately twofold increase in error for Pyr, BbF, and BkF. In the adjusted model, a decrease in the impact coefficient was observed only for TPCB.

Figure 4-29 summarizes significant odds ratios for childhood asthma caused by selected EDCs in indoor dust.

In a 2016 study in China by Meng et al., associations between the development of asthma in childhood and typical Hal-POPs in indoor dust were analyzed with continuous unit increments of Hal-POPs concentrations (1 ng/g dust). ORs were adjusted for sex, non-stick pan use, poor ventilation, mildew in house and allergic history. Through the combination of OR values with unit increases (1 ng/g dust) of the concentration, a positive

connection between p,p'-DDE and asthma was discovered. Conversely, BDE-99 and o,p'-DDT had unfavorable outcomes, with ORs that were significantly less than 1. One could argue that the diverse physical and chemical properties of the substances, along with the variations in environmental destiny and individual metabolism, are the sources of the results. The tendency did not change even after being corrected for a number of relevant covariables [175].

Jung et al. reported in an inner-city cohort that indoor air exposure to high concentrations of pyrene (a kind of semi-volatile PAHs) was positively associated with asthma in 5- to 6-year-old children [127]. Positive correlations between urine pyrene and the risk of asthma diagnosis in children aged 6 to 19 and between urinary phenanthrene and the risk of asthma diagnosis in males aged 13 to 19 were discovered by Liu et al. [273]. Children with asthma who were exposed to air phenanthrene and 4- to 6-ring particle bound PAHs showed increased wheeze [274]. In their summary of potential processes, Burton et al. suggested that fetal lung or immune system DNA methylation caused by prenatal exposure to PAHs may be the trigger for childhood asthma [275].

Many studies have revealed an increased risk of asthma, wheezing and eczema in children exposed to PCBs [276–278].

The few studies on the health effects of PBDE exposure have mostly focused on developmental neurotoxicity and possible endocrine disruption rather than respiratory health issues. The few studies investigating the effect of exposure to PBDEs on asthma have not found a relationship [137,174].

Because NPDE are poorly soluble, highly hydrophobic, and have no estrogenic effect, they can accumulate in the human body and cause allergic diseases. NPDE can affect T cells, which are essential for the onset and maintenance of asthma in mice, by altering cytokine production [143,144].

The interaction of environmental factors and chemicals and their effects on asthma is a very complex issue and more research is needed.

4.7.5 Association of pediatric asthma severity with exposure to common household dust chemicals

We used the Global Initiative for Asthma (GINA) spirometry-based definition for asthma severity classification. This classification is based on the Forced Expiratory Volume in the first second (FEV1) percentage of the predicted value after bronchodilator

administration. The spirometry-based classification of asthma severity according to GINA:

Mild Asthma: FEV1 ≥ 80% of the predicted value.

Moderate Asthma: FEV1 between 60-79% of the predicted value.

Severe Asthma: FEV1 < 60% of the predicted value.

In this study, we grouped the asthma severity classification into mild and moderate /severe asthma to evaluate the relationship between target pollutants and asthma severity and considered moderate/mild asthma as the reference to create Logistic Regression moles. As in the previous section, two models (crude and multivariate adjusted) were created. The results are given in Table 4-11 and Figure 4-30.

Table 4-11. Odds ratios for risk of severe asthma in association with EDCs concentrations in children with asthma

Asthma Severity (mild and moderate /severe)		
	Model 0 (Crude model) OR (95 % CI)	**Model 1** (Multivariate adjusted model) OR (95 % CI)
Nap	1.22 (0.419-3.582)	1.67 (0.426-6.569)
Ace	**1.94 (0.677-5.569)**	**3.62 (1.027-5.163)**
Flue	1.71 (0.605-4.836)	2.74 (0.717-8.506)
Acy	**2.64 (1.035-4.452)**	**2.84 (1.091-4.893)**
Phe	1.78 (0.666-4.763)	2.37 (0.807-6.963)
Ant	2.02 (0.761-5.372)	2.31 (0.808-6.611)
Flt	1.57 (0.625-3.966)	1.59 (0.612-4.148)
Pyr	1.31 (0.471-3.672)	1.46 (0.497-4.305)
BaA	1.21 (0.484-3.055)	1.26 (0.474-3.364)
Chr	1.03 (0.397-2.707)	1.13 (0.414-3.098)
BbF	1.41 (0.628-3.171)	1.51 (0.210-1.266)
BkF	1.31 (0.549-3.138)	1.15 (0.455-2.918)
BaP	1.13 (0.463-2.761)	1.43 (0.365-2.385)
Ind	1.52 (0.449-2.864)	1.91 (0.335-2.442)
DahA	1.55 (0.583-4.153)	1.49 (0.541-4.161)
BgP	1.71 (0.704-4.145)	1.51 (0.197-1.312)
Toplam PAH	1.08 (0.482-2.443)	1.18 (0.507-2.751)
HMW	1.22 (0.511-2.961)	1.26 (0.504-3.154)
LWM	1.24 (0.569-2.737)	1.08 (0.433-2.681)
PCB 28	**3.25 (1.183-8.937)**	**3.57 (1.205-5.598)**
PCB 52	1.28 (0.597-2.768)	1.11 (0.491-2.516)
PCB 101	1.61 (0.635-4.054)	1.64 (0.244-1.686)
PCB 118	2.32 (0.234-3.136)	2.69 (0.242-5.029)
PCB 138	1.46 (0.499-4.311)	1.31 (0.421-4.035)
PCB 153	1.31 (0.589-2.877)	1.78 (0.347-1.782)
PCB 166	1.62 (0.689-3.831)	1.68 (0.691-4.114)
PCB 180	2.04 (0.776-5.361)	2.24 (0.801-6.315)
Toplam PCB	1.36 (0.491-3.794)	1.28 (0.445-3.718)
PBDE 15	1.34 (0.609-2.952)	1.38 (0.595-3.216)
PBDE 17	1.06 (0.456-2.475)	1.13 (0.468-2.726)
PBDE 28	1.32 (0.611-2.855)	1.11 (0.451-2.712)

Model 0 (Crude model) OR (95 % CI)	**Model 1** (Multivariate adjusted model) OR (95 % CI)	**Model 0** (Crude model) OR (95 % CI)
PBDE 47	1.65 (0.919-2.984)	1.71 (0.686-4.257)
PBDE 66	1.11 (0.469-2.615)	1.06 (0.418-2.688)
PBDE 71	1.79 (0.764-4.193)	1.81 (0.729-4.446)
PBDE 85	1.15 (0.496-2.661)	1.22 (0.502-2.961)
PBDE 99	1.17 (0.472-2.921)	1.15 (0.426-3.104)
PBDE 100	1.05 (0.428-2.602)	1.05 (0.412-2.691)
PBDE 138	1.13 (0.467-2.743)	1.04 (0.415-2.687)
PBDE 153	1.07 (0.422-2.407)	1.01 (0.408-2.532)
PBDE 154	1.26 (0.485-3.291)	1.17 (0.422-3.271)
PBDE 183	1.23 (0.511-2.961)	1.11 (0.446-2.778)
PBDE 190	1.24 (0.377-4.066)	1.08 (0.307-3.834)
PBDE 209	1.04 (0.425-2.536)	1.06 (0.421-2.777)
PentaBDE	1.08 (0.449-2.607)	1.18 (0.464-3.008)
OctaBDE	1.13 (0.467-2.743)	1.03 (0.401-2.507)
DecaPBDE	1.04 (0.425-2.536)	1.08 (0.421-2.777)
Toplam PBDE	1.24 (0.503-3.077)	1.11 (0.431-2.894)
Alkyıphenols	1.42 (0.637-3.171)	1.44 (0.629-3.339)
NPDE	2.21 (0.716-6.488)	2.01 (0.628-7.004)

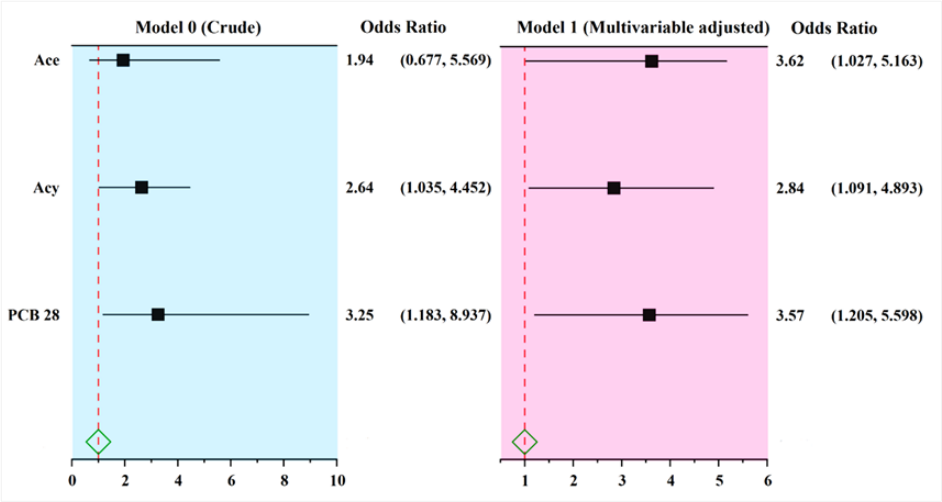


Figure 4-30. Significant odds ratios for childhood asthma by EDCs in indoor dust

According to Logistic Regression models (Table 4-11 and Figure 4-30), only Ace, Acy and PCB28, among the identified EDCs analyzed in house dust samples, affect pulse intensity. According to the results of Model 1, the rate of impact on asthma severity increases as a result of the interaction between different environmental factors and chemicals. This value has increased 3 times for Ace.

It is thought that the fact that the asthmatic children included in this study were more often classified as having mild asthma and that there was no statistical balance between the asthma severity classes affected the scope of these models. Future research should aim to address the limitations identified, particularly by employing longitudinal designs,

expanding geographical and temporal coverage, and ensuring a balanced representation of asthma severity classes to enhance the robustness and generalizability of the findings. Li and colleagues have presented evidence suggesting that PAHs present in diesel exhaust can trigger a sequence of oxidative stress events that ultimately result in airway inflammation [279]. In a study examining exposure to air pollutants in Southern California, variants of the PAH-metabolizing enzyme microsomal epoxide hydrolase, which lead to elevated levels of reactive oxygen species, were linked to a heightened risk of asthma. Furthermore, this risk was found to increase with the proximity of residences to freeways [280]. A preliminary study involving 70 participants has indicated an association between the same PAH metrics utilized in this study and several adverse respiratory outcomes. These outcomes include decreased Forced Expiratory Volume in 1 second (FEV1), increased asthma severity, and the suppression of regulatory T-cell function. This suppression is believed to occur through methylation of the FoxP3 gene, which is responsible for upregulating the function of these cells [281].

In a study involving Japanese children, Nakanishi observed a high frequency of respiratory symptoms in response to exposure to high levels of PCBs [282].

5 CONCLUSION

The relationship between EDCs exposure in indoor dust and asthma has not been widely investigated. Hence, we were strongly motivated to investigate the associations between selected EDCs in indoor dust and the development of childhood asthma. Here, the primary purpose of this thesis is to examine the effects of indoor environmental quality, which is determined by the analysis of endocrine-disrupting chemicals (EDCs) (persistent organic pollutants and surface-active substance) of house dust, on the development of asthma in school-age children. To our knowledge, this study is a novelty in Türkiye and the only study investigating the relationships between typical EDCs in indoor dust and childhood asthma. The concentrations of selected endocrine-disrupting chemicals (EDCs) analyzed in house dust samples from case (asthmatic) and control (non-asthmatic) children. The house dust were collected from the homes of 110 asthmatic and 130 control school-aged children and were analyzed for persistent organic pollutants (POPs) (polycyclic aromatic hydrocarbons (PAHs), polychlorinated biphenyls (PCBs), and polybrominated diphenyl ethers (PBDEs)), alkylphenols (4-tert-octylphenol (4-t-OP), and 4-n-nonylphenol) and alkylphenol ethoxylates (nonylphenol diethoxylate (di-NPE)).

We analyzed dust for the 16 EPA Priority PAHs. Some of the PAHs we analyzed are semi-volatile organic carbons (1, 2, 3 rings), while others are heavy molecular weight PAHs (4,5,6 rings). Flt (11%), Chr (9%), and Pyr (9%) were determined as the dominant isomers in the ∑16 PAHs. High molecular weight 4, 5 and 6 ring PAH isomers (BaA, BbF, BkF, BaP, IcdP, DahA, and BghiP) constitute 65% of the total measured PAH concentrations, and low molecular weight PAH isomers (Nap, Acy, Ace, Flu, Phe, Ant, Flt, Pyr, and Chr) constitute 45% of the total PAH amount. There was a strong correlation between low molecular weight PAH isomers (Nap, Ace, Flue, Acy, Phe, and Ant), suggesting that they originate from similar sources. Specifically, correlations were noted between Ind and BgP, which are associated with vehicle exhaust emissions, and between BaA, BbF, BkF, and BaP isomers, which are linked to natural gas and cooking.

A survey was conducted during the sampling period through personal interviews with families residing in the households of participating children. The survey encompassed inquiries about various aspects of the houses (e.g., floor type, age of the house, proximity to the main street) and the living habits of family members (e.g., smoking indoors,

frequency of frying, frequency of house cleaning). According to the findings of Kruskal-Wallis test, 4-5-6 ring PAHs were linked with traffic emissions, likely due to proximity to the main street. 2 and 3-ring PAHs showed associations with different factors, including the heating system, and smoking at home.

According to the PMF model results, 4 factors can be defined to determine the sources of PAH isomers measured in house dust. The first factor (F1) was predominantly loaded by Flt, Pyr, BbF, BkF, and DahA, which are related to traffic sources, accounting for 19% of the total PAHs. The second factor (F2) was dominated by Nap, Ace, Flue, Acy, Phe, and Ant contributing with 48% of the total PAHs. F2 is acknowledged as a petrogenic source that is typically abundant in 2- and 3-ringed PAHs that are discharged into the environment as a result of incomplete combustion, crude and fuel oil spills, and other incidents. The third factor (F3) contained 12% of the total PAHs and was characterized by BaA and Chr, which are typical indicators of traffic sources. The fourth factor (F4) showed the dominance of BaP, Ind, and BgP and made 21% of total PAHs. High-ring PAHs were previously found to be primarily released outdoors, such as from gasoline combustion engines. Conversely, BaP is a common genotoxin found in cigarette smoke. As a result, emissions from vehicle emissions and cigarette smoke revealed a mixed source for factor 4.

PCB concentrations measured in a total of 240 indoor dust samples collected from homes. The median value of ∑8 PCBs measured in dust samples collected from a total of 240 houses was 68.4 ng/g (11.7 – 264.7 ng/g). Based on the results, the highest measured PCB in indoor dust was PCB101, with a value of 8.5 ng/g. PCB138 and PCB153 were detected as PCB isomers with high concentrations, measuring 8.1 and 5.8 ng/g, respectively. The least common isomer found in homes was PCB180, with a median value of 0.4 ng/g. The most dominant PCB homolog group is 6-chlorinated biphenyls, comprising 51.18% of the total. This is followed by 5-chlorinated biphenyls at 31.99%, 3 and 4-chlorinated biphenyls at 4.09%, and 7-chlorinated biphenyls at 1.06%. Additionally, the measured PCBs were found to be like Arochlor 1254 and 1260, which are commercial production PCBs. The correlation between PCB isomers generally indicates that they have similar sources in the indoor environment.

According to the results of the Kruskal-Wallis Test, the concentrations of PCB isomers in the collected and analyzed dust samples were influenced by several factors, including proximity to the main street, frequency of house cleaning, recent purchases of new items

(including electronic devices), and home repairs and painting conducted within the last year.

Results of the PMF analysis indicated two identified factors that are characteristic for PCBs found in indoor dust. The first factor (F1) was predominantly loaded by Cl5 and Cl6 PCBs, accounting for 37.6% of the total PCBs. This factor can be defined as vehicle exhaust and combustion emissions for heating purposes. The second factor consists of low molecular weight 3,4 and 5 chlorinated PCBs compared to the other factor and explains 62.4 % of the total variance. While low molecular weight PCBs are associated with gasoline exhaust emissions, PCB-28, 44, 52, 101, 118 are shown as emissions from coal and wood combustion.

15 PBDEs concentrations measured in a total of 240 indoor dust samples collected from homes. The median value of total PBDEs measured in 240 dust samples in this study was 431.6 ng/g (ranging from 16.5 to 2544 ng/g). As a result, BDE-209 was measured as the highest pollutant with a median value of 26.6 ng/g. The results indicate a significant correlation between PBDE isomers. Strong correlations between penta BDEs suggest that these isomers likely share common sources. Furthermore, the correlation between the commercial PBDE 209 isomer and other isomers highlights televisions, computers, plastic toys, and textile products as the primary sources of PBDEs in indoor environments.

The age of a house can significantly influence the levels and types of PBDEs present. In this study, PBDE isomers released into the environment from new items purchased in the last year affected the PBDE values measured in the collected dust ($P<0.05$). Especially BDE 209 was measured higher in the dust samples collected from the homes of children whose beds were made of wood/plastic. According to these results, it has been estimated that the use of PBDE isomers as fire retardant in products made of wood or plastic materials, such as the child's bed, is one of the sources of PBDE in houses made of this material.

Results of the PMF analysis indicated two identified factors that are characteristic for PBDEs found in indoor dust. The contribution frequencies of factors 1 and 2 to ∑PBDEs are 6.9% and 93.1%, respectively, suggesting that the primary source of ∑PBDEs in household dust is the use of electrical and electronic equipment.

The range of 4-n-NP concentration in residential dust was 49–8740 ng/g , with a median of 520 ng/ g. The highest concentrations were observed for di-NPE in indoor dust, with a

median value of 1910 ng/g (ranging from 113 to 9070 ng/g). Among the APs, 4-t-OP showed lower concentrations in all the settled dust (35 ng/g, ranging from 8 to 227 ng/g). Our findings indicate that house characteristics, such as apartment versus house and the presence of polished wood flooring, as well as residents' behaviors, including household size, frequency of detergent use for cleaning, home repairs, and purchasing habits, significantly influence the levels of APs and APEs in residential dust. Measures to reduce human exposure to non-ionic surfactants (APs and APEs) in house dust, such as employing herbal detergents for indoor cleaning and enhancing ventilation systems, could be effective strategies to mitigate this exposure. These results serve as a valuable contribution to raising awareness about the significance of alkylphenols and alkylphenol ethoxylates, as well as reducing their associated health risks and environmental impact. While the daily exposure levels are currently minimal, there is cause for concern due to the lack of regulations on these surfactants in many developing nations, where their use may be increasing.

Logistic regression models were employed to examine the impact of the lifestyles and environmental conditions of the families on the risk of asthma in children. When OR values are examined, living on the first or lower floors, using wallpaper as wall covering, not having a separate kitchen, and having pets can affect the development of asthma approximately 4 times. However, living in houses built before 20 years, having more than 4 people at home, frying two or more times a week, smoking at home, using new furniture, frequency of detergent use, and using wool for the child's mattress can affect the development of asthma approximately 2 times.

Compared with the above, several traditionally accepted risk factors, such as polished wood flooring, wool carpet, wood windows, near the main street, and repair and painting in the last year (sampling period), were not statistically associated with the occurrence of childhood asthma.

Apart from individual isomers, we examined the differences of homologous and total isomers between the case and control groups. According to these results, a significant difference was observed in the median values of low molecular weight PAHs and TPAHs between case and control groups. These isomers were measured higher in the case group than in the control group. When we evaluated PCB isomers, a significant difference emerged between the two groups for Cl5 and TPCB. This shows that the most important isomers affecting the TPCB values measured for the case and control groups are the isomers in the homologous group Cl5 (PCB101 and PCB118). Isomers of PCBs and

consequently TPCB values were measured higher in the control group than in the case group. The reason for this is that there were construction activities around some of the houses in the control group, which used materials that are main sources of PCBs. This affected the PCB concentrations in these houses. When Penta, Octa, Deca and TPBDE values are examined, although there is no significant difference between the case and control groups, the measured values are higher in the case group.

Alkylphenol ethoxylate values differed between the case and control groups, with higher values measured in the control group. The most important source of alkylphenol ethoxylates is detergents. Families of children in the case group prefer less chemical-intensive detergents for cleaning, as recommended by doctors. Therefore, it is thought that alkylphenol ethoxylates were measured lower in dust samples collected from the homes of children with asthma.

This study employs logistic regression models to investigate the impact of specific endocrine-disrupting chemicals (EDCs) on asthma in school-aged children. Each variable was initially included individually to assess its unadjusted association with asthma risk and create model 0 (crude model). Model 1 (multivariate adjusted model), adjusted for level of House, dwelling age, number of occupants, type of wall paint, separate kitchen, frying frequency, smoking at home, new furniture, pet keeping, frequency of using detergent, and the material of the child's mattress. Exposure to Phe, Ant, Pyr, BbF, BkF, TPAH, PCB118, PCB138, PCB153, TPCB, BDE15, BDE190, and NPDE were identified as having a significant relationship with asthma. According to OR values, Phe (5 times), Ant (3 times), Pyr (2 times), BbF (about 2 times), BkF (about 2 times), TPAH (about 2 times), PCB118 (about 10 times), PCB138 (about 2 times), PCB153 (3 times), TPCB (2 times), BDE15 (3 times), BDE190 (3 times), and NPDE (2 times) may be effective in the development of asthma.

When the cross-effects of chemicals adjusted for the factors specified in Model 1 on the development of asthma were examined, it was generally revealed that the effect increased. There is an approximately twofold increase in error for Pyr, BbF, and BkF. In the adjusted model, a decrease in the impact coefficient was observed only for TPCB.

In this study, we grouped the asthma severity classification into mild and moderate /severe asthma to evaluate the relationship between target pollutants and asthma severity and considered moderate/mild asthma as the reference to create Logistic Regression moles. According to Logistic Regression models, only Ace, Acy and PCB28, among the identified EDCs analyzed in house dust samples, affect pulse intensity. According to the

results of Model 1, the rate of impact on asthma severity increases as a result of the interaction between different environmental factors and chemicals. This value has increased 3 times for Ace.

It is thought that the fact that the asthmatic children included in this study were more often classified as having mild asthma and that there was no statistical balance between the asthma severity classes affected the scope of these models. Future research should aim to address the limitations identified, particularly by employing longitudinal designs, expanding geographical and temporal coverage, and ensuring a balanced representation of asthma severity classes to enhance the robustness and generalizability of the findings. Despite its limitations, this study significantly advances the understanding of how indoor air pollutants affect asthma severity in children. It underscores the need for targeted interventions to reduce exposure to harmful pollutants in indoor environments, particularly in homes with vulnerable populations such as children with asthma. Future research should aim to address the limitations identified, particularly by employing longitudinal designs, expanding geographical and temporal coverage, and ensuring a balanced representation of asthma severity classes to enhance the robustness and generalizability of the findings.

5.1 Recommendations for future studies

In recent times, endocrine researchers, regulatory bodies, and public and political bodies at the national and EU levels have been paying more and more attention to EDCs. The degree of interest gave ESE the chance to share its knowledge at the highest political echelons, aid in raising awareness, and guarantee the availability of impartial scientific data regarding EDCs.

A strategy for the prevention of exposures to EDCs is urgently needed. This will require different strategies at the level of individual capacity and beyond, by developing and implementing recommendations towards the protection of individuals and prevention strategies by policymakers and local governors. Education programs in schools and hospitals will be helpful to improve the general understanding of EDCs and the consequences of exposure to such pollutants, especially in early life. The majority of health professionals (general doctors, specialist doctors, dentists, physiotherapists, chiropodists, psychologists, nurses, etc.) receive no initial training at all on endocrine disruptors but could have to answer questions on the subject from patients. Chemical

risks in general, and those associated with the substitution of dangerous substances in particular, are too rarely an innovation factor or a factor integrated into innovations. They should be taken into account from the design stage by industrial sectors.

Studies on Endocrine Disrupting Chemicals (EDCs) are expected to be carried out cooperatively with specialists from all around the world for the following reasons:

- Improving comprehension: Joint research projects seek to expand our knowledge of the causes, consequences, and dangers of exposure to EDCs.
- Identifying sources: Studies can characterize the sources of EDCs, including their distribution, paths of exposure, and environmental destiny, by combining their expertise.
- Evaluating health effects: Cooperative research projects provide thorough evaluations of the long- and short-term consequences of EDC exposure on human health and the environment.
- Creating mitigation strategies: By collaborating, scientists may create practical plans to lessen the dangers associated with EDC exposure to the environment and public health.
- Informing policy and regulation: Collaborative research provides robust evidence to inform policy-making and regulatory decisions aimed at minimizing EDC exposure and protecting public health and the environment.

In general, cooperation between specialists from various fields and geographical areas is crucial to expanding our understanding of EDCs and creating workable plans to deal with related issues.

To encourage independent research in this field, a concerted financial effort is required in addition to better laws and practices. People will still be exposed to substances that have the potential to seriously harm their health and well-being if further efforts are not made.

6 REFERENCES

[1] L.A. Wallace, E.D. Pellizzari, T.D. Hartwell, R. Whitmore, C. Sparacino, H. Zelon, Total exposure assessment methodology (team) study: Personal exposures, indoor-outdoor relationships, and breath levels of volatile organic compounds in New Jersey, Environ Int 12 **(1986)** 369–387. https://doi.org/10.1016/0160-4120(86)90051-6.

[2] I. Asher, S. Montefort, B. Björkstén, C.K.W. Lai, D.P. Strachan, S.K. Weiland, H. Williams, I. Phase, Worldwide time trends in the prevalence of symptoms of asthma, allergic rhinoconjunctivitis, and eczema in childhood: ISAAC Phases One and Three repeat multicountry cross-sectional surveys, **(2006)**. www.thelancet.com.

[3] J. Heinrich, Influence of indoor factors in dwellings on the development of childhood asthma, Int J Hyg Environ Health 214 **(2011)** 1–25. https://doi.org/10.1016/j.ijheh.2010.08.009.

[4] W. Eder, M.J. Ege, E. Von Mutius, The Asthma Epidemic, **(2006)**. www.nejm.org. (Accessed 19 May 2024).

[5] E. Yardımcısı Doç Ömür AYDIN, A. Kronik Tedavisi Basamak Tedavisi, G. Türkiye Ulusal Allerji ve Klinik İmmünoloji Derneği, Astım Tanı ve Tedavi Rehberi, n.d. **(2020).** https://www.toraks.org.tr. (Accessed 24 May 2024)

[6] E. Yardımcıları, Ö. Soyer, D. Ömür, A. Ankara, T. Ulusal, A. Ve Klinik, İ. Derneği, Astım Tanı ve Tedavi Rehberi 2020 Güncellemesi, n.d**.(2020).** https://www.toraks.org.tr. (Accessed 19 May 2024)

[7] PROGRESS MONITOR **(2017),** n.d. (Accessed 20 May 2024)

[8] O. Hänninen, A.B. Knol, M. Jantunen, T.A. Lim, A. Conrad, M. Rappolder, P. Carrer, A.C. Fanetti, R. Kim, J. Buekers, R. Torfs, I. Iavarone, T. Classen, C. Hornberg, O.C.L. Mekel, Environmental burden of disease in Europe: Assessing nine risk factors in six countries, Environ Health Perspect 122 **(2014)** 439–446. https://doi.org/10.1289/ehp.1206154.

[9] A. Demond, A. Franzblau, D. Garabrant, X. Jiang, P. Adriaens, Q. Chen, B. Gillespie, W. Hao, B. Hong, O. Jolliet, J. Lepkowski, Human exposure from dioxins in soil, Environ Sci Technol 46 **(2012)** 1296–1302. https://doi.org/10.1021/es2022363.

[10] B. TÜRK, Important Factors Affecting the Quality of Indoor Air and a Bibliometric Analysis, Sakarya University Journal of Science 26 **(2022)** 608–619. https://doi.org/10.16984/saufenbilder.996443.

[11] Indoor air pollution : national burden of disease estimates, (n.d.). **(2019).** https://www.who.int/publications/i/item/WHO-SDE-PHE-07.01rev (accessed April 30, 2024).

[12] Indoor Air Pollution - Our World in Data, (n.d.). **(2015)** https://ourworldindata-org.translate.goog/indoor-air (accessed May 3, 2024).

[13] Household air pollution, (n.d.). **(2016)** https://www.who.int/news-room/fact-sheets/detail/household-air-pollution-and-health (accessed April 30, 2024).

[14] J.C. Nwanaji-Enwerem, J.G. Allen, P.I. Beamer, Another invisible enemy indoors: COVID-19, human health, the home, and United States indoor air policy, J Expo Sci Environ Epidemiol 30 **(2020)** 773–775. https://doi.org/10.1038/s41370-020-0247-x.

[15] C.J. Weschler, N. Carslaw, Indoor Chemistry, Environ Sci Technol 52 **(2018)** 2419–2428. https://doi.org/10.1021/acs.est.7b06387.

[16] R.I. Adams, A.C. Bateman, H.M. Bik, J.F. Meadow, Microbiota of the indoor environment: a meta-analysis, Microbiome 3 **(2015)** 49. https://doi.org/10.1186/s40168-015-0108-3.

[17] A. Besis, E. Botsaropoulou, D. Balla, D. Voutsa, C. Samara, Toxic organic pollutants in Greek house dust: Implications for human exposure and health risk, Chemosphere 284 **(2021).** https://doi.org/10.1016/j.chemosphere.2021.131318.

[18] H.M. Hwang, E.K. Park, T.M. Young, B.D. Hammock, Occurrence of endocrine-disrupting chemicals in indoor dust, Science of the Total Environment 404 **(2008)** 26–35. https://doi.org/10.1016/j.scitotenv.2008.05.031.

[19] A.C. Dirtu, A. Covaci, Estimation of daily intake of organohalogenated contaminants from food consumption and indoor dust ingestion in Romania, Environ Sci Technol 44 **(2010)** 6297–6304. https://doi.org/10.1021/es101233z.

[20] S.K. Dutta, P.S. Mitra, S. Ghosh, S. Zang, D. Sonneborn, I. Hertz-Picciotto, T. Trnovec, L. Palkovicova, E. Sovcikova, S. Ghimbovschi, E.P. Hoffman, Differential gene expression and a functional analysis of PCB-exposed children: Understanding disease

and disorder development, Environ Int 40 **(2012)** 143–154. https://doi.org/10.1016/j.envint.2011.07.008.

[21] T.P. Whitehead, F.R. Brown, C. Metayer, J.S. Park, M. Does, M.X. Petreas, P.A. Buffler, S.M. Rappaport, Polybrominated diphenyl ethers in residential dust: Sources of variability, Environ Int 57–58 **(2013)** 11–24. https://doi.org/10.1016/j.envint.2013.03.003.

[22] P. Xu, X. Lou, G. Ding, H. Shen, L. Wu, Z. Chen, J. Han, G. Han, X. Wang, Association of PCB, PBDE and PCDD/F body burdens with hormone levels for children in an e-waste dismantling area of Zhejiang Province, China, Science of the Total Environment 499 **(2014)** 55–61. https://doi.org/10.1016/j.scitotenv.2014.08.057.

[23] M. Lorber, Exposure of Americans to polybrominated diphenyl ethers, J Expo Sci Environ Epidemiol 18 **(2008)** 2–19. https://doi.org/10.1038/sj.jes.7500572.

[24] T. Damstra, S. Barlow, A. Bergman, R. Kavlock, G. Van, D. Kraak, International Programme on Chemical Safety Global Assessment Of The Science Of Endocrine Disruptors N.D. **(2020)** (Accessed 19 May 2024)

[26] L.N. Vandenberg, T. Colborn, T.B. Hayes, J.J. Heindel, D.R. Jacobs, D.H. Lee, T. Shioda, A.M. Soto, F.S. vom Saal, W. V. Welshons, R.T. Zoeller, J.P. Myers, Hormones and endocrine-disrupting chemicals: Low-dose effects and nonmonotonic dose responses, Endocr Rev 33 **(2012)** 378–455. https://doi.org/10.1210/er.2011-1050.

[27] T.M. Crisp, E.D. Clegg, R.L. Cooper, W.R. Wood, D.G. Anderson, K.R. Baetcke, J.L. Hoffmann, M.S. Morrow, D.J. Rodier, J.E. Schaeffer, L.W. Touart, M.G. Zeeman, Y.M. Patel4, Environmental Endocrine Disruption: An Effects Assessment and Analysis. **(2019)** (Accessed 19 May 2024)

[28] Å. Bergman, J.J. Heindel, S. Jobling, K.A. Kidd, R. Thomas Zoeller, State of the Science of Endocrine Disrupting Chemicals-**(2012)** Inter-Organızatıon Programme For The Sound Management Of Chemıcals, N.D.

[29] E.A. Cohen Hubal, T. de Wet, L. Du Toit, M.P. Firestone, M. Ruchirawat, J. van Engelen, C. Vickers, Identifying important life stages for monitoring and assessing risks from exposures to environmental contaminants: Results of a World Health Organization review, Regulatory Toxicology and Pharmacology 69 **(2014)** 113–124. https://doi.org/10.1016/j.yrtph.2013.09.008.

[30] S. Flint, T. Markle, S. Thompson, E. Wallace, Bisphenol A exposure, effects, and policy: A wildlife perspective, J Environ Manage 104 **(2012)** 19–34. https://doi.org/10.1016/j.jenvman.2012.03.021.

[31] H.M. Shin, T.E. McKone, N.S. Tulve, M.S. Clifton, D.H. Bennett, Indoor residence times of semivolatile organic compounds: Model estimation and field evaluation, Environ Sci Technol 47 **(2013)** 859–867. https://doi.org/10.1021/es303316d.

[32] W. Wei, C. Mandin, O. Ramalho, Influence of indoor environmental factors on mass transfer parameters and concentrations of semi-volatile organic compounds, Chemosphere 195 **(2018)** 223–235. https://doi.org/10.1016/j.chemosphere.2017.12.072.

[33] P.S. Mitra, S. Ghosh, S. Zang, D. Sonneborn, I. Hertz-Picciotto, T. Trnovec, L. Palkovicova, E. Sovcikova, S. Ghimbovschi, E.P. Hoffman, S.K. Dutta, Analysis of the toxicogenomic effects of exposure to persistent organic pollutants (POPs) in Slovakian girls: Correlations between gene expression and disease risk, Environ Int 39 **(2012)** 188–199. https://doi.org/10.1016/j.envint.2011.09.003.

[34] D.H. Lee, D.R. Jacobs, T. Kocher, Associations of serum concentrations of persistent organic pollutants with the prevalence of periodontal disease and subpopulations of white blood cells, Environ Health Perspect 116 **(2008)** 1558–1562. https://doi.org/10.1289/ehp.11425.

[35] G. Meng, Y. Feng, Z. Nie, X. Wu, H. Wei, S. Wu, Y. Yin, Y. Wang, Internal exposure levels of typical POPs and their associations with childhood asthma in Shanghai, China, Environ Res 146 **(2016)** 125–135. https://doi.org/10.1016/j.envres.2015.12.026.

[36] A. Sjödin, D.G. Patterson, Å. Bergman Åke, A review on human exposure to brominated flame retardants - Particularly polybrominated diphenyl ethers, Environ Int 29 **(2003)** 829–839. https://doi.org/10.1016/S0160-4120(03)00108-9.

[37] K. Ni, Y. Lu, T. Wang, K. Kannan, J. Gosens, L. Xu, Q. Li, L. Wang, S. Liu, A review of human exposure to polybrominated diphenyl ethers (PBDEs) in China, Int J Hyg Environ Health 216 **(2013)** 607–623. https://doi.org/10.1016/j.ijheh.2013.02.002.

[38] M. Andersson, R.T. Ottesen, T. Volden, Building materials as a source of PCB pollution in Bergen, Norway, Science of the Total Environment 325 **(2004)** 139–144. https://doi.org/10.1016/j.scitotenv.2003.11.014.

[39] C. Vanden Bilcke, The Stockholm Convention on Persistent Organic Pollutants, American Journal of International Law 95 **(2001)** 692–708. https://doi.org/10.2307/2668517.

[40] Stockholm Convention - Home page, (n.d.). https://www.pops.int/ (accessed May 16, **2024**).

[41] R.L. Miller, R. Garfinkel, M. Horton, D. Camann, F.P. Perera, R.M. Whyatt, P.L. Kinney, Polycyclic Aromatic Hydrocarbons, Environmental Tobacco Smoke, and Respiratory Symptoms in an Inner-city Birth Cohort*, (**2004).** www.chestjournal.orgCHEST/126/4/.

[42] T.T.T. Dong, B.K. Lee, Characteristics, toxicity, and source apportionment of polycylic aromatic hydrocarbons (PAHs) in road dust of Ulsan, Korea, Chemosphere 74 **(2009)** 1245–1253. https://doi.org/10.1016/j.chemosphere.2008.11.035.

[43] A.K. Haritash, C.P. Kaushik, Biodegradation aspects of Polycyclic Aromatic Hydrocarbons (PAHs): A review, J Hazard Mater 169 **(2009)** 1–15. https://doi.org/10.1016/j.jhazmat.2009.03.137.

[44] R.M. Maertens, X. Yang, J. Zhu, R.W. Gagne, G.R. Douglas, P.A. White, Mutagenic and carcinogenic hazards of settled house dust I: Polycyclic aromatic hydrocarbon content and excess lifetime cancer risk from preschool exposure, Environ Sci Technol 42 **(2008)** 1747–1753. https://doi.org/10.1021/es702449c.

[45] Integrated Risk Information System | US EPA, (n.d.). https://www.epa.gov/iris (accessed May 16, **2024**).

[46] P.R.S. Kodavanti, Polychlorinated Biphenyls (PCBs), in: Encyclopedia of the Neurological Sciences, Elsevier Inc., **(2014)** pp. 917–921. https://doi.org/10.1016/B978-0-12-385157-4.00271-2.

[47] Polychlorınated Bıphenyls And Polybromınated Bıphenyls Volume 107 Iarc Monographs On The Evaluatıon Of Carcınogenıc Rısks To Humans, N.D. (Accessed 19 May **2024**)

[48] E.P. Dekoning, W. Karmaus, PCB exposure in utero and via breast milk. A review, (**2000)**. www.nature.com/jea. (Accessed 19 May 2024)

[49] Compilation of EU Dioxin Exposure and Health Data Summary Report Report produced for European Commission DG Environment Compilation of EU Dioxin

Exposure and Health Data Summary Report Report produced for, (**1999)**. (Accessed 21 May 2024)

[50] Fire safety requirements and alternatives to brominated flame-retardants : a LOUS follow-up project, Miljøstyrelsen, **(2016)**. (Accessed 19 May 2024)

[51] L.S. Birnbaum, D.F. Staskal, Brominated flame retardants: Cause for concern?, Environ Health Perspect 112 **(2004)** 9–17. https://doi.org/10.1289/ehp.6559.

[52] M.A. Siddiqi, R.H. Laessig, K.D. Reed, Polybrominated Diphenyl Ethers (PBDEs): New Pollutants-Old Diseases, **(2003).** http://www.mfldclin.edu/clinmedres.

[53] G.M. Solomon, P.M. Weiss, Chemical Contaminants in Breast Milk: Time Trends and Regional Variability, **(2002).** http://ehpnet1.niehs.nih.gov/docs/2002/110pA339-A347solomon/abstract.html.

[54] P.O. Darnerud, Toxic effects of brominated flame retardants in man and in wildlife, Environ Int 29 **(2003)** 841–853. https://doi.org/10.1016/S0160-4120(03)00107-7.

[55] M.A. Siddiqi, R.H. Laessig, K.D. Reed, Polybrominated Diphenyl Ethers (PBDEs): New Pollutants-Old Diseases, **(2003)**. http://www.mfldclin.edu/clinmedres.

[56] P.O. Darnerud, G.S. Eriksen, T. Jóhannesson, P.B. Larsen, M. Viluksela, Polybrominated diphenyl ethers: Occurence, dietary exposure, and toxicology, Environ Health Perspect 109 **(2001)** 49–68. https://doi.org/10.1289/ehp.01109s149.

[57] Fire safety requirements and alternatives to brominated flame-retardants : a LOUS follow-up project, Miljøstyrelsen, 2016. (Accessed 19 May **2024**)

[58] G.-G. Ying, B. Williams, R. Kookana, Environmental fate of alkylphenols and alkylphenol ethoxylates-a review, n.d. www.elsevier.com/locate/envint. (Accessed 19 May **2024**)

[59] M.V. Alan, L.B. Barber, J.L. Gray, E.M. Lopez, J.D. Woodling, D.O. Norris, Reproductive disruption in fish downstream from an estrogenic wastewater effluent, Environ Sci Technol 42 (**2008**) 3407–3414. https://doi.org/10.1021/es0720661.

[60] C. Lassen, Danmark. Miljøstyrelsen, Survey of alkylphenols and alkylphenol ethoxylates : part of the LOUS-review, Environmental Protection Agency, (**2013**).

[61] Fact Sheet: Nonylphenols and Nonylphenol Ethoxylates, n.d. http://www.epa.gov/dfe/pubs/projects. (Accessed 19 May **2024**)

[62] T.Y. Chiu, N. Paterakis, E. Cartmell, M.D. Scrimshaw, J.N. Lester, A critical review of the formation of mono-and dicarboxylated metabolic intermediates of alkylphenol polyethoxylates during wastewater treatment and their environmental significance, Crit Rev Environ Sci Technol 40 **(2010)** 199–238. https://doi.org/10.1080/10643380802219517.

[63] S. Datta, J.E. Loyo-Rosales, C.P. Rice, A simple method for the determination of trace levels of alkylphenolic compounds in fish tissue using pressurized fluid extraction, solid phase cleanup, and high-performance liquid chromatography fluorescence detection, J Agric Food Chem 50 **(2002)** 1350–1354. https://doi.org/10.1021/jf0111357.

[64] X. Lu, M. Chen, X. Zhang, Y. Sun, D. Zhu, Q. Zhang, B. Wang, Z. Zhang, Simultaneous quantification of five phenols in settled house dust using ultra-high performance liquid chromatography-tandem mass spectrometry, Analytical Methods 5 **(2013)** 5339–5344. https://doi.org/10.1039/c3ay40602d.

[65] R.A. Rudel, D.E. Camann, J.D. Spengler, L.R. Korn, J.G. Brody, Phthalates, alkylphenols, pesticides, polybrominated diphenyl ethers, and other endocrine-disrupting compounds in indoor air and dust, Environ Sci Technol 37 **(2003)** 4543–4553. https://doi.org/10.1021/es0264596.

[66] R. Goto, T. Kubota, Y. Ibuki, K. Kaji, A. Goto, Degradation of nonylphenol polyethoxylates by ultraviolet B irradiation and effects of their products on mammalian cultured cells, Toxicology 202 **(2004)** 237–247. https://doi.org/10.1016/j.tox.2004.05.017.

[67] X. Wang, X. Han, Y. Hou, G. Yao, Y. Wang, Effect of nonylphenol on apoptosis of Sertoli cells in vitro, Bull Environ Contam Toxicol 70 **(2003)** 898–904. https://doi.org/10.1007/s00128-003-0067-4.

[68] M. Zumbado, L.D. Boada, S. Torres, J.G. Monterde, B.N. Díaz-Chico, J.L. Afonso, J.J. Cabrera, A. Blanco, Evaluation of acute hepatotoxic effects exerted by environmental estrogens nonylphenol and 4-octylphenol in immature male rats, **(2002).** www.elsevier.com/locate/toxicol.

[69] I.A. Sheikh, Molecular interactions of thyroxine binding globulin and thyroid hormone receptor with estrogenic compounds 4-nonylphenol, 4-tert-octylphenol and bisphenol A metabolite (MBP), Life Sci 253 **(2020).** https://doi.org/10.1016/j.lfs.2020.117738.

[70] J.A. Adewunmi, Fate Of Nonylphenol, Nonylphenol Monoethoxylate, Nonylphenol Dıethoxylate, Octylphenol, And Bısphenol A In Sludge, Bıosolıds And Bıosolıds-Amended Soıls, **(2015).** (Accessed 19 May 2024)

[71] A. Kanazawa, I. Saito, A. Araki, M. Takeda, M. Ma, Y. Saijo, R. Kishi, Association between indoor exposure to semi-volatile organic compounds and building-related symptoms among the occupants of residential dwellings, Indoor Air 20 **(2010)** 72–84. https://doi.org/10.1111/j.1600-0668.2009.00629.x.

[72] N.K. Wilson, J.C. Chuang, C. Lyu, R. Menton, M.K. Morgan, Aggregate exposures of nine preschool children to persistent organic pollutants at day care and at home, J Expo Anal Environ Epidemiol 13 **(2003)** 187–202. https://doi.org/10.1038/sj.jea.7500270.

[73] C. Kubwabo, P.E. Rasmussen, X. Fan, I. Kosarac, G. Grenier, K. Coleman, Simultaneous quantification of bisphenol A, alkylphenols and alkylphenol ethoxylates in indoor dust by gas chromatography-tandem mass spectrometry and a comparison between two sampling techniques, Analytical Methods 8 **(2016)** 4093–4100. https://doi.org/10.1039/c6ay00774k.

[74] O.A. Abafe, T.B. Chokwe, J.O. Okonkwo, B.S. Martincigh, Alkylphenols and alkylphenol ethoxylates in dust from homes, offices and computer laboratories: Implication for personal exposure via inadvertent dust ingestion, Emerg Contam 3 **(2017)** 127–131. https://doi.org/10.1016/j.emcon.2018.01.001.

[75] L. Cevhertas, I. Ogulur, D.J. Maurer, D. Burla, M. Ding, K. Jansen, J. Koch, C. Liu, S. Ma, Y. Mitamura, Y. Peng, U. Radzikowska, A.O. Rinaldi, P. Satitsuksanoa, A. Globinska, W. van de Veen, M. Sokolowska, K. Baerenfaller, Y. dong Gao, I. Agache, M. Akdis, C.A. Akdis, Advances and recent developments in asthma in 2020, Allergy: European Journal of Allergy and Clinical Immunology 75 **(2020)** 3124–3146. https://doi.org/10.1111/all.14607.

[76] K.F. Chung, I.M. Adcock, Precision medicine for the discovery of treatable mechanisms in severe asthma, Allergy: European Journal of Allergy and Clinical Immunology 74 **(2019)** 1649–1659. https://doi.org/10.1111/all.13771.

[77] Y. Guo, J.Y. Moon, C.C. Laurie, K.E. North, L.A.P. Sanchez-Johnsen, S. Davis, B. Yu, S.M. Nyenhuis, R. Kaplan, D. Rastogi, Q. Qi, Genetic predisposition to obesity is associated with asthma in US Hispanics/Latinos: Results from the Hispanic Community Health Study/Study of Latinos, Allergy: European Journal of Allergy and Clinical Immunology 73 **(2018)** 1547–1550. https://doi.org/10.1111/all.13450.

[78] T.A. Olafsdottir, F. Theodors, K. Bjarnadottir, U.S. Bjornsdottir, Eighty-eight variants highlight the role of T cell regulation and airway remodeling in asthma pathogenesis, Nat Commun 11 **(2020).** https://doi.org/10.1038/s41467-019-14144-8.

[79] M. Guarnieri, J.R. Balmes, Asthma 1 Outdoor air pollution and asthma, **(2014).** www.thelancet.com. (Accessed 19 May 2024)

[80] Z. Celebi Sözener, L. Cevhertas, K. Nadeau, M. Akdis, C.A. Akdis, et al. Environmental factors in epithelial barrier dysfunction, Journal of Allergy and Clinical Immunology 145 **(2020)** 1517–1528. https://doi.org/10.1016/J.JACI.2020.04.024.

[81] GINA-2023-Full-report-23_07_06-WMS (1), (Accessed 19 May **2024**)

[82] E. Yardımcıları, Ö. Soyer, D. Ömür, A. Ankara, T. Ulusal, A. Ve Klinik, İ. Derneği, Astım Tanı ve Tedavi Rehberi **2020** Güncellemesi, n.d. https://www.toraks.org.tr. (Accessed 20 May 2024)

[83] B. Yazar, A. Meydanlioglu, The prevalence and associated factors of asthma, allergic rhinitis, and eczema in Turkish children and adolescents, Pediatr Pulmonol 57 **(2022)** 2491–2501. https://doi.org/10.1002/ppul.26065.

[84] V. Çelik, H. Tanrıverdi, F. Kılıç, T. Tutal, Prevalence of Asthma and Allergic Diseases Among Children in Adıyaman, Türkiye: a Cross-sectional Study, Çocuk Dergisi / Journal of Child 23 **(2023)** 77–82. https://doi.org/10.26650/jchild.2023.1353232.

[85] A. Baççıoğlu, A. Söğüt, Ö. Kılıç, E. Beyhun, The prevalence of allergic diseases and associated risk factors in school-age children and adults in Erzurum, Turkey, Turk Toraks Dergisi 16 **(2015)** 68–72. https://doi.org/10.5152/ttd.2015.4229.

[86] B. Mikkelsen, The Global Asthma Report 2022, International Journal of Tuberculosis and Lung Disease 26 **(2022).** https://doi.org/10.5588/ijtld.22.1010.

[87] M.R. Sears, Epidemiology of asthma exacerbations, Journal of Allergy and Clinical Immunology 122 **(2008)** 662–668. https://doi.org/10.1016/j.jaci.2008.08.003.

[88] I. Agache, C.A. Akdis, Precision medicine and phenotypes, endotypes, genotypes, regiotypes, and theratypes of allergic diseases, Journal of Clinical Investigation 129 **(2019)** 1493–1503. https://doi.org/10.1172/JCI124611.

[89] E. Toskala, D.W. Kennedy, Asthma risk factors, Int Forum Allergy Rhinol 5 **(2015)** S11–S16. https://doi.org/10.1002/alr.21557.

[90] M. Barne, Gaps in asthma diagnosis and treatment in low- and middle-income countries, Frontiers in Allergy 4 **(2023).** https://doi.org/10.3389/falgy.2023.1240259.

[91] F. Wu, T.K. Takaro, Childhood asthma and environmental interventions, Environ Health Perspect 115 **(2007)** 971–975. https://doi.org/10.1289/ehp.8989.

[92] G. Drago, S. Ruggieri, F. Bianchi, S. Sampino, F. Cibella, Birth Cohorts in Highly Contaminated Sites: A Tool for Monitoring the Relationships Between Environmental Pollutants and Children's Health, Front Public Health 8 **(2020).** https://doi.org/10.3389/fpubh.2020.00125.

[93] J.J. Heeringa, C.I. McKenzie, N. Varese, M. Hew, A.T.C.M. Bakx, P.M. Aui, J.M. Rolland, R.E. O'Hehir, M.C. van Zelm, Induction of IgG2 and IgG4 B-cell memory following sublingual immunotherapy for ryegrass pollen allergy, Allergy: European Journal of Allergy and Clinical Immunology 75 **(2020)** 1121–1132. https://doi.org/10.1111/all.14073.

[94] S.I. Yang, S.Y. Lee, H. Bin Kim, H.C. Kim, J.H. Leem, H.J. Yang, H. Kwon, J.H. Seo, H.J. Cho, J. Yoon, E. Lee, Y.H. Jung, Y. Kim, S. Jung, H.J. Kwon, S.J. Hong, Prenatal particulate matter affects new asthma via airway hyperresponsiveness in schoolchildren, Allergy: European Journal of Allergy and Clinical Immunology 74 **(2019)** 675–684. https://doi.org/10.1111/all.13649.

[95] F. Feo-Brito, T. Alfaya Arias, M. Amo-Salas, M.L. Somoza Álvarez, E. Haroun Díaz, C. Mayorga Mayorga, R. Fernández Santamaría, J.M. Urra Ardanaz, Clinical impact and immunological alterations in asthmatic patients allergic to grass pollen subjected to high urban pollution in Madrid, Clinical and Experimental Allergy 52 **(2022)** 530–539. https://doi.org/10.1111/cea.14041.

[96] F. Feo Brito, P. Mur Gimeno, C. Martínez, A. Tobías, L. Suárez, F. Guerra, J.M. Borja, A.M. Alonso, Air pollution and seasonal asthma during the pollen season. A cohort study in Puertollano and Ciudad Real (Spain), Allergy: European Journal of Allergy and Clinical Immunology 62 **(2007)** 1152–1157. https://doi.org/10.1111/j.1398-9995.2007.01438.x.

[97] E. Toskala, D.W. Kennedy, Asthma risk factors, Int Forum Allergy Rhinol 5 **(2015)** S11–S16. https://doi.org/10.1002/alr.21557.

[98] A.J. Burbank, A.K. Sood, M.J. Kesic, D.B. Peden, M.L. Hernandez, Environmental determinants of allergy and asthma in early life, Journal of Allergy and Clinical Immunology 140 **(2017)** 1–12. https://doi.org/10.1016/j.jaci.2017.05.010.

[99] H.G. Margolis, J.K. Mann, F.W. Lurmann, K.M. Mortimer, J.R. Balmes, S.K. Hammond, I.B. Tager, Altered pulmonary function in children with asthma associated with highway traffic near residence, Int J Environ Health Res 19 **(2009)** 139–155. https://doi.org/10.1080/09603120802415792.

[100] K.M. Shea, R.T. Truckner, R.W. Weber, D.B. Peden, Climate change and allergic disease, Journal of Allergy and Clinical Immunology 122 **(2008)** 443–453. https://doi.org/10.1016/j.jaci.2008.06.032.

[101] M. Jerrett, K. Shankardass, K. Berhane, W.J. Gauderman, N. Künzli, E. Avol, F. Gilliland, F. Lurmann, J.N. Molitor, J.T. Molitor, D.C. Thomas, J. Peters, R. McConnell, Traffic-related air pollution and asthma onset in children: A prospective cohort study with individual exposure measurement, Environ Health Perspect 116 **(2008)** 1433–1438. https://doi.org/10.1289/ehp.10968.

[102] L.J. Akinbami, J.E. Moorman, P.L. Garbe, E.J. Sondik, Status of childhood asthma in the United States, 1980-2007, Pediatrics 123 **(2009).** https://doi.org/10.1542/peds.2008-2233C.

[103] J. Bai, J. Zhao, K.-L. Shen, L.I. Xiang, A.-H. Chen, S. Huang, Y. Huang, J.-S. Wang $, R.-W. Ye, Current Trends of the Prevalence of Childhood Asthma in Three Chinese Cities : A Multicenter Epidemiological Survey 1, **(2010).** www.besjournal.com.

[104] S.E. Bergström, G. Boman, L. Eriksson, H. Formgren, T. Foucard, L.G. Hörte, C. Janson, U. Spetz-Nyström, G. Hedlin, Asthma mortality among Swedish children and

young adults, a 10-year study, Respir Med 102 **(2008)** 1335–1341. https://doi.org/10.1016/j.rmed.2008.03.020.

[105] M.S. Jassal, Special considerations-asthma in children, Int Forum Allergy Rhinol 5 **(2015)** S61–S67. https://doi.org/10.1002/alr.21577.

[106] D. V. Henley, K.S. Korach, Endocrine-disrupting chemicals use distinct mechanisms of action to modulate endocrine system function, Endocrinology 147 **(2006)**. https://doi.org/10.1210/en.2005-1117.

[107] I. Paciência, J. Cavaleiro Rufo, D. Silva, C. Martins, F. Mendes, M. Farraia, L. Delgado, E. de Oliveira Fernandes, P. Padrão, P. Moreira, M. Severo, H. Barros, A. Moreira, Exposure to indoor endocrine-disrupting chemicals and childhood asthma and obesity, Allergy: European Journal of Allergy and Clinical Immunology 74 **(2019)** 1277–1291. https://doi.org/10.1111/all.13740.

[108] World Health Organization, State of the science of endocrine disrupting chemicals 2012, **(2012).**

[109] S. Benjamin, E. Masai, N. Kamimura, K. Takahashi, R.C. Anderson, P.A. Faisal, Phthalates impact human health: Epidemiological evidences and plausible mechanism of action, J Hazard Mater 340 **(2017)** 360–383. https://doi.org/10.1016/j.jhazmat.2017.06.036.

[110] S.N. Yang, C.C. Hsieh, H.F. Kuo, M.S. Lee, M.Y. Huang, C.H. Kuo, C.H. Hung, The effects of environmental toxins on allergic inflammation, Allergy Asthma Immunol Res 6 **(2014)** 478–484. https://doi.org/10.4168/aair.2014.6.6.478.

[111] I. Paciência, J. Cavaleiro Rufo, D. Silva, C. Martins, F. Mendes, M. Farraia, L. Delgado, E. de Oliveira Fernandes, P. Padrão, P. Moreira, M. Severo, H. Barros, A. Moreira, Exposure to indoor endocrine-disrupting chemicals and childhood asthma and obesity, Allergy: European Journal of Allergy and Clinical Immunology 74 **(2019)** 1277–1291. https://doi.org/10.1111/all.13740.

[112] R.E. Dodson, M. Nishioka, L.J. Standley, L.J. Perovich, J.G. Brody, R.A. Rudel, Endocrine disruptors and asthma-associated chemicals in consumer products, Environ Health Perspect 120 **(2012)** 935–943. https://doi.org/10.1289/ehp.1104052.

[113] J.S. Helm, M. Nishioka, J.G. Brody, R.A. Rudel, R.E. Dodson, Measurement of endocrine disrupting and asthma-associated chemicals in hair products used by Black women, Environ Res 165 **(2018)** 448–458. https://doi.org/10.1016/j.envres.2018.03.030.

[114] C. Franken, N. Lambrechts, E. Govarts, G. Koppen, E. Den Hond, D. Ooms, S. Voorspoels, L. Bruckers, I. Loots, V. Nelen, I. Sioen, T.S. Nawrot, W. Baeyens, N. Van Larebeke, G. Schoeters, Phthalate-induced oxidative stress and association with asthma-related airway inflammation in adolescents, Int J Hyg Environ Health 220 **(2017)** 468–477. https://doi.org/10.1016/j.ijheh.2017.01.006.

[115] M. Gascon, M. Casas, E. Morales, D. Valvi, A. Ballesteros-Gómez, N. Luque, S. Rubio, N. Monfort, R. Ventura, D. Martínez, J. Sunyer, M. Vrijheid, Prenatal exposure to bisphenol A and phthalates and childhood respiratory tract infections and allergy, Journal of Allergy and Clinical Immunology 135 **(2015)** 370-378.e7. https://doi.org/10.1016/j.jaci.2014.09.030.

[116] K.M. Donohue, R.L. Miller, M.S. Perzanowski, A.C. Just, L.A. Hoepner, S. Arunajadai, S. Canfield, D. Resnick, A.M. Calafat, F.P. Perera, R.M. Whyatt, Prenatal and postnatal bisphenol A exposure and asthma development among inner-city children, Journal of Allergy and Clinical Immunology 131 **(2013)**. https://doi.org/10.1016/j.jaci.2012.12.1573.

[117] I.J. Wang, W.J.J. Karmaus, S.L. Chen, J.W. Holloway, S. Ewart, Effects of phthalate exposure on asthma may be mediated through alterations in DNA methylation, Clin Epigenetics 7 **(2015)**. https://doi.org/10.1186/s13148-015-0060-x.

[118] G.W.K. Wong, C.M. Chow, Childhood asthma epidemiology: Insights from comparative studies of rural and urban populations, Pediatr Pulmonol 43 **(2008)** 107–116. https://doi.org/10.1002/ppul.20755.

[119] M. Gascon, E. Morales, J. Sunyer, M. Vrijheid, Effects of persistent organic pollutants on the developing respiratory and immune systems: A systematic review, Environ Int 52 **(2013)** 51–65. https://doi.org/10.1016/j.envint.2012.11.005.

[120] P. Duramad, I.B. Tager, N.T. Holland, Cytokines and other immunological biomarkers in children's environmental health studies, Toxicol Lett 172 **(2007)** 48–59. https://doi.org/10.1016/j.toxlet.2007.05.017.

[121] A. Araki, I. Saito, A. Kanazawa, K. Morimoto, K. Nakayama, E. Shibata, M. Tanaka, T. Takigawa, T. Yoshimura, H. Chikara, Y. Saijo, R. Kishi, Phosphorus flame retardants in indoor dust and their relation to asthma and allergies of inhabitants, Indoor Air 24 **(2014)** 3–15. https://doi.org/10.1111/ina.12054.

[122] W. Jedrychowski, F.P. Perera, R. Whyatt, E. Mroz, E. Flak, R. Jacek, A. Penar, J. Spengler, D. Camman, Wheezing and lung function measured in subjects exposed to various levels of fine particles and polycyclic aromatic hydrocarbons, Cent Eur J Med 2 **(2007)** 66–78. https://doi.org/10.2478/s11536-006-0043-6.

[123] H. Bömmel, M. Li-Weber, E. Serfling, A. Duschl, The environmental pollutant pyrene induces the production of IL-4, Journal of Allergy and Clinical Immunology 105 **(2000)** 796–802. https://doi.org/10.1067/mai.2000.105124.

[124] H. Bömmel, M. Haake, P. Luft, J. Horejs-Hoeck, H. Hein, J. Bartels, C. Schauer, U. Pöschl, M. Kracht, A. Duschl, The diesel exhaust component pyrene induces expression of IL-8 but not of eotaxin, Int Immunopharmacol 3 **(2003)** 1371–1379. https://doi.org/10.1016/S1567-5769(03)00135-8.

[125] R.L. Miller, R. Garfinkel, M. Horton, D. Camann, F.P. Perera, R.M. Whyatt, P.L. Kinney, Polycyclic Aromatic Hydrocarbons, Environmental Tobacco Smoke, and Respiratory Symptoms in an Inner-city Birth Cohort*, **(2004)**. www.chestjournal.orgCHEST/126/4/.

[126] M.J. Rosa, K.H. Jung, M.S. Perzanowski, E.A. Kelvin, K.W. Darling, D.E. Camann, S.N. Chillrud, R.M. Whyatt, P.L. Kinney, F.P. Perera, R.L. Miller, Prenatal exposure to polycyclic aromatic hydrocarbons, environmental tobacco smoke and asthma, Respir Med 105 **(2011)** 869–876. https://doi.org/10.1016/j.rmed.2010.11.022.

[127] K.H. Jung, B. Yan, K. Moors, S.N. Chillrud, M.S. Perzanowski, R.M. Whyatt, L. Hoepner, I. Goldstein, B. Zhang, D. Camann, P.L. Kinney, F.P. Perera, R.L. Miller, Repeated exposure to polycyclic aromatic hydrocarbons and asthma: Effect of seroatopy, Annals of Allergy, Asthma and Immunology 109 **(2012)** 249–254. https://doi.org/10.1016/j.anai.2012.07.019.

[128] S.L. Gale, E.M. Noth, J. Mann, J. Balmes, S.K. Hammond, I.B. Tager, Polycyclic aromatic hydrocarbon exposure and wheeze in a cohort of children with asthma in Fresno, CA, J Expo Sci Environ Epidemiol 22 **(2012)** 386–392. https://doi.org/10.1038/jes.2012.29.

[129] I.J. Wang, W.J.J. Karmaus, C.C. Yang, Polycyclic aromatic hydrocarbons exposure, oxidative stress, and asthma in children, Int Arch Occup Environ Health 90 **(2017)** 297–303. https://doi.org/10.1007/s00420-017-1198-y.

[130] F.P. Perera, V. Rauh, R.M. Whyatt, W.Y. Tsai, D. Tang, D. Diaz, L. Hoepner, D. Barr, Y.H. Tu, D. Camann, P. Kinney, Effect of prenatal exposure to airborne polycyclic aromatic hydocarbons on neurodevelopment in the first 3 years of life among inner-city children, Environ Health Perspect 114 **(2006)** 1287–1292. https://doi.org/10.1289/ehp.9084.

[131] J. Liu, L. Zhang, L.C. Winterroth, M. Garcia, S. Weiman, J.W. Wong, J.B. Sunwoo, K.C. Nadeau, Epigenetically mediated pathogenic effects of phenanthrene on regulatory T cells, J Toxicol 2013 **(2013).** https://doi.org/10.1155/2013/967029.

[132] A.M. Padula, J.R. Balmes, E.A. Eisen, J. Mann, E.M. Noth, F.W. Lurmann, B. Pratt, I.B. Tager, K. Nadeau, S.K. Hammond, Ambient polycyclic aromatic hydrocarbons and pulmonary function in children, J Expo Sci Environ Epidemiol 25 **(2015)** 295–302. https://doi.org/10.1038/jes.2014.42.

[133] N.M. Al-Daghri, M.S. Alokail, S.H. Abd-Alrahman, H.M. Draz, S.M. Yakout, M. Clerici, Polycyclic aromatic hydrocarbon exposure and pediatric asthma in children: A case-control study, Environ Health 12 **(2013).** https://doi.org/10.1186/1476-069X-12-1.

[134] I. Hertz-Picciotto, H.Y. Park, M. Dostal, A. Kocan, T. Trnovec, R. Sram, Prenatal exposures to persistent and non-persistent organic compounds and effects on immune system development, in: Basic Clin Pharmacol Toxicol, **(2008)**. pp. 146–154. https://doi.org/10.1111/j.1742-7843.2007.00190.x.

[135] F. Dallaire, É. Dewailly, C. Vézina, G. Muckle, J.P. Weber, S. Bruneau, P. Ayotte, Effect of prenatal exposure to polychlorinated biphenyls on incidence of acute respiratory infections in preschool inuit children, Environ Health Perspect 114 **(2006)** 1301–1305. https://doi.org/10.1289/ehp.8683.

[136] S.B. Stølevik, U.C. Nygaard, E. Namork, M. Haugen, H.E. Kvalem, H.M. Meltzer, J. Alexander, J.H.M. van Delft, H. van Loveren, M. Løvik, B. Granum, Prenatal exposure to polychlorinated biphenyls and dioxins is associated with increased risk of wheeze and infections in infants, Food and Chemical Toxicology 49 **(2011)** 1843–1848. https://doi.org/10.1016/j.fct.2011.05.002.

[137] G. Meng, Y. Feng, Z. Nie, X. Wu, H. Wei, S. Wu, Y. Yin, Y. Wang, Internal exposure levels of typical POPs and their associations with childhood asthma in Shanghai, China, Environ Res 146 **(2016)** 125–135. https://doi.org/10.1016/j.envres.2015.12.026.

[138] R.L. Van Den Heuvel, G. Koppen, J.A. Staessen, E. Den Hond, G. Verheyen, T.S. Nawrot, H.A. Roels, R. Vlietinck, G.E.R. Schoeters, Immunologic Biomarkers in Relation to Exposure Markers of PCBs and Dioxins in Flemish Adolescents (Belgium), **(2002)**. http://ehpnet1.niehs.nih.gov/docs/2002/110p595-600vandenheuvel/abstract.html.

[139] M. Levin, B. Morsey, C. Mori, P.R. Nambiar, S. De Guise, Non-coplanar PCB-mediated modulation of human leukocyte phagocytosis: A new mechanism for immunotoxicity, J Toxicol Environ Health A 68 **(2005**) 1977–1993. https://doi.org/10.1080/15287390500227126.

[140] C.M. Vezina, N.J. Walker, J.R. Olson, Subchronic exposure to TCDD, PeCDF, PCB126, and PCB153: Effect on hepatic gene expression, Environ Health Perspect 112 **(2004)** 1636–1644. https://doi.org/10.1289/txg.7253.

[141] A.S. Ahmed, D.A. Ibrahim, T.H. Hassan, W.G. Abd-El-Azem, Prevalence and predictors of occupational asthma among workers in detergent and cleaning products industry and its impact on quality of life in El Asher Men Ramadan, Egypt, Environmental Science and Pollution Research 29 **(2022)** 33901–33908. https://doi.org/10.1007/s11356-022-18558-8.

[142] I. Folletti, A. Siracusa, G. Paolocci, Update on asthma and cleaning agents, Curr Opin Allergy Clin Immunol 17 **(2017)** 90–95. https://doi.org/10.1097/ACI.0000000000000349.

[143] M. Iwata, Y. Eshima, H. Kagechika, H. Miyaura, The endocrine disruptors nonylphenol and octylphenol exert direct effects on T cells to suppress Th1 development and enhance Th2 development, Immunol Lett 94 **(2004)** 135–139. https://doi.org/10.1016/j.imlet.2004.04.013.

[144] M.H. Lee, E. Kim, T.S. Kim, Exposure to 4-tert-octylphenol, an environmentally persistent alkylphenol, enhances interleukin-4 production in T cells via NF-AT activation, Toxicol Appl Pharmacol 197 (2004) 19–28. https://doi.org/10.1016/j.taap.2004.02.003.

[145] J. Heinrich, Influence of indoor factors in dwellings on the development of childhood asthma, Int J Hyg Environ Health 214 **(2011)** 1–25. https://doi.org/10.1016/j.ijheh.2010.08.009.

[146] I. Asher, S. Montefort, B. Björkstén, C.K.W. Lai, D.P. Strachan, S.K. Weiland, H. Williams, I. Phase, Worldwide time trends in the prevalence of symptoms of asthma, allergic rhinoconjunctivitis, and eczema in childhood: ISAAC Phases One and Three repeat multicountry cross-sectional surveys, **(2006)**. www.thelancet.com.

[147] W. Eder, M.J. Ege, E. Von Mutius, The Asthma Epidemic, **(2006)**. www.nejm.org. (Accessed 19 May 2024)

[148] E. Yardımcıları Özge SOYER Doç Ömür AYDIN, T. Ulusal Allerji ve Klinik İmmünoloji Derneği, Çocukluk Çağı Astım Tanı ve Tedavi Rehberi El Kitapçığı **(2020)**, n.d. https://www.toraks.org.tr.

[149] N.S. Ozcan, K.M. Cubukcu, Evaluation of Air Pollution Effects on Asthma Disease: The case of Izmir, Procedia Soc Behav Sci 202 **(2015)** 448–455. https://doi.org/10.1016/j.sbspro.2015.08.201.

[150] S. Mentese, N.A. Mirici, M.T. Otkun, C. Bakar, E. Palaz, D. Tasdibi, S. Cevizci, O. Cotuker, Association between respiratory health and indoor air pollution exposure in Canakkale, Turkey, Build Environ 93 **(2015)** 72–83. https://doi.org/10.1016/j.buildenv.2015.01.023.

[151] M. Ali CENGİZ, T. Şenel, E. Terzi, N. Savaş, Y. Terzi, O. Mayıs Üniversitesi, İ. Bölümü, E. Üniversitesi, İ. Bölümü, Samsun Bölgesindeki Hava Kirliliğinin Neden Olduğu Hastalıkların İstatistiksel Modellenmesi, **(2013).** http://kfbd.giresun.edu.tr.

[152] E. Kara, H.G. Özdilek, E.E. Kara, Ambient air quality and asthma cases in Niğde, Turkey, Environmental Science and Pollution Research 20 **(2013)** 4225–4234. https://doi.org/10.1007/s11356-012-1376-0.

[153] B. Mehmet Berktas, Effects of atmospheric sulphur dioxide and particulate matter concentrations on emergency room admissions due to asthma in Ankara, **(2014)**. https://www.researchgate.net/publication/8563470.

[154] M. Saygın, T. Gonca, Ö. Öztürk, M. Has, S. Çalışkan, Z.G. Has, A. Akkaya, To investigate the effects of air pollution (PM10 and SO2) on the respiratory diseases asthma

and chronic obstructive pulmonary disease, Turk Thorac J 18 **(2017)** 33–39. https://doi.org/10.5152/TurkThoracJ.2017.16016.

[155] N. Ali, N. Van den Eede, A.C. Dirtu, H. Neels, A. Covaci, Assessment of human exposure to indoor organic contaminants via dust ingestion in Pakistan, Indoor Air 22 **(2012)** 200–211. https://doi.org/10.1111/j.1600-0668.2011.00757.x.

[156] PROGRESS MONITOR **2017**, (Accessed 19 May 2024)

[157] J. Metsälä, A. Lundqvist, L.J. Virta, M. Kaila, M. Gissler, S.M. Virtanen, Prenatal and post-natal exposure to antibiotics and risk of asthma in childhood, Clinical and Experimental Allergy 45 **(2015)** 137–145. https://doi.org/10.1111/cea.12356.

[158] G.H. Dong, K.Y. Tung, C.H. Tsai, M.M. Liu, D. Wang, W. Liu, Y.H. Jin, W.S. Hsieh, Y.L. Lee, P.C. Chen, Serum polyfluoroalkyl concentrations, asthma outcomes, and immunological markers in a case-control study of Taiwanese children, Environ Health Perspect 121 **(2013)** 507–513. https://doi.org/10.1289/ehp.1205351.

[159] S. Hansen, M. Strøm, S.F. Olsen, E. Maslova, P. Rantakokko, H. Kiviranta, D. Rytter, B.H. Bech, L. V. Hansen, T.I. Halldorsson, Maternal concentrations of persistent organochlorine pollutants and the risk of asthma in offspring: Results from a prospective cohort with 20 years of follow-up, Environ Health Perspect 122 **(2014)** 93–99. https://doi.org/10.1289/ehp.1206397.

[160] P. Karimi, K.O. Peters, K. Bidad, P.T. Strickland, Polycyclic aromatic hydrocarbons and childhood asthma, Eur J Epidemiol 30 **(2015)** 91–101. https://doi.org/10.1007/s10654-015-9988-6.

[161] L.J. Akinbami, K.C. Schoendorf, Trends in Childhood Asthma: Prevalence, Health Care Utilization, and Mortality, **(2002).** http://publications.aap.org/pediatrics/article

[162] J. Liu, L. Zhang, L.C. Winterroth, M. Garcia, S. Weiman, J.W. Wong, J.B. Sunwoo, K.C. Nadeau, Epigenetically mediated pathogenic effects of phenanthrene on regulatory T cells, J Toxicol 2013 **(2013).** https://doi.org/10.1155/2013/967029.

[163] R.L. Miller, R. Garfinkel, M. Horton, D. Camann, F.P. Perera, R.M. Whyatt, P.L. Kinney, Polycyclic Aromatic Hydrocarbons, Environmental Tobacco Smoke, and Respiratory Symptoms in an Inner-city Birth Cohort*, **(2004).** www.chestjournal.orgCHEST/126/4/.

[164] F.P. Perera, V. Rauh, R.M. Whyatt, W.Y. Tsai, D. Tang, D. Diaz, L. Hoepner, D. Barr, Y.H. Tu, D. Camann, P. Kinney, Effect of prenatal exposure to airborne polycyclic aromatic hydocarbons on neurodevelopment in the first 3 years of life among inner-city children, Environ Health Perspect 114 **(2006)** 1287–1292. https://doi.org/10.1289/ehp.9084.

[165] The 12 Initial POPs, https://chm.pops.int/TheConvention/ThePOPs/The12InitialPOPs/tabid/296/Default.aspx (accessed May 4, **2024**).

[166] N. Ali, N. Van den Eede, A.C. Dirtu, H. Neels, A. Covaci, Assessment of human exposure to indoor organic contaminants via dust ingestion in Pakistan, Indoor Air 22 **(2012)** 200–211. https://doi.org/10.1111/j.1600-0668.2011.00757.x.

[167] X. Zheng, F. Xu, K. Chen, Y. Zeng, X. Luo, S. Chen, B. Mai, A. Covaci, Flame retardants and organochlorines in indoor dust from several e-waste recycling sites in South China: Composition variations and implications for human exposure, Environ Int 78 **(2015)** 1–7. https://doi.org/10.1016/j.envint.2015.02.006.

[168] Q. Dai, X. Min, M. Weng, A review of polychlorinated biphenyls (PCBs) pollution in indoor air environment, J Air Waste Manage Assoc 66 **(2016)** 941–950. https://doi.org/10.1080/10962247.2016.1184193.

[169] K. Kademoglou, F. Xu, J.A. Padilla-Sanchez, L.S. Haug, A. Covaci, C.D. Collins, Legacy and alternative flame retardants in Norwegian and UK indoor environment: Implications of human exposure via dust ingestion, Environ Int 102 **(2017)** 48–56. https://doi.org/10.1016/j.envint.2016.12.012.

[170] S. Harrad, C. Ibarra, M. Diamond, L. Melymuk, M. Robson, J. Douwes, L. Roosens, A.C. Dirtu, A. Covaci, Polybrominated diphenyl ethers in domestic indoor dust from Canada, New Zealand, United Kingdom and United States, Environ Int 34 **(2008)** 232–238. https://doi.org/10.1016/j.envint.2007.08.008.

[171] A. Araki, I. Saito, A. Kanazawa, K. Morimoto, K. Nakayama, E. Shibata, M. Tanaka, T. Takigawa, T. Yoshimura, H. Chikara, Y. Saijo, R. Kishi, Phosphorus flame retardants in indoor dust and their relation to asthma and allergies of inhabitants, Indoor Air 24 **(2014)** 3–15. https://doi.org/10.1111/ina.12054.

[172] J.D. Meeker, E.M. Cooper, H.M. Stapleton, R. Hauser, Urinary metabolites of organophosphate flame retardants: Temporal variability and correlations with house dust

concentrations, Environ Health Perspect 121 **(2013)** 580–585. https://doi.org/10.1289/ehp.1205907.

[173] I. van der Veen, J. de Boer, Phosphorus flame retardants: Properties, production, environmental occurrence, toxicity and analysis, Chemosphere 88 **(2012)** 1119–1153. https://doi.org/10.1016/j.chemosphere.2012.03.067.

[174] D. Canbaz, M.J.M. van Velzen, E. Hallner, A.H. Zwinderman, M. Wickman, P.E.G. Leonards, R. van Ree, L.S. van Rijt, Exposure to organophosphate and polybrominated diphenyl ether flame retardants via indoor dust and childhood asthma, Indoor Air 26 **(2016)** 403–413. https://doi.org/10.1111/ina.12221.

[175] G. Meng, Z. Nie, Y. Feng, X. Wu, Y. Yin, Y. Wang, Typical halogenated persistent organic pollutants in indoor dust and the associations with childhood asthma in Shanghai, China, Environmental Pollution 211 **(2016)** 389–398. https://doi.org/10.1016/j.envpol.2015.12.006.

[176] G.G. Ying, Fate, behavior and effects of surfactants and their degradation products in the environment, Environ Int 32 **(2006)** 417–431. https://doi.org/10.1016/j.envint.2005.07.004.

[177] Å. Bergman, J.J. Heindel, S. Jobling, K.A. Kidd, R. Thomas Zoeller, State of the Science of Endocrine Disrupting Chemicals-**(2012)**. Inter-Organızatıon Programme For The Sound Management Of Chemıcals, N.D.

[178] W.-L. Ma, B. Subedi, K. Kannan, The Occurrence of Bisphenol A, Phthalates, Parabens and Other Environmental Phenolic Compounds in House. Data in Current Organic Chemistry ·, **(2015).** https://www.researchgate.net/publication/272793611.

[179] J. Li, L. Yang, Y. Gao, P. Jiang, Y. Li, T. Zhao, J. Zhang, W. Wang, Seasonal variations of NPAHs and OPAHs in PM 2.5 at heavily polluted urban and suburban sites in North China: Concentrations, molecular compositions, cancer risk assessments and sources, Ecotoxicol Environ Saf 178 **(2019)** 58–65. https://doi.org/10.1016/j.ecoenv.2019.04.009.

[180] U. Epa, N. Exposure Research Laboratory, EPA Positive M atrix Factorization (PM F) 5.0 Fundamentals and User Guide, n.d. www.epa.gov. (Accessed 19 May **2024**)

[181] Y.H. Lang, G.L. Li, X.M. Wang, P. Peng, Combination of Unmix and PMF receptor model to apportion the potential sources and contributions of PAHs in wetland

soils from Jiaozhou Bay, China, Mar Pollut Bull 90 **(2015)** 129–134. https://doi.org/10.1016/j.marpolbul.2014.11.009.

[182] J. Živančev, I. Antić, M. Buljovčić, N. Đurišić-Mladenović, A case study on the occurrence of polycyclic aromatic hydrocarbons in indoor dust of Serbian households: Distribution, source apportionment and health risk assessment, Chemosphere 295 **(2022).** https://doi.org/10.1016/j.chemosphere.2022.133856.

[183] J.K. Harris, Primer on binary logistic regression, Fam Med Community Health 9 **(2021)**. https://doi.org/10.1136/fmch-2021-001290.

[184] G. Meng, Y. Feng, Z. Nie, X. Wu, H. Wei, S. Wu, Y. Yin, Y. Wang, Internal exposure levels of typical POPs and their associations with childhood asthma in Shanghai, China, Environ Res 146 **(2016)** 125–135. https://doi.org/10.1016/j.envres.2015.12.026.

[185] A.J. Barros, V.N. Hirakata, Alternatives for logistic regression in cross-sectional studies: an empirical comparison of models that directly estimate the prevalence ratio Cox regressioncross-sectional studieslogistic regressionodds ratioPoisson regressionprevalence ratiorobust variancestatistical models, **2003.** http://www.biomedcentral.com/1471-2288/3/21.

[186] H. Qi, W.L. Li, N.Z. Zhu, W.L. Ma, L.Y. Liu, F. Zhang, Y.F. Li, Concentrations and sources of polycyclic aromatic hydrocarbons in indoor dust in China, Science of the Total Environment 491–492 **(2014)** 100–107. https://doi.org/10.1016/j.scitotenv.2014.01.119.

[187] Y. Kang, K.C. Cheung, M.H. Wong, Mutagenicity, genotoxicity and carcinogenic risk assessment of indoor dust from three major cities around the Pearl River Delta, Environ Int 37 **(2011)** 637–643. https://doi.org/10.1016/j.envint.2011.01.001.

[188] R.M. Maertens, X. Yang, J. Zhu, R.W. Gagne, G.R. Douglas, P.A. White, Mutagenic and carcinogenic hazards of settled house dust I: Polycyclic aromatic hydrocarbon content and excess lifetime cancer risk from preschool exposure, Environ Sci Technol 42 **(2008)** 1747–1753. https://doi.org/10.1021/es702449c.

[189] M. Lim, L. Morawska, G.A. Ayoko, M.C.H. Lim, Assessing Health Risk Associated with Airborne Polycyclic Aromatic Hydrocarbons by Chemometrics and Toxic Equivalency Factors, **(2005)**. http://eprints.qut.edu.au/.

[190] H. Fromme, T. Lahrz, M. Piloty, H. Gebhardt, A. Oddoy, H. Rüden, Polycyclic aromatic hydrocarbons inside and outside of apartments in an urban area, Science of the Total Environment 326 **(2004)** 143–149. https://doi.org/10.1016/j.scitotenv.2004.02.002.

[191] N. Ali, Polycyclic aromatic hydrocarbons (PAHs) in indoor air and dust samples of different Saudi microenvironments; health and carcinogenic risk assessment for the general population, Science of the Total Environment 696 **(2019)**. https://doi.org/10.1016/j.scitotenv.2019.133995.

[192] Kocaeli Üniversitesi Fen Bilimleri Enstitüsü Çevre Mühendisliği Anabilim Dalı Yüksek Lisans Tezi Kocaeli'de Evlerin İç Ortam Tozunda Pbde, Pcb Ve Pah, (Accessed 19 May **2024**)

[193] A. Besis, E. Botsaropoulou, D. Balla, D. Voutsa, C. Samara, Toxic organic pollutants in Greek house dust: Implications for human exposure and health risk, Chemosphere 284 **(2021)**. https://doi.org/10.1016/j.chemosphere.2021.131318.

[194] N. Ali, I.M.I. Ismail, M. Khoder, M. Shamy, M. Alghamdi, M. Costa, L.N. Ali, W. Wang, S.A.M.A.S. Eqani, Polycyclic aromatic hydrocarbons (PAHs) in indoor dust samples from Cities of Jeddah and Kuwait: Levels, sources and non-dietary human exposure, Science of the Total Environment 573 **(2016)** 1607–1614. https://doi.org/10.1016/j.scitotenv.2016.09.134.

[195] Ev İçi Ortamlarında Pah Kirletici Düzeyinin Tespiti Investıgatıon Of Pah Contamınant Level In Indoor Envıronments, (Accessed 19 May **2024)**

[196] H.Q. Anh, N.M. Tue, L.H. Tuyen, T.B. Minh, P.H. Viet, S. Takahashi, Polycyclic aromatic hydrocarbons and their methylated derivatives in settled dusts from end-of-life vehicle processing, urban, and rural areas, northern Vietnam: Occurrence, source apportionment, and risk assessment, Science of the Total Environment 672 **(2019)** 468–478. https://doi.org/10.1016/j.scitotenv.2019.04.018.

[197] T.P. Whitehead, C. Metayer, M. Petreas, M. Does, P.A. Buffler, S.M. Rappaport, Polycyclic aromatic hydrocarbons in residential dust: Sources of variability, Environ Health Perspect 121 **(2013)** 543–550. https://doi.org/10.1289/ehp.1205821.

[198] H. Stuart, C. Ibarra, M.A.E. Abdallah, R. Boon, H. Neels, A. Covaci, Concentrations of brominated flame retardants in dust from United Kingdom cars, homes,

and offices: Causes of variability and implications for human exposure, Environ Int 34 **(2008)** 1170–1175. https://doi.org/10.1016/j.envint.2008.05.001.

[199] W. Wang, M.J. Huang, J.S. Zheng, K.C. Cheung, M.H. Wong, Exposure assessment and distribution of polychlorinated biphenyls (PCBs) contained in indoor and outdoor dusts and the impacts of particle size and bioaccessibility, Science of the Total Environment 463–464 **(2013)** 1201–1209. https://doi.org/10.1016/j.scitotenv.2013.04.059.

[200] O. Audy, L. Melymuk, M. Venier, S. Vojta, J. Becanova, K. Romanak, M. Vykoukalova, R. Prokes, P. Kukucka, M.L. Diamond, J. Klanova, PCBs and organochlorine pesticides in indoor environments - A comparison of indoor contamination in Canada and Czech Republic, Chemosphere 206 **(2018)** 622–631. https://doi.org/10.1016/j.chemosphere.2018.05.016.

[201] C.T. He, X.B. Zheng, X. Yan, J. Zheng, M.H. Wang, X. Tan, L. Qiao, S.J. Chen, Z.Y. Yang, B.X. Mai, Organic contaminants and heavy metals in indoor dust from e-waste recycling, rural, and urban areas in South China: Spatial characteristics and implications for human exposure, Ecotoxicol Environ Saf 140 **(2017)** 109–115. https://doi.org/10.1016/j.ecoenv.2017.02.041.

[202] N. Ali, L. Ali, T. Mehdi, A.C. Dirtu, F. Al-Shammari, H. Neels, A. Covaci, Levels and profiles of organochlorines and flame retardants in car and house dust from Kuwait and Pakistan: Implication for human exposure via dust ingestion, Environ Int 55 **(2013)** 62–70. https://doi.org/10.1016/j.envint.2013.02.001.

[203] P.B. Kurt-Karakus, H. Alegria, L. Jantunen, A. Birgul, A. Topcu, K.C. Jones, C. Turgut, Polybrominated diphenyl ethers (PBDEs) and alternative flame retardants (NFRs) in indoor and outdoor air and indoor dust from Istanbul-Turkey: Levels and an assessment of human exposure, Atmos Pollut Res 8 **(2017)** 801–815. https://doi.org/10.1016/j.apr.2017.01.010.

[204] M. Shoeib, T. Harner, G.M. Webster, E. Sverko, Y. Cheng, Legacy and current-use flame retardants in house dust from Vancouver, Canada, Environmental Pollution 169 **(2012)** 175–182. https://doi.org/10.1016/j.envpol.2012.01.043.

[205] M.Y. Civan, U.M. Kara, Risk assessment of PBDEs and PAHs in house dust in Kocaeli, Turkey: levels and sources, Environmental Science and Pollution Research 23 **(2016)** 23369–23384. https://doi.org/10.1007/s11356-016-7512-5.

[206] B. Gevao, M. Al-Bahloul, A.N. Al-Ghadban, A. Al-Omair, L. Ali, J. Zafar, M. Helaleh, House dust as a source of human exposure to polybrominated diphenyl ethers in Kuwait, Chemosphere 64 **(2006)** 603–608. https://doi.org/10.1016/j.chemosphere.2005.11.055.

[207] R.A. Rudel, L.J. Perovich, Endocrine disrupting chemicals in indoor and outdoor air, Atmos Environ 43 **(2009)** 170–181. https://doi.org/10.1016/j.atmosenv.2008.09.025.

[208] X. Lu, M. Chen, X. Zhang, Y. Sun, D. Zhu, Q. Zhang, B. Wang, Z. Zhang, Simultaneous quantification of five phenols in settled house dust using ultra-high performance liquid chromatography-tandem mass spectrometry, Analytical Methods 5 **(2013)** 5339–5344. https://doi.org/10.1039/c3ay40602d.

[209] Levels and human health risk of polycyclic aromatic hydrocarbons (PAHs) in indoor dust in Hanoi, Vietnam, Journal of Science and Technology - HaUI 59 **(2023).** https://doi.org/10.57001/huih5804.2023.231.

[210] A.Q. Hoang, S. Takahashi, L.H. Tuyen, N.M. Tue, N.M. Tu, T.T.T. Nguyen, M.B. Tu, Polycyclic Aromatic Hydrocarbons in Air and Dust Samples from Vietnamese End-of-life Vehicle Processing Workshops: Contamination Status, Sources, and Exposure Risks, Bull Environ Contam Toxicol 110 **(2023).** https://doi.org/10.1007/s00128-023-03757-x.

[211] P. Baumard, H. Budzinski, P. Garrigues, H. Dizer, P.D. Hansen, Baltic Sea in March, **(1995).** (Accessed 19 May 2024)

[212] H. Budzinski, I. Jones, J. Bellocq, C. Picrard, P. Garrigues, Evaluation of sediment contamination by polycyclic aromatic hydrocarbons in the Gironde estuary, **(1997).** (Accessed 19 May 2024)

[213] B. Mai, S. Qi, E.Y. Zeng, Q. Yang, G. Zhang, J. Fu, G. Sheng, P. Peng, Z. Wang, Distribution of Polycyclic Aromatic Hydrocarbons in the Coastal Region off Macao, China: Assessment of Input Sources and Transport Pathways Using Compositional Analysis, Environ Sci Technol 37 **(2003)** 4855–4863. https://doi.org/10.1021/es034514k.

[214] S.S. Park, Y.J. Kim, C.H. Kang, Atmospheric polycyclic aromatic hydrocarbons in Seoul, Korea, **(2002).**

[215] M. Živković, M. Jovašević-Stojanović, A. Cvetković, I. Lazović, V. Tasić, Ž. Stevanović, I. Gržetić, Nivo policikličnih aromatičnih ugljovodonika u gasnoj i čestičnoj

fazi u školama na različitim lokacijama u Srbiji, Chemical Industry and Chemical Engineering Quarterly 21 **(2015)** 159–167. https://doi.org/10.2298/CICEQ140206016Z.

[216] I.C. Yadav, N.L. Devi, V.K. Singh, J. Li, G. Zhang, Concentrations, sources and health risk of nitrated- and oxygenated-polycyclic aromatic hydrocarbon in urban indoor air and dust from four cities of Nepal, Science of the Total Environment 643 **(2018)** 1013–1023. https://doi.org/10.1016/j.scitotenv.2018.06.265.

[217] P. Chaber, B. Gworek, Surface horizons of forest soils for the diagnosis of soil environment contamination and toxicity caused by polycyclic aromatic hydrocarbons (PAHs), PLoS One 15 **(2020).** https://doi.org/10.1371/journal.pone.0231359.

[218] E. Manoli, A. Kouras, O. Karagkiozidou, G. Argyropoulos, D. Voutsa, C. Samara, Polycyclic aromatic hydrocarbons (PAHs) at traffic and urban background sites of northern Greece: source apportionment of ambient PAH levels and PAH-induced lung cancer risk, Environmental Science and Pollution Research 23 **(2016)** 3556–3568. https://doi.org/10.1007/s11356-015-5573-5.

[219] S.C. Edwards, W. Jedrychowski, M. Butscher, D. Camann, A. Kieltyka, E. Mroz, E. Flak, Z. Li, S. Wang, V. Rauh, F. Perera, Prenatal exposure to airborne polycyclic aromatic hydrocarbons and children's intelligence at 5 years of age in a prospective cohort study in Poland, Environ Health Perspect 118 **(2010)** 1326–1331. https://doi.org/10.1289/ehp.0901070.

[220] G.J. Liu, Z.Y. Niu, J. Xue, L.G. Zheng, D. Van Niekerk, G. Liu, Z. Niu, D. Van Niekerk, J. Xue, L. Zheng, Polycyclic Aromatic Hydrocarbons (PAHs) from Coal Combustion: Emissions, Analysis, and Toxicology, **(2008).**

[221] C. hui Wang, S. hua Wu, S. lu Zhou, H. Wang, B. jie Li, H. Chen, Y. na Yu, Y. xing Shi, Polycyclic aromatic hydrocarbons in soils from urban to rural areas in Nanjing: Concentration, source, spatial distribution, and potential human health risk, Science of the Total Environment 527–528 **(2015)** 375–383. https://doi.org/10.1016/j.scitotenv.2015.05.025.

[222] M.M. Mahfouz, H.M. Hassan, E.A. Elobaid, O. Yigiterhan, B. Alfoldy, PAH concentrations and exposure assessment from house dust retained in air-conditioning filters collected from Greater Doha, Qatar, Environ Geochem Health 41 **(2019)** 2251–2263. https://doi.org/10.1007/s10653-019-00271-0.

[223] Q. Yang, H. Chen, B. Li, Polycyclic aromatic hydrocarbons (PAHs) in indoor dusts of guizhou, southwest of china: Status, sources and potential human health risk, PLoS One 10 **(2015).** https://doi.org/10.1371/journal.pone.0118141.

[224] K. Ren, Y. Wei, J. Li, C. Han, Y. Deng, G. Su, Polycyclic aromatic hydrocarbons (PAHs) and their derivatives (oxygenated PAHs, azaarenes, and sulfur / oxygen-containing heterocyclic PAHs) in surface soils from a typical city, south China, Chemosphere 283 **(2021).** https://doi.org/10.1016/j.chemosphere.2021.131190.

[225] T.T.T. Dong, B.K. Lee, Characteristics, toxicity, and source apportionment of polycylic aromatic hydrocarbons (PAHs) in road dust of Ulsan, Korea, Chemosphere 74 **(2009)** 1245–1253. https://doi.org/10.1016/j.chemosphere.2008.11.035.

[226] L. Zhu, H. Lu, S. Chen, T. Amagai, Pollution level, phase distribution and source analysis of polycyclic aromatic hydrocarbons in residential air in Hangzhou, China, J Hazard Mater 162 **(2009)** 1165–1170. https://doi.org/10.1016/j.jhazmat.2008.05.150.

[227] J. Gao, Y. Jian, C. Cao, L. Chen, X. Zhang, Indoor emission, dispersion and exposure of total particle-bound polycyclic aromatic hydrocarbons during cooking, Atmos Environ 120 **(2015)** 191–199. https://doi.org/10.1016/j.atmosenv.2015.08.030.

[228] Y.J. Hu, L.J. Bao, C.L. Huang, S.M. Li, P. Liu, E.Y. Zeng, Assessment of airborne polycyclic aromatic hydrocarbons in a megacity of South China: Spatiotemporal variability, indoor-outdoor interplay and potential human health risk, Environmental Pollution 238 **(2018)** 431–439. https://doi.org/10.1016/j.envpol.2018.03.040.

[229] B. Strandberg, C. Österman, H. Koca Akdeva, J. Moldanová, S. Langer, The Use of Polyurethane Foam (PUF) Passive Air Samplers in Exposure Studies to PAHs in Swedish Seafarers, Polycycl Aromat Compd 42 **(2022)** 448–459. https://doi.org/10.1080/10406638.2020.1739084.

[230] H. Lu, L. Zhu, S. Chen, Pollution level, phase distribution and health risk of polycyclic aromatic hydrocarbons in indoor air at public places of Hangzhou, China, Environmental Pollution 152 **(2008)** 569–575. https://doi.org/10.1016/j.envpol.2007.07.005.

[231] X. Wan, J. Chen, F. Tian, W. Sun, F. Yang, K. Saiki, Source apportionment of PAHs in atmospheric particulates of Dalian: Factor analysis with nonnegative constraints

and emission inventory analysis, Atmos Environ 40 **(2006)** 6666–6675. https://doi.org/10.1016/j.atmosenv.2006.05.049.

[232] H. Sharma, V.K. Jain, Z.H. Khan, Characterization and source identification of polycyclic aromatic hydrocarbons (PAHs) in the urban environment of Delhi, Chemosphere 66 **(2007)** 302–310. https://doi.org/10.1016/j.chemosphere.2006.05.003.

[233] Smoking Increases Carcinogenic Polycyclic Aromatic Hydrocarbons in Human Lung Tissue 1, (**2001)**. http://aacrjournals.org/cancerres/article-pdf/61/17/6367/2487050/ch1701006367.pdf.

[234] A. Cachada, L. V. Lopes, A.S. Hursthouse, M. Biasioli, H. Grčman, E. Otabbong, C.M. Davidson, A.C. Duarte, The variability of polychlorinated biphenyls levels in urban soils from five European cities, Environmental Pollution 157 **(2009)** 511–518. https://doi.org/10.1016/j.envpol.2008.09.002.

[235] H. Stuart, C. Ibarra, M.A.E. Abdallah, R. Boon, H. Neels, A. Covaci, Concentrations of brominated flame retardants in dust from United Kingdom cars, homes, and offices: Causes of variability and implications for human exposure, Environ Int 34 **(2008)** 1170–1175. https://doi.org/10.1016/j.envint.2008.05.001.

[236] R.F. Herrick, M.D. McClean, J.D. Meeker, L.K. Baxter, G.A. Weymouth, An unrecognized source of PCB contamination in schools and other buildings, Environ Health Perspect 112 **(2004)** 1051–1053. https://doi.org/10.1289/ehp.6912.

[237] S.D. Coelho, A.C.A. Sousa, T. Isobe, J.W. Kim, T. Kunisue, A.J.A. Nogueira, S. Tanabe, Brominated, chlorinated and phosphate organic contaminants in house dust from Portugal, Science of the Total Environment 569–570 **(2016)** 442–449. https://doi.org/10.1016/j.scitotenv.2016.06.137.

[238] L. Knobeloch, P. Imm, H. Anderson, Perfluoroalkyl chemicals in vacuum cleaner dust from 39 Wisconsin homes, Chemosphere 88 **(2012)** 779–783. https://doi.org/10.1016/j.chemosphere.2012.03.082.

[239] H.V. Andersen, L. Gunnarsen, L.E. Knudsen, M. Frederiksen, PCB in air, dust and surface wipes in 73 Danish homes, Int J Hyg Environ Health 229 **(2020).** https://doi.org/10.1016/j.ijheh.2019.113429.

[240] J. Tan, S.M. Cheng, A. Loganath, Y.S. Chong, J.P. Obbard, Selected organochlorine pesticide and polychlorinated biphenyl residues in house dust in

Singapore, Chemosphere 68 **(2007)** 1675–1682. https://doi.org/10.1016/j.chemosphere.2007.03.051.

[241] T.P. Whitehead, F.R. Brown, C. Metayer, J.S. Park, M. Does, J. Dhaliwal, M.X. Petreas, P.A. Buffler, S.M. Rappaport, Polychlorinated biphenyls in residential dust: Sources of variability, Environ Sci Technol 48 **(2014)** 157–164. https://doi.org/10.1021/es403863m.

[242] G. Salihoglu, N.K. Salihoglu, E. Aksoy, Y. Tasdemir, Spatial and temporal distribution of polychlorinated biphenyl (PCB) concentrations in soils of an industrialized city in Turkey, J Environ Manage 92 **(2011)** 724–732. https://doi.org/10.1016/j.jenvman.2010.10.019.

[243] X. Zhang, M.L. Diamond, M. Robson, S. Harrad, Sources, emissions, and fate of polybrominated diphenyl ethers and polychlorinated biphenyls indoors in Toronto, Canada, Environ Sci Technol 45 **(2011)** 3268–3274. https://doi.org/10.1021/es102767g.

[244] H.V. Andersen, L. Gunnarsen, L.E. Knudsen, M. Frederiksen, PCB in air, dust and surface wipes in 73 Danish homes, Int J Hyg Environ Health 229 **(2020).** https://doi.org/10.1016/j.ijheh.2019.113429.

[245] R.G.M. Lee, P. Coleman, J.L. Jones, K.C. Jones, R. Lohmann, Emission factors and importance of PCDD/Fs, PCBs, PCNs, PAHs and PM 10 from the domestic burning of coal and wood in the U.K., Environ Sci Technol 39 **(2005)** 1436–1447. https://doi.org/10.1021/es048745i.

[246] H.Q. Anh, I. Watanabe, T.B. Minh, N.M. Tue, L.H. Tuyen, P.H. Viet, S. Takahashi, Polychlorinated biphenyls in settled dusts from an end-of-life vehicle processing area and normal house dusts in northern Vietnam: Occurrence, potential sources, and risk assessment, Science of the Total Environment 728 **(2020).** https://doi.org/10.1016/j.scitotenv.2020.138823.

[247] S. Kemmlein, D. Herzke, R.J. Law, Brominated flame retardants in the European chemicals policy of REACH-Regulation and determination in materials, J Chromatogr A 1216 **(2009)** 320–333. https://doi.org/10.1016/j.chroma.2008.05.085.

[248] M.J. La Guardia, R.C. Hale, E. Harvey, Detailed polybrominated diphenyl ether (PBDE) congener composition of the widely used penta-, octa-, and deca-PBDE technical

flame-retardant mixtures, Environ Sci Technol 40 **(2006)** 6247–6254. https://doi.org/10.1021/es060630m.

[249] M.Y. Civan, U.M. Kara, Risk assessment of PBDEs and PAHs in house dust in Kocaeli, Turkey: levels and sources, Environmental Science and Pollution Research 23 **(2016)** 23369–23384. https://doi.org/10.1007/s11356-016-7512-5.

[250] P.B. Kurt-Karakus, H. Alegria, L. Jantunen, A. Birgul, A. Topcu, K.C. Jones, C. Turgut, Polybrominated diphenyl ethers (PBDEs) and alternative flame retardants (NFRs) in indoor and outdoor air and indoor dust from Istanbul-Turkey: Levels and an assessment of human exposure, Atmos Pollut Res 8 **(2017)** 801–815. https://doi.org/10.1016/j.apr.2017.01.010.

[251] A. Sjödin, O. Päpke, E. McGahee, J.F. Focant, R.S. Jones, T. Pless-Mulloli, L.M.L. Toms, T. Herrmann, J. Müller, L.L. Needham, D.G. Patterson, Concentration of polybrominated diphenyl ethers (PBDEs) in household dust from various countries, Chemosphere 73 **(2008).** https://doi.org/10.1016/j.chemosphere.2007.08.075.

[252] A.R. Zota, J.S. Park, Y. Wang, M. Petreas, R.T. Zoeller, T.J. Woodruff, Polybrominated diphenyl ethers, hydroxylated polybrominated diphenyl ethers, and measures of thyroid function in second trimester pregnant women in California, Environ Sci Technol 45 **(2011)** 7896–7905. https://doi.org/10.1021/es200422b.

[253] H.M. Stapleton, N.G. Dodder, J.H. Offenberg, M.M. Schantz, S.A. Wise, Polybrominated diphenyl ethers in house dust and clothes dryer lint, Environ Sci Technol 39 **(2005)** 925–931. https://doi.org/10.1021/es0486824.

[254] J. Cristale, A. Hurtado, C. Gómez-Canela, S. Lacorte, Occurrence and sources of brominated and organophosphorus flame retardants in dust from different indoor environments in Barcelona, Spain, Environ Res 149 **(2016)** 66–76. https://doi.org/10.1016/j.envres.2016.05.001.

[255] T.H. Kim, Y.J. Lee, E. Lee, N. Patra, J. Lee, S.J. Kwack, K.B. Kim, K.K. Chung, S.Y. Han, J.Y. Han, B.M. Lee, H.S. Kim, Exposure assessment of polybrominated diphenyl ethers (PBDE) in umbilical cord blood of Korean infants, Journal of Toxicology and Environmental Health - Part A: Current Issues 72 **(2009)** 1318–1326. https://doi.org/10.1080/15287390903212436.

[256] R.E. Dodson, M. Nishioka, L.J. Standley, L.J. Perovich, J.G. Brody, R.A. Rudel, Endocrine disruptors and asthma-associated chemicals in consumer products, Environ Health Perspect 120 **(2012)** 935–943. https://doi.org/10.1289/ehp.1104052.

[257] W.-L. Ma, B. Subedi, K. Kannan, The Occurrence of Bisphenol A, Phthalates, Parabens and Other Environmental Phenolic Compounds in House Dust: A Review, **(2014)**. https://www.researchgate.net/publication/272793611.

[258] Y. Liu, X. Dai, J. Wei, Toxicity of the xenoestrogen nonylphenol and its biodegradation by the alga Cyclotella caspia, J Environ Sci (China) 25 **(2013)** 1662–1671. https://doi.org/10.1016/S1001-0742(12)60182-X.

[259] A. Priac, Lactuca sativa L. : Evaluation écotoxicologique de rejets industriels complexes et de solutions synthétiques, n.d. https://theses.hal.science/tel-01228010. (Accessed 19 May **2024**)

[260] H. Cabana, J.P. Jones, S.N. Agathos, Elimination of endocrine disrupting chemicals using white rot fungi and their lignin modifying enzymes: A review, Eng Life Sci 7 **(2007)** 429–456. https://doi.org/10.1002/elsc.200700017.

[261] C. Liao, K. Kannan, A survey of alkylphenols, bisphenols, and triclosan in personal care products from China and the United States, Arch Environ Contam Toxicol 67 **(2014)** 50–59. https://doi.org/10.1007/s00244-014-0016-8.

[262] Annex Xv Restrıctıon Report-Nonylphenol And Nonylphenoletoxylates In Textıles Annex Xv Restrıctıon Report Proposal For A Restrıctıon Nonylphenol And Nonylphenolethoxylates In Textıles Substance Name(S): Nonylphenol And Nonylphenol Ethoxylate, N.D. www.kemikalieinspektionen.se. (Accessed 19 May **2024)**

[263] C. Lassen, Danmark. Miljøstyrelsen, Survey of alkylphenols and alkylphenol ethoxylates : part of the LOUS-review, Environmental Protection Agency, **(2013).**

[264] K. Lamprea, A. Bressy, C. Mirande-Bret, E. Caupos, M.C. Gromaire, Alkylphenol and bisphenol A contamination of urban runoff: an evaluation of the emission potentials of various construction materials and automotive supplies, Environmental Science and Pollution Research 25 **(2018)** 21887–21900. https://doi.org/10.1007/s11356-018-2272-z.

[265] M.F. Saputra, D. Rifa, N. Rahmawati, Pengaruh corporate governance, profitabilitas dan karakter eksekutif terhadap tax avoidance pada perusahaan yang

terdaftar di BEI, Jurnal Akuntansi & Auditing Indonesia 19 **(2015)** 1–12. https://doi.org/10.20885/jaai.vol19.iss1.art1.

[266] B. Thiele, K. Günther, M.J. Schwuger, Alkylphenol Ethoxylates: Trace Analysis and Environmental Behavior, **1994.** https://pubs.acs.org/sharingguidelines. (Accessed 19 May 2024)

[267] C. Karakaş, Substance Flow Analysıs Of Nonylphenol And Nonylphenol Ethoxylates In Turkey A Thesıs Submıtted To The Graduate School Of Natural And Applıed Scıences Of Mıddle East Technıcal Unıversıty, **(2014).**

[268] K. Lamprea, A. Bressy, C. Mirande-Bret, E. Caupos, M.C. Gromaire, Alkylphenol and bisphenol A contamination of urban runoff: an evaluation of the emission potentials of various construction materials and automotive supplies, Environmental Science and Pollution Research 25 **(2018)** 21887–21900. https://doi.org/10.1007/s11356-018-2272-z.

[269] I. Saito, A. Onuki, H. Seto, Indoor air pollution by alkylphenols in Tokyo, Indoor Air 14 **(2004)** 325–332. https://doi.org/10.1111/j.1600-0668.2004.00250.x.

[270] RESEARCH AND PRACTICE , (Accessed 19 May **2024**)

[271] J.U. Lee, J.D. Kim, C.S. Park, Gene-environment interactions in asthma: Genetic and epigenetic effects, Yonsei Med J 56 **(2015)** 877–886. https://doi.org/10.3349/ymj.2015.56.4.877.

[272] A. Papi, C. Brightling, S.E. Pedersen, H.K. Reddel, Asthma, The Lancet 391 **(2018)** 783–800. https://doi.org/10.1016/S0140-6736(17)33311-1.

[273] H. Liu, C. Xu, Z.Y. Jiang, A. Gu, Association of polycyclic aromatic hydrocarbons and asthma among children 6-19 years: NHANES 2001-2008 and NHANES 2011-2012, Respir Med 110 **(2016)** 20–27. https://doi.org/10.1016/j.rmed.2015.11.003.

[274] S.L. Gale, E.M. Noth, J. Mann, J. Balmes, S.K. Hammond, I.B. Tager, Polycyclic aromatic hydrocarbon exposure and wheeze in a cohort of children with asthma in Fresno, CA, J Expo Sci Environ Epidemiol 22 **(2012)** 386–392. https://doi.org/10.1038/jes.2012.29.

[275] Peptide Arrays Break the Species Barrier, (Accessed 19 May **2024**)

[276] S. Hansen, M. Strøm, S.F. Olsen, E. Maslova, P. Rantakokko, H. Kiviranta, D. Rytter, B.H. Bech, L. V. Hansen, T.I. Halldorsson, Maternal concentrations of persistent organochlorine pollutants and the risk of asthma in offspring: Results from a prospective cohort with 20 years of follow-up, Environ Health Perspect 122 **(2014)** 93–99. https://doi.org/10.1289/ehp.1206397.

[277] M. Parker-Lalomio, K. McCann, J. Piorkowski, S. Freels, V.W. Persky, Prenatal exposure to polychlorinated biphenyls and asthma, eczema/hay fever, and frequent ear infections, Journal of Asthma 55 **(2018)** 1105–1115. https://doi.org/10.1080/02770903.2017.1396470.

[278] A. Mamane, C. Raherison, J.F. Tessier, I. Baldi, G. Bouvier, Environmental exposure to pesticides and respiratory health, European Respiratory Review 24 **(2015)** 462–473. https://doi.org/10.1183/16000617.00006114.

[279] N. Li, C. Sioutas, A. Cho, D. Schmitz, C. Misra, J. Sempf, M. Wang, T. Oberley, J. Froines, A. Nel, Ultrafine particulate pollutants induce oxidative stress and mitochondrial damage, Environ Health Perspect 111 **(2003)** 455–460. https://doi.org/10.1289/ehp.6000.

[280] M.T. Salam, P.C. Lin, E.L. Avol, W.J. Gauderman, F.D. Gilliland, Microsomal epoxide hydrolase, glutathione S-transferase P1, traffic and childhood asthma, Thorax 62 **(2007)** 1050–1057. https://doi.org/10.1136/thx.2007.080127.

[281] K. Nadeau, C. McDonald-Hyman, E.M. Noth, B. Pratt, S.K. Hammond, J. Balmes, I. Tager, Ambient air pollution impairs regulatory T-cell function in asthma, Journal of Allergy and Clinical Immunology 126 **(2010).** https://doi.org/10.1016/j.jaci.2010.08.008.

[282] Y. Nakanishi, N. Shigematsu, Y. Kurita, K. Matsuba, H. Kanegae, S. Ishimaru, Y. Kawazoe, Respiratory Involvement and Immune Status in Yusho Patients, **(1985).** (Accessed 20 May 2024)

APPENDIX D

Peer Reviewed Journals

Parisa Babaei, Efsun Nikravan Madan, Gülen Güllü, Ismail Ethem Goren, Hatice Kübra Gül, Nebile Daglıoglu, Perihan Binnur Kurt Karakus, Levels, distribution, sources and human exposure pathways of alkylphenol and alkylphenol ethoxylates in indoor dust in Turkiye, Environmental Pollution 344 (2024) 123447, https://doi.org/10.1016/j.envpol.2024.123447

APPENDIX E

Conference proceedıngs

Parisa Babaei, Afsoun Nikravan, Gülen Güllü, Okul Çağı Çocukların Evlerinde Kalıcı Organik Kirletici Seviyelerinin Belirlenmesi: Vaka Kontrol Çalışması, 15. ULUSAL TESİSAT MÜHENDİSLİĞİ KONGRESİ // 26-29 NİSAN 2023 / İZMİR, Oral Presentation, The full text of the study was published.